ICES COOPERATIVE RESEARCH REPORT

RAPPORT DES RECHERCHES COLLECTIVES

NO. 262

Report of the ICES Advisory Committee on Ecosystems, 2003

Copenhagen, 19–23 May 2003

International Council for the Exploration of the Sea

Conseil International pour l'Exploration de la Mer

Palægade 2–4 DK-1261 Copenhagen K Denmark

December 2003

For purposes of citation, the 2003 ACE Report should be cited as follows:

ICES. 2003. Report of the ICES Advisory Committee on Ecosystems, 2003. ICES Cooperative Research Report, 262. 229 pp.

ISBN 87-7482-006-0
ISSN 1017-6195

TABLE OF CONTENTS

Section **Page**

TABLE OF CONTENTS

TABLE OF CONTENTS

Section **Page**

TABLE OF CONTENTS

Section **Page**

TABLE OF CONTENTS

TABLE OF CONTENTS

Section **Page**

ICES ADVISORY COMMITTEE ON ECOSYSTEMS
LIST OF MEMBERS

19–23 May 2003

Participant	Affiliation
H.R. Skjoldal	Chair
P. Keizer	Chair, Marine Habitat Committee
S. Walsh	Chair, Fisheries Technology Committee
B. Mackenzie	Chair, Baltic Committee
P. Degnbol	Chair, Advisory Committee on Fishery Management
J. Haelters	Belgium
J. Rice	Canada
M. Vinther	Denmark
R. Aps	Estonia
M. Rask	Finland
J. Boucher	France
K. Hermann Kock	Germany
K. Gunnarson	Iceland
T. McMahon	Ireland
A. Andrushaitis	Latvia
N. Daan	The Netherlands
S. Tjelmeland	Norway
P. Margonski	Poland
M. Borges	Portugal
A. Krovnin	Russia
S. Lens	Spain
M. Sköld	Sweden
S. Jennings	United Kingdom
T. Noji	United States
M. Tasker	Chair, SGCOR
C. Frid	Chair, WGECO
J. Weissenberger	Observer, European Commission
O. Hagström	Observer, European Commission
J. Pawlak	ICES Environment Adviser
K. Brander	ICES GLOBEC Coordinator
H. Sparholt	ICES Fishery Assessment Scientist

EXECUTIVE SUMMARY

The ICES Advisory Committee on Ecosystems (ACE) met from 19 to 23 May 2003. During this meeting, ACE prepared an initial response to the request from the European Commission Directorate General for Fisheries concerning the ecosystem impacts of industrial fishing. ACE also prepared advice, in addition to that in the 2002 ACE report, in response to EC requests on the by-catch of small cetaceans in fisheries and on the occurrence of cold-water corals that may be impacted by fisheries, as well as providing some further advice on other issues of concern to the EC in relation to the impacts of fishing on the ecosystem. Furthermore, ACE provided responses to requests from the Helsinki Commission on the status of populations of marine mammals in the Baltic marine area and advice on monitoring programmes to estimate the abundance of seals and other marine mammals in the Baltic Sea; ACE also provided brief additional material in relation to a request on marine habitat classification. In response to requests from the OSPAR Commission, ACE has prepared an extensive review, and advice for further development, of four of the Ecological Quality Objectives in the Pilot Project for the North Sea, as well as initial consideration of nine other Ecological Quality Elements that are not part of this Pilot Project. ACE also completed its review, begun in 2002, of the evidence for the justification for the proposed OSPAR Priority List of Threatened and Declining Species and Habitats.

ADVICE IN RELATION TO MARINE MAMMALS

Status of populations of marine mammals in the Baltic marine area

ACE has prepared a triennial review of the status of marine mammal populations in the Baltic marine area including, where possible, an evaluation of the impact of human activities on these populations (Section 2 of this report). The populations and area of distribution of all species of marine mammals in the Baltic Sea are considerably smaller than they were a century ago, with harbour porpoise and grey and ringed seal populations a fraction of their pristine levels. Human activities, including fishing, hunting of seals, and the discharge of contaminants, have all impacted these populations. Currently, there is inadequate data on the by-catch of all three species of seals in the Baltic Sea, but by-catch is the most important cause of death of grey seals. The health status of grey seals is not normal, with colonic ulcers an important cause of mortality. ICES advises that the present population of grey seals is not in a favourable conservation status and the extra mortality caused by humans should be reduced. For harbour porpoises, ICES supports the recommendations of the ASCOBANS recovery plan for the Baltic Sea.

ACE has also provided advice on survey methods to estimate populations of harbour porpoises and the three species of seals in the Baltic Sea (Section 3 of this report). Given the very low abundance of harbour porpoises in this area, a step-wise approach is recommended to obtain information on their abundance.

By-catch of small cetaceans in fisheries

As a supplement to information and advice prepared in 2002, ACE has provided new information on the by-catch of small cetaceans in fishing gear in fisheries in the Northeast Atlantic (Section 4 of this report). Due to the limited amount of new information, ICES does not provide new advice but reiterates its 2002 recommendations concerning the by-catch of cetaceans in fisheries. ICES also endorses the ASCOBANS "Jastarnia Plan" for the recovery of the harbour porpoise in the Baltic Sea.

THREATENED AND DECLINING SPECIES AND HABITATS

ACE completed a review, begun in 2002, of the evidence to support the nominations of species and habitats to the OSPAR Priority List of Threatened and Declining Species and Habitats, supplementing the advice provided in 2002. The review in 2003 covers several non-commercial species of fish, seahorses, turtles, and marine mammals (Section 5 and Annex 1 of this report).

ECOLOGICAL QUALITY OBJECTIVES

ACE reviewed general aspects of the Ecological Quality–Ecological Quality Objective (EcoQ–EcoQO) framework and prepared a template for the review of the Pilot Project on EcoQOs in the North Sea by ICES Working Groups in 2004 (Section 6 of this report). ACE then considered the individual EcoQ elements in the OSPAR request, and prepared a detailed review of and advice on the four EcoQ elements, out of the ten selected for the North Sea Pilot Project, for which ICES has been requested to provide further development and advice in 2003. Initial consideration and advice have been provided for nine of the EcoQ elements that are not part of the Pilot Project, but for which OSPAR has requested advice on their further development.

EcoQOs for the four EcoQ elements on the North Sea Pilot Project, i.e, EcoQ element (a) Spawning stock biomass of commercial fish species, EcoQ element (c) Seal population trends in the North Sea, EcoQ element (e) By-catch of harbour porpoises, and EcoQ element (f) Proportion of oiled common guillemots among those found dead or dying on beaches, were subjected, to the extent possible, to a performance analysis using the approach of signal-detection and decision theory, as a means of determining their value in a management context. Based on these analyses, advice is provided on the further development of these EcoQOs.

A review of the other EcoQ elements and EcoQOs shows that some are in an advanced stage of development, while others may prove difficult to implement in a useful way. Advice is given regarding further development of these EcoQs.

HABITAT ISSUES

Marine habitat classification and mapping

Progress in the development of marine habitat classification systems, notably the EUNIS classification, and in marine habitat mapping was reviewed (Section 7 of this report). ICES recommends the continued development of classification systems, including continued development of the EUNIS system. ICES endorsed the OSPAR mapping initiative as well as data management initiatives to archive metadata held by relevant agencies, as they are critical to the development of habitat maps and associated information. With respect to the Baltic Sea, this is a dynamic environment and the development of a classification scheme for this area will be best achieved through a dedicated project.

Cold-water corals in the North Atlantic and their relation to fisheries in the Northeast Atlantic

As a follow-up to information on the distribution of cold-water corals, mainly *Lophelia pertusa*, in the Northeast Atlantic presented in the 2002 ACE report, new information is provided in Section 8, now also covering certain areas in the Northwest Atlantic. Further information on the precise location of cold-water corals, as well as on the fishing pressure in these areas, is needed to best tailor advice on the protection of these corals from fishing activities.

Sensitive habitats in relation to fishing activities

In Section 9 of this report, further consideration is given to the issue of sensitive habitats in relation to the impact of current fishing practices, following on from the material in last year's report. After a brief review of sensitive habitats and their general distribution in the ICES area, ACE provides two case studies on approaches to the application of mitigation in contrasting sensitive habitats: 1) on *Lophelia pertusa* reefs, and 2) in seapen and burrowing megafauna biotopes.

RELATIVE IMPORTANCE OF EXTRINSIC FACTORS ON FISH POPULATION DYNAMICS COMPARED TO FISHING

ACE evaluated the relative impact of four factors (fishing, predation, oceanographic conditions/climate variation, and pollution) on three main processes in fish population dynamics: mortality, recruitment, and growth (Section 10 of this report). In general, for mortality of "adult" fish of commercially exploited species, the effect of fishing usually dominates all others. For recruitment, predation and oceanographic conditions are the dominant factors. The effects of oceanographic conditions are usually the dominant factor in recruitment and appear to be a more dominant factor for pelagic species than for demersal species. The relative contribution of the four factors to growth was less clear.

POTENTIAL IMPACTS OF CURRENT FISHING PRACTICES

Ecosystem impacts of industrial fishing

ACE reviewed the scope of industrial fisheries in the Northeast Atlantic and their impacts (Section 11 of this report). This review included both direct effects on the industrial fish species themselves and on fish species for human consumption taken as by-catch, as well as indirect effects on both the prey and the predators (fish, seabirds, and marine mammals) of industrial fish species. Based on these analyses, the impacts of industrial fishing that have been identified are relatively small in comparison with the effects of directed fisheries for human consumption species. However, information is lacking in a number of areas, particularly for the large blue whiting fishery in the Atlantic, for which there is no information on the catch rates of other species or on the food-web effects of fishing for blue whiting. Also, studies of the local interactions between industrial fisheries and foraging seabirds or predatory fish are still required in areas where industrial fisheries occur and such relationships have not already been considered.

Preservation of genetic diversity of exploited stocks

Additional information to that provided in 2002 on factors relevant to the preservation of the genetic diversity of exploited fish stocks is presented in Section 13. This section considers management objectives for maintaining genetic diversity within a species and provides further advice to meet these objectives, including a proposed prioritization scheme to assist with the decision on which populations to protect.

Ecological dependence in fisheries management advice

A further response, in addition to that provided in 2002, is given to a request concerning consideration of ecological dependence in management advice (Section 14). This includes a brief description of approaches for assessing potential ecological dependence and the strength of such dependence.

FRAMEWORK FOR THE PROVISION OF INTEGRATED ADVICE

In Section 15, ACE presents a discussion document with ideas that ICES needs to consider internally and discuss with partners, and then take actions to more forward in relation to the provision of integrated advice. This document follows on from a review in 2002 of initiatives to provide integrated management advice at the national, regional, and international level. In 2003, further relevant activities were reviewed and discussed, including the EC Stakeholders Conference “Towards a Strategy to Protect the Marine Environment” held in Køge, Denmark, and meetings of ICES regional ecosystem study groups for the North Sea and the Baltic Sea. A number of features of an integrated ecosystem approach that are common across these initiatives are described and discussed.

1 INTRODUCTION

The Advisory Committee on Ecosystems (ACE) was created in 2000 as the Council's official body for the provision of scientific information and advice on the status and outlook for marine ecosystems, and on exploitation of living marine resources in an ecosystem context. ACE provides a focus for advice that integrates consideration of the marine environment and fisheries in an ecosystem context, such as ecosystem effects of fishing. ACE will be at the forefront of the development of advice on ecosystem management.

ACE provides advice as may be requested by ICES Member Countries, other bodies within ICES, relevant regulatory Commissions, and other organizations.

In handling the requests, ACE draws on the expertise of its own members and on the work of various expert ICES Working Groups and Study Groups. ACE considers the reports of these groups and may request them to carry out specific activities or to provide information on specific topics.

Request

Item 2 of the 2003 requests from the Helsinki Commission, which states as follows:

To evaluate every third year the populations of seals and harbour porpoise in the Baltic marine area, including the size of the populations, distribution, migration, reproductive capacity, effects of contaminants and health status, and additional mortality owing to interactions with commercial fisheries (by-catch, intentional killing).

The evaluation is based on annual submission of data to ICES from ICES member states.

Source of information

The 2003 Report of the Working Group on Marine Mammal Ecology (WGMME) (ICES CM 2003/ACE:03).

Summary

Table 2.1. Summary of the status of seals and harbour porpoise in the Baltic Sea.

	Baltic ringed seal	**Saimaa seal**	**Ladoga seal**	**Grey seal**	**Kalmarsund harbour seal**	**SW Baltic harbour seal**	**Harbour porpoise**
Distribution	Resident in three separate regions	Fragmented, 60% of lake area	90% of lake area	Northern and central Baltic Proper	Kalmarsund, resident	To west of 13 °E	Southern Baltic Proper, Belt Seas, Kattegat
Population size in year 1900	200,000	<1,300	10,000	100,000	5,000	10,000	Unknown but larger than at present
Current population estimate	9,000[1]	240–250	5,000	13,000	580[2]	4,500	36,046[3] 588[4] 599[5] 817[6]
Population trend	+5% GoB; unknown in GoR, GoF	+3–4%	Unknown	+ 7.8% SE, other areas unknown	+9%	−53% (epizootic loss)	Unknown
Current reproductive rate[7]	0.65	0.80	Unknown	>0.60	>0.85	Unknown	Unknown
Health status	Sterility, renal lesions	Normal	Skin lesions	Colonic ulcers; renal, bone lesions	Unknown	Bone lesions, skin lesions	Many lesions and parasites
By-catch (per year)	120 FIN, SE	10	Several tens	430 SE, 150 EST, c10 POL, other countries unknown	Unknown	300 SE, other countries unknown	10s Baltic Proper, 100s in Belt Sea/ Kattegat
Intentional killing	0	0	Tens poached	Quota 586, less than 50% taken. 35 pups poached in EST (2002)	0	30 DK, 6 SE	0

Key: GoB = Gulf of Bothnia; GoF = Gulf of Finland; GoR = Gulf of Riga; DK = Denmark; EST = Estonia; FIN = Finland; POL = Poland; SE = Sweden.
[1]Estimated from basking population of 5,400 individuals in 1996; [2]Estimated from basking population of 2,000 in 1994; [3]ICES Divisions IIIa,b; [4]ICES Division IIIc; [5]Baltic Sea Sub-divisions 24 and 25; [6]Kiel and Mecklenburg Bight; [7]Numbers of pups produced per breeding female per year.

Baltic ringed seal (*Phoca hispida*)

Baltic ringed seals are found in three main areas: the Bothnian Bay, Gulf of Finland, and Gulf of Riga. The population has undergone a dramatic decline in population size since the beginning of the 20th century. In the first decade of the 20th century, about 190,000 to 220,000 ringed seals were present in the Baltic. By 1939 their number had declined to 23,000 to 27,000 seals. In the mid-1970s, population size was considerably below 5,000 animals. Organochlorines have affected reproductive success in ringed seals. Quantitative data on pesticide loads were lacking. Neurotoxins may have been involved in a high mortality event of seals in the Gulf of Finland in 1991.

Saimaa ringed seal (*Phoca hispida saimensis*)

Ringed seals in Lake Saimaa took up 90–95% of the area of the lake at the beginning of the 20th century. The range was only 30–40% at the beginning of the 1990s. Population size was estimated at 2,000–2,500 seals in Lake Saimaa and 3,800–4,900 including Lake Kolovesi at the beginning of the 20th century. Population size had decreased to about 100 seals at the beginning of the 1980s due to habitat degradation, introduction of more harmful fishing methods, and environmental toxins swept into the lake. Since then, the population has been slowly increasing. It is currently estimated at about 240–250 seals. The burden with organochlorines is low and did not seem to have affected reproductive performance. Drowning in fishing nets and lair mortality were the major causes of incidental killing.

Ladoga ringed seal (*Phoca hispida ladogensis*)

The population declined by 50–75% in the late 19th century due to hunting, to about 10,000 individuals, and was estimated at about 5,000 individuals in the mid-1990s. Skin lesions were observed in August 2001 and the season thereafter in part of the lake. There is no evidence for increased mortality due to the lesions.

Harbour seal (*Phoca vitulina*)

Two populations have been described as living in the Baltic: the Kalmarsund population (or eastern Baltic population) and the southwestern Baltic population, extending from the Kattegat into the western Baltic Sea. The Kalmarsund population, which is substantially smaller than the southwestern population, has not been affected by the 2002 Phocine Distemper Virus (PDV) epizootic, while the Kattegat population has been.

Kalmarsund population

Seals numbered up to 5,000 animals historically and until about 1905, when numbers started to decline substantially. The population was at its lowest level between 1960 and 1985, when hardly more than 100–200 seals were present. Population size has increased again, to 580 seals in 2002. The population was not affected by the PDV epizootic in 1988 nor by the epizootic in 2002. Organochlorines have probably affected the reproductive capacity of the population between 1977 and 1989. By-catch in eel pound nets was high in the past, but has been reduced to low levels in recent years.

Southwestern population

50% of the seals of the western population died in the course of the 1988 PDV epizootic. Population size had increased to 10,000 seals again when a second PDV epizootic hit the population in 2002 and reduced the population size again by approximately 50%. The part of the population in the southwestern Baltic had increased from 224 in 1990 to 315 in 1998. The PDV epizootic in 2002 reduced this part of the population by 50%. The high prevalence of alveolar exostosis (not found in reference material from 1850–1930) was thought to be indicative of organochlorine pollution. By-catches amounted to 300 seals in the Swedish fishery along the west coast in 2001. No information was available from the Danish or German gillnet fisheries in the area.

Grey seal (*Halichoerus grypus*)

Grey seals are mostly found in the central Baltic between 57° N and the ice edge. They do not breed in the southern part of the central Baltic or in the western Baltic. At the beginning of the 20th century, the minimum population size was estimated to be 100,000 individuals. Population size for the year 2000 has been estimated as just over 12,000 seals using a capture-recapture method based on photographic identification studies. The mean annual population increase along the Swedish coast was 7.8% from 1990 to 2002. Incidental catch and hunting were the main causes of death. 150 seals were reported to have been taken in commercial fisheries in Estonia in 2001. 430 grey seals were taken in Swedish fisheries, with half of this occurring in the gillnet fisheries for turbot and cod. The relative proportion of by-caught animals, however, has declined from 14% to less than 10%. To mitigate the fisheries impact on seals, Finland has established seven nature areas mainly for grey seals, and to some extent, also for ringed seals. In addition to incidental takes, licenses to kill 586 seals were issued, but only about half of these were taken in the seal hunt. Colonic ulcers were found to be the second most important cause of death after incidental catches and hunting.

Harbour porpoise (*Phocoena phocoena*)

Current knowledge indicates that two populations of harbour porpoise live in the Baltic: one in the Baltic Proper east of the Darss sill and one in the western Baltic Sea, Mecklenburg Bight, which is part of a larger Kattegat/Belt Sea population.

Baltic Proper

The population size historically is not precisely known, but was much larger at the beginning of the 20th century. The population size decreased dramatically thereafter due to four main reasons:

- Ongoing hunt in the first decades of the 20th century;
- At least two severe ice winters in the 1920s and 1930s, which caused substantial losses;
- Habitat degradation due to organochlorine pollution from the late 1950s onwards; and
- By-catch in fisheries.

The initial range of the harbour porpoise included most of the Baltic Sea and has now decreased to southwestwards of a line from Gotland to the Lithuanian border, where the current population size was most recently estimated to be 599 (C.I. 200–3300). East of this line, harbour porpoises were found only irregularly. A series of systematic aerial surveys was undertaken on German and some Danish waters in the Baltic from May 2002 to March 2003. An unusually large concentration of porpoises was found on the eastern part of Oderbank close to the Polish border in the mating season May-June 2002 but not during any other flight.

Organochlorine levels in harbour porpoises from the Baltic Proper have been found to be high. A large number of lesions and pathological changes including heavy parasite loads have been reported. By-catches are known to occur in most areas where porpoises are found. Forty-five porpoises were reported as being incidentally taken in Polish fisheries within a ten-year period since the late 1980s, with half taken in Puck Bay. Three to five porpoises were estimated as being taken in German fisheries annually including the western Baltic. Limited data were available from the Danish gillnet fisheries.

Given the poor state of the harbour porpoise in the Baltic Proper, ASCOBANS (Agreement on Small Cetaceans in the Baltic and the North Seas) developed a recovery plan in January 2002 ("Jastarnia Plan") in order to support the recovery of the harbour porpoise. Main aspects of the plan are the replacement of salmon driftnets by longlines and gillnets by large traps and similar gear less harmful to porpoises. Pingers should be introduced into Baltic fisheries as soon as possible. This plan, which was scheduled for discussion and agreement at the Fourth Meeting of the Parties of ASCOBANS in August 2003, is seeking the support of other agreements working on conservation and rational use of the Baltic, such as IBSFC (International Baltic Sea Fisheries Commission), HELCOM (Helsinki Commission), and the European Commission.

Kiel and Mecklenburg Bight

Harbour porpoises in the Mecklenburg/Kiel Bight area are part of the larger Skagerrak/Kattegat/Belt Sea population, which was estimated at about 36,000 porpoises in 1994 (SCANS I). The size of the Mecklenburg/Kiel Bight part of the population was estimated to be 817 porpoises (C.I. 300–2,400) in 1995. These figures are likely to be revised in the near future when results from aerial surveys in German waters of the Baltic in 2002 and 2003 will be analysed for porpoise distribution and abundance. A second international survey on harbour porpoises as part of an overall survey of small cetaceans (SCANS II) is planned in the southwestern Baltic in 2005.

Organochlorine and parasite loads and diseases of porpoises are similar to those in the Baltic Proper.

Recommendations and advice

1) The by-catch of ringed seals has been recorded in some fisheries. In order to extrapolate these records to be able to estimate the total by-catch in the Baltic, ICES recommends that a study be initiated to compile the number of fishing vessels and to estimate fishing effort in the salmon and whitefish trap fisheries in the Baltic.
2) The by-catch of Lagoda ringed seals has not been studied. ICES recommends that such a study be undertaken, using an independent observer scheme. This will also require a compilation of the number of fishing vessels and an estimate of fishing effort in Lake Ladoga.
3) The by-catch of harbour seals in Danish and German fisheries in the Baltic has not been studied. ICES recommends that such a study be undertaken, using an independent observer scheme. This will require a greater knowledge of the effort in relevant fisheries, particularly those for lumpsucker and flatfish in bottom-set gillnets.
4) The population of grey seals in the Baltic is about 15% of pristine levels. The prevalence of colonic ulcers has increased and is the second most important cause of death, after by-catch. ICES notes that renal lesions and bone lesions persist in the population, such that the Baltic grey seal population cannot be regarded as having a normal health status. The estimated mortality that arises from the by-catch and the quotas given for intentional killing approaches 10% of the population size (and no upper limit has been set for the hunting quota in Åland). ICES therefore notes that the present population of grey seals in the Baltic is not in a favourable conservation status and advises that the extra mortality caused by humans should be reduced.

To this end, ICES advises:

- that the by-catch of grey seals be investigated and where possible reduced;
- that upper limits should be set for the intentional killing of grey seals in the Baltic; these upper limits should be set such that the overall population will not be further depleted and should allow the population to be able to increase.

ICES is not advising at present on population limits for the Baltic grey seal, but these should be consistent with international standards (e.g., favourable conservation status). Some information exists that could serve as a basis to estimate such limits. ICES is able to evaluate this information and, if the information permits, to advise on potential management strategies and possible values for the limits.

5) The number of harbour porpoises in the Baltic Proper is a small fraction of their former abundance. Their status has been reviewed extensively, and although the causes of their decline are unclear, by-catch in fisheries will be inhibiting any possible recovery. Ways of reducing this by-catch have been reviewed, and working with stakeholders (including fishermen), a recovery plan (ASCOBANS, 2002) has been drawn up. Key recommendations of this plan include:

- Fishermen and their representatives need to be closely involved in any implementation process for by-catch reduction.
- Measures should be taken by the Baltic Range States to reduce the fishing effort of driftnet and bottom-set gillnet fisheries in the Baltic.
- Fishing methods should be changed away from gear known to be associated with high porpoise by-catch (i.e., driftnets and bottom-set gillnets) and towards alternative gear that is considered less harmful.
- Trials of fish traps, fish pots, and longlines should be initiated immediately, with the long-term goal of replacing gillnets in the cod fishery, particularly in areas where porpoises are known or expected to occur frequently.
- Serious consideration should be given to replacing driftnets in areas where porpoise by-catch is known or likely to occur.
- A study is needed to compile data on the spatial and temporal distribution of fishing effort in the Baltic.
- Pinger use should be made mandatory in specific high-risk areas and fisheries, on a short-term basis (2–3 years).

ICES supports all of these recommendations, and other parts of the recovery plan.

Scientific background

2.1 Baltic ringed seal (*Phoca hispida botnica*)

2.1.1 Distribution and migration

Baltic ringed seals are found in three main areas in modern times (Reeves, 1998): the Bothnian Bay (Figure 2.1.1), the Gulf of Finland (Figure 2.1.2), and the Gulf of Riga (Figure 2.1.3). About 70% of the total population is found in the north, 25% in the Gulf of Riga, and 5 % in the Gulf of Finland. The winter distribution of the species is largely determined by the occurrence of dense pack ice and fast ice. The main breeding areas are found in the central northern part of the Bothnian Bay (Härkönen and Lunneryd, 1992), the eastern part of the Gulf of Finland, and in the Gulf of Riga (Härkönen *et al.*, 1998). Outside these areas, small numbers of ringed seals are found in the Bothnian Sea (Härkönen and Heide-Jørgensen, 1990; Härkönen *et al.*, 2003) and in the southwestern archipelago of Finland (Helle and Stenman, 1990).

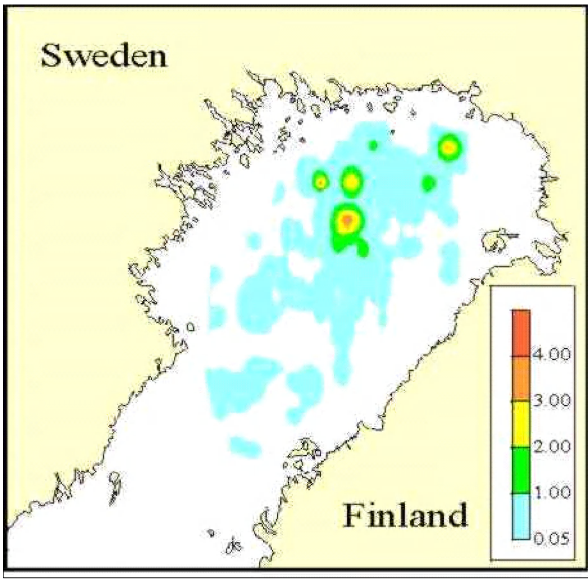


Figure 2.1.1. Late winter distribution of ringed seals in the Bothnian Bay (no. of seals km^{-2}) (Härkönen *et al.*, 1998).

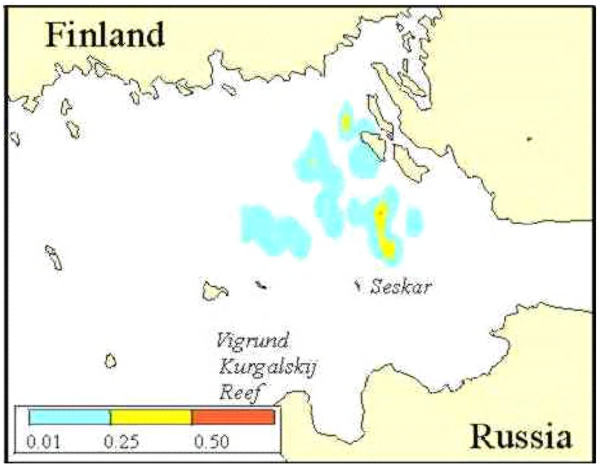


Figure 2.1.2. Distribution of ringed seals breeding in the Gulf of Finland (no. of seals km^{-2}) (Härkönen *et al.*, 1998).

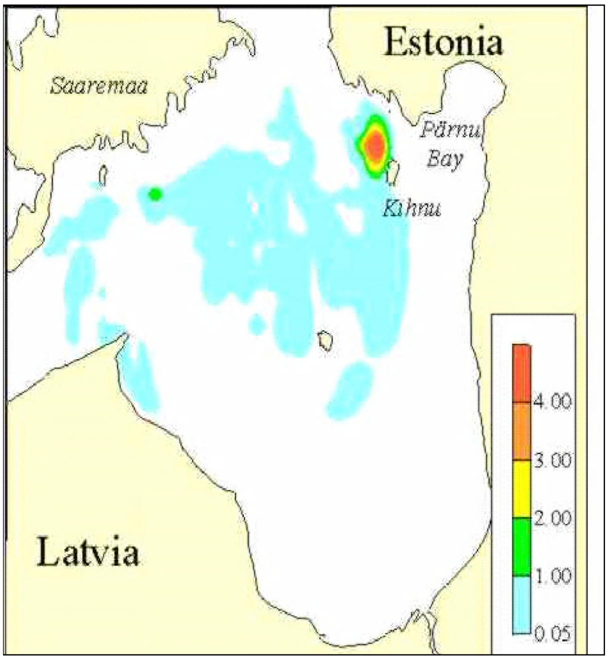


Figure 2.1.3. Distribution of ringed seals breeding in the Gulf of Riga (no. of seals km^{-2}) (Härkönen *et al.*, 1998).

2.1.1.1 Distribution during the ice-free period

Limited information is available on distribution during the period spring to autumn, since ringed seals in the Bothnian Bay do not haul out in large numbers during the ice-free period and there have been no at-sea surveys. In contrast, the species does haul out in the Russian part of the Gulf of Finland. Several haul-out sites were found on islands, islets, and rocks during boat surveys in spring and summer 1993–1996. The largest numbers of ringed seals, in groups of up to 80 individuals, were observed at the Kurgalskij Reef and Vigrund Island along the southeastern coast of the Gulf during late May, early June, and in August.

Several groups, comprising up to a few tens of animals, haul out on rocks around the islands Hiiumaa and Saaremaa on the Estonian west coast. There is evidence of seasonal changes in the haul-out pattern of these seals, as the highest numbers are observed in early spring and late autumn (Härkönen *et al.*, 1998).

2.1.1.2 Late winter distribution

In the Bothnian Bay and in the Gulf of Finland, the largest numbers of ringed seals are hauled out on ice during the moulting period in April to May (e.g., Helle, 1980a). The largest numbers on ice in the Gulf of Riga occur by mid-April, as ice conditions there deteriorate two weeks earlier than in the Gulf of Finland.

Information on the late winter distribution of Baltic ringed seals is based on results from the 1996 survey, when all three areas were surveyed using the same method (Härkönen *et al.*, 1998). The distribution in the Bothnian Bay was similar in all years from 1988–1996, with the largest concentrations of seals found in the central northern drift ice. This pattern was similar when ice fields extended south of the Quark archipelago in 1988, indicating that seal distribution in this area is not directly correlated to the size of the ice fields.

In the Gulf of Finland, the highest densities were found in the east. This pattern of distribution was also found in 1993–1995, most probably a consequence of the distribution of suitable seal ice. A survey in Estonian parts of the Gulf on 18 April 1996 found low numbers of ringed seals in the area.

The whole of the Gulf of Riga was covered by ice in 1996, but ringed seal distribution was strongly concentrated in the northeastern section. The highest concentrations were found around Kihnu Island (a traditional seal hunting area) close to Pärnu Bay and along cracks at the fast ice edge in the northeast.

2.1.1.3 Movements

Ringed seals in the Baltic are relatively sedentary.

In late May, seals in Estonian coastal waters equipped with satellite transmitters left the shallow areas in the Moonsund Archipelago, and most of their “at-sea locations” during June and July were over deeper waters in the Gulf of Riga, but also at the mouth of the Gulf of Finland. In September and October, all seals moved towards the main coastline. In December and January, all seals moved to the northern parts of the Gulf of Riga, a behaviour correlated to ice formation. In February, the Moonsund is covered by fast ice and none of the seals remained in the area. The transmitter-tagged seals in the Gulf of Finland showed similar patterns.

Estonian seals spend the winter in the northern parts of the Gulf of Riga and move northwards to the Moonsund Archipelago at the break-up of the fast ice in April (Härkönen *et al.*, 1998). At the same time, catches of herring in the Moonsund increase and peak in early May. In June and July, herring abandon the shallow areas and move to the deeper waters of the Gulf of Riga and off the Estonian west coast, a pattern also followed by the seals.

2.1.2 Population size

2.1.2.1 Historical population size

Modelling based on numbers of killed ringed seals shows that 190,000 to 220,000 ringed seals occurred in the Baltic up to the first decade of the 20th century (Harding and Härkönen, 1999).

The traditional methods used for hunting seals before the 20th century (stalking, clubbing, netting, and harpoon (Bergman, 1956)) were less efficient for catching solitary ringed seals compared with group-forming grey seals. With the introduction of bounties in 1903 (Sweden) and 1909 (Finland) and the use of modern rifles, the situation pressure on seals increased considerably. The intensive hunting during the 1910s reduced the population to about half of the original size. The decline in the population continued in the 1920s, but at a considerably lower rate. As a consequence of favourable weather conditions, but also for economic reasons, hunting pressure increased again in the 1930s, which resulted in a new dramatic drop in ringed seal numbers up to 1939, when about 23,000 to 27,000 seals remained (Figure 2.1.2.1.1).

During the next 25 years (1940–1965), the population appears to have been stable, but a new decline occurred in the mid-1960s as a result of increased hunting pressure in Finland and Estonia (Figure 2.1.2.1.1). Although catches decreased after 1969, the population continued to decline until 1975 as a consequence of lowered net reproductive rates after 1965. In the mid-1970s, the population was considerably below 5,000 animals.

2.1.2.2 Current population in the Bothnian Bay

The first surveys, conducted in 1975 and 1978, provided estimates of about 3,000 ringed seals in the area for both years (Helle, 1980a). Surveys in 1984 and 1987 indicated a decreasing trend in the local population up to the mid-1980s (Helle, 1990). From 1988 to 2002, the population increased at about 5% per year over this period (Figure 2.1.2.2.1) to an estimated hauled-out population in 2002 of 4,498.

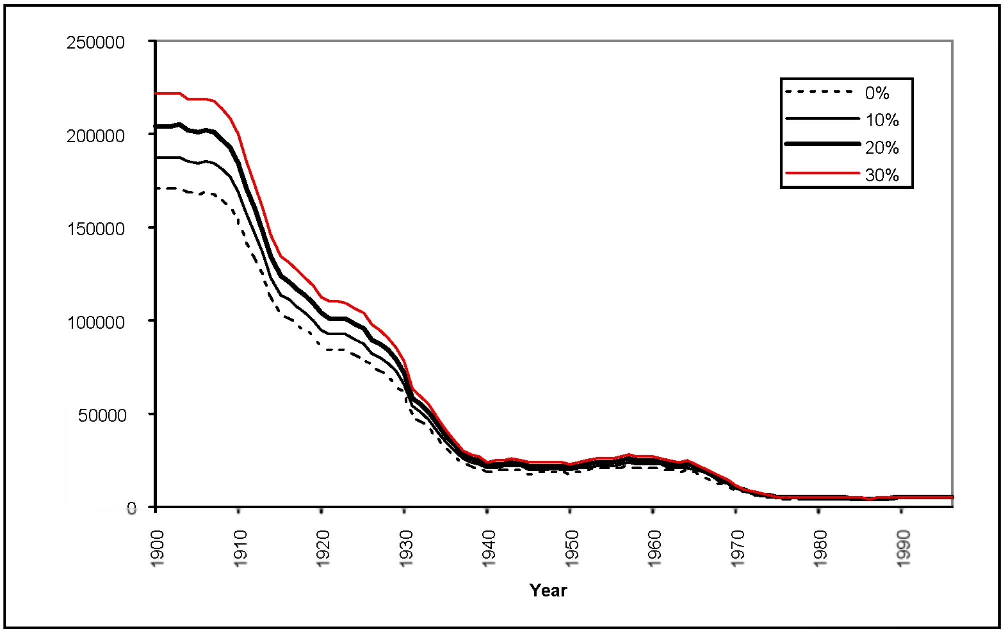


Figure 2.1.2.1.1. Projection of past population sizes of ringed seals in the Baltic (Harding and Härkönen, 1999). Ranges given for 0 to 30% hunting losses.

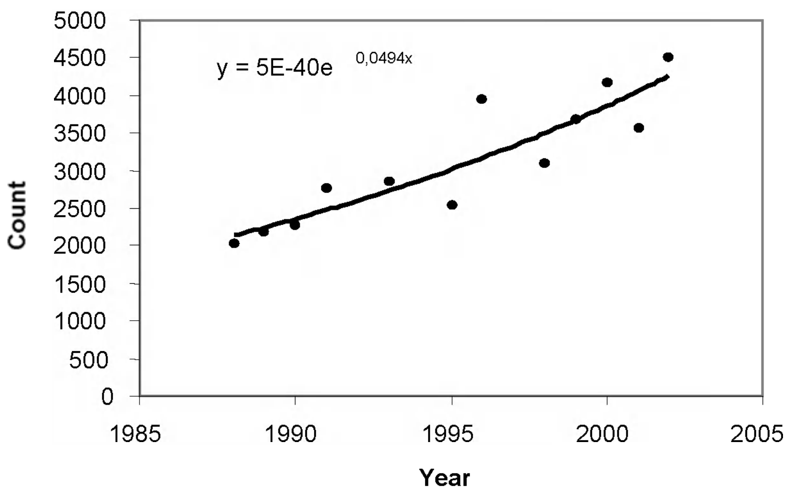


Figure 2.1.2.2.1. Counts of hauled-out ringed seals in the Bothnian Bay, 1988–2002 (Härkönen *et al.*, 1998, 2003).

2.1.2.3 Current population in the Gulf of Finland

The first estimate of ringed seal population size for the Soviet area of the Gulf of Finland was of 5,000 individuals in 1970 (Rezvov, 1975). A second estimate of 8,200 ringed seals in Soviet waters in the Gulf of Finland was made in 1973 (Tormosov and Rezvov, 1978). Both estimates are difficult to evaluate as no information is available on survey methods. A further estimate was made of 2,100 ringed seals for the Gulf of Finland based on surveys in 1979 (Tormosov *et al.*, unpublished). However, a revision of this estimate indicated that it should have been 793 hauled-out ringed seals for the Gulf of Finland (Härkönen *et al.*, 1998).

With the exception of the Härkönen *et al.* (1998, 2003) review of the study of Tormosov *et al.* (unpublished), there is no useful information on earlier population size in the Gulf of Finland, and the published assessments are probably gross overestimates. Early estimates cannot therefore be used for evaluations of past trends.

Aerial surveys using strip transect methods undertaken from 1992 to 1996 gave estimates of between 92 and 169 seals (Table 2.1.2.3.1). Sampling fractions for the surveys of 1992 and 1993 are not available, and thus the estimates are unreliable. A variant of the strip census method was used in 1994 and 1995 (Härkönen *et al.*, 1998, 2003), and the population assessments can be used only as indications of population size. In 1996 the total ice-covered area was 2,688 km^2, but only 60% (1,613 km^2) could be covered by the range of strips due to military regulations. Thus, if the mean seal density in strips west of 28°20'E at 0.053 seals km^{-2} is applied to the remaining ice-covered area (1,075 km^2), a calculated number of 57 seals results for that area. Therefore, the estimate for the whole ice-covered area would be 149 ringed seals.

2.1.2.4 Current population in the Gulf of Riga and the Estonian west coast

Notes in the literature (Greve, 1909) and statistics of seal hunting (Anon., 1939) indicate that in earlier times ringed seals were numerous in the area. However, earlier data cannot be relied upon as a basis for estimates of abundance or distribution. Surveys were made also in April 1994 and 1996 (Table 2.1.2.4.1).

2.1.3 Reproductive capacity

Population surveys in the Gulf of Bothnia show a 5% annual increase in population, which is roughly 50% of the intrinsic rate of increase of ringed seals breeding in the Arctic. Population models suggest that pregnancy rates should be about 0.65 to yield the measured rate of population increase. The pregnancy rate of Arctic ringed seals is 0.90 (e.g., Smith, 1987). Although limited information is available from the Gulf of Finland, it is suggested that impaired reproduction also occurs in that area (Westerling and Stenman, 1992).

Table 2.1.2.3.1. Survey results of hauled-out ringed seals in the Gulf of Finland in 1992–1996 (Härkönen *et al.*, 1998, 2003).

Year	Date	Ice area (km^2)	Seal density (seals km^{-2})	SD	Count	Sampling Fraction (%)	Population estimate, Russia	±CI 95%
1992		-			89			
1993		-			40	<30	150	
1994	30/4	-			61	36	169	
1995	15/4	-			54	32	169	
1996	5/5	1,613	0.057	0.135	22	24	92	41

Table 2.1.2.4.1. Survey results of hauled-out ringed seals in the Gulf of Riga on 14–21 April 1994 and on 15–17 April 1996 (Härkönen *et al.*, 1998). The estimate for 1994 is only approximate.

Year	Ice area (km^2)	Seal density (seals km^{-2})	SD	Count	Sampling fraction (%)	Population estimate	±CI 95%
1994	1,000–4,000	-	-	450	unknown	(680)	
1996	9,945	0.142	0.526	228	16.2	1,407	590

2.1.4 Effect of contaminants and health status

In the autumn of 1991, a high mortality was observed among ringed seals in the Russian part of the Gulf of Finland. About 150 dead seals drifted ashore both in Russia and in Finland, and there was speculation that natural or man-made neurotoxins were involved (Westerling and Stenman, 1992). Between two and seven corpses of adult ringed seals were found annually between 1992 and 1996 (Westerling and Stenman, 1992).

Pathological changes in reproductive tracts observed in the 1980s (Helle, 1980b) still persist in the ringed seal population but at lower frequencies (Mattson and Helle, 1995; E. Helle, pers. comm.). Physiological studies show that Baltic ringed seals are affected by persistent organic pollutants (Nyman *et al.*, 2001).

2.1.5 Interactions with commercial fisheries and intentional killing

By-catches of ringed seals in the Swedish and Finnish fishery were estimated to amount to about 50 and 70 individuals, respectively, in 2001 (Lunneryd *et al.*, 2003; E. Helle, pers. comm.). The majority of these are drowned in salmon and whitefish traps. By-catch information is not available from other areas, but is likely to occur. There is no intentional killing apart from the scientific sampling in some recent years (up to ten mature females per year to study reproductive state) in the Gulf of Bothnia. A small number of ringed seals are shot by mistake under licences issued for the grey seal hunt in Finland and Sweden.

2.2 Saimaa ringed seal (*Phoca hispida saimensis*)

2.2.1 Distribution, population size and trends

Historical estimates showed that the range of the Saimaa ringed seal covered about 90–95% of the surface area of Lake Saimaa at the start of the 20th century. Towards the end of the 20th century, the range was about 30–40% (Sipilä, 1994). There was no notable reduction in the area of distribution during the 1980s and 1990s.

Modelling suggests that the number of Saimaa ringed seals was between 100 and 1,300 seals in the 1890s (Kokko *et al.*, 1999). The highest density of 0.88–1.12 km^{-2} is at present in Lake Kolovesi. The population size of Lake Saimaa in pristine state was assumed to be at least 2,000–2,500 seals (Hyvärinen and Sipilä, 1992). Extrapolating, based on Lake Kolovesi, gives a potential total population size to Lake Saimaa of about 3,800–4,900 seals. The size of the Lake Saimaa seal stock decreased substantially in the mid-1950s (Kokko *et al.*, 1999). In the late 1960s, the population decreased further, probably as a result of changes in the habitat, new fishing methods, and environmental toxins such as mercury (Marttinen, 1946; Sipilä, 1981, 1990; Becker, 1984; Sipilä *et al.*, 1990). Counts in the 1980s were most likely underestimates, due to insufficient coverage (Helle *et al.*, 1981; Sipilä, 1983; Sipilä *et al.*, 1990). The first estimate based on systematically collected data was made in 1990 (Table 2.2.1.1, Figure 2.2.1.1). The highest density of seals, and about 50% of the population, is now found in the central parts of Lake Saimaa (Lake Haukivesi and Lake Pihlajavesi).

In the late 1990s, the seal stock increased by about 50 animals (Table 2.2.1.1, Figure 2.2.1.1), corresponding to an annual growth rate of about 4%. The improved accuracy in the late 1990s in the lair counting method can explain 25% of the observed rate of increase in the late 1990s. The present annual growth rate is substantially lower than the possible maximum rate of increase for ringed seal populations at 10% (Reeves, 1998), but substantially higher than the 1% estimated for the period 1977–1995.

2.2.2 Reproductive capacity

The pregnancy rate in the Saimaa ringed seal population was about 70% from the early 1980s to 1991. For later periods, it varied between 75% and 83%. The mean pregnancy rate also varied in different areas of Lake Saimaa during the study period. Numbers of pups in relation to total numbers of seals were about 15% in the early 1980s, 17% in 1990, 21% in 1995, and 24% in 2000 (Figure 2.2.2.1).

2.2.3 Effect of contaminants (mercury and organochlorines) and health status

Mercury loads in the environment severely influence many life history features. Very high concentrations of mercury in Saimaa ringed seal tissues were found in the 1960s (Helminen *et al.*, 1968; Henriksson *et al.*, 1969). A substantial reduction has been found in mercury levels in adult seal tissue from the early 1980s to the first half of the 1990s (Hyvärinen *et al.*, 1998). By contrast, mercury concentrations in liver and muscle tissues of seals less than one month of age, as well as in lanugo hair of pups, did not show notable changes over the period 1981–1995 (Hyvärinen *et al.*, 1998).

There is a clear positive correlation between seal age and mercury concentration, and the mean accumulation rate of mercury in liver was 11 mg yr^{-1}. About 80% of the total mercury burden is found in the liver in adult seals, whereas the rest is found in muscle tissue.

In the Bothnian Bay, high concentrations of organochlorines are correlated with the observed decrease in birth rate of ringed seals (Helle, 1980b, 1985; Helle *et al.*, 1976). Concentrations of PCBs and DDT are relatively low in Lake Saimaa and the burdens of organochlorines have not been shown to affect the reproduction of Saimaa ringed seals (Helle, 1985; Helle *et al.*, 1983, 1985; Kostamo *et al.*, 2000).

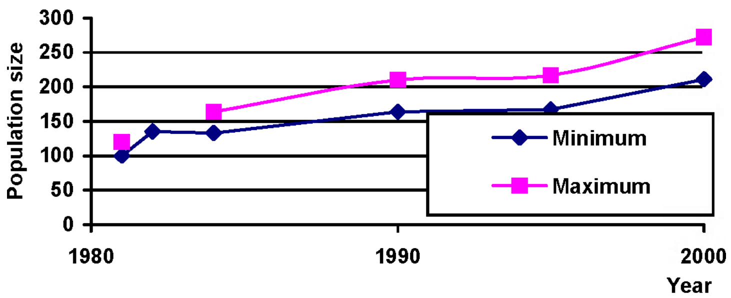


Figure 2.2.1.1. Maximum and minimum estimates for the Saimaa ringed seal population size from 1982–2000 (Sipilä, 1992, updated information).

Table 2.2.1.1. Estimated numbers of Saimaa ringed seals in the early winters 1984, 1990, 1995, and 2000 in different parts of the lake. These figures do not include pups born in the year of count (Sipilä, 1992, updated information).

Area	**1984**	**1990**	**1995**	**2000**
Pyhäselkä	13	13	8–10	3–5
Orivesi	9–10	13–15	11–15	10–14
Pyy- and Enonvesi	6–7	5–8	6–8	15–18
Joutenvesi	8–10	13–18	13–18	20–30
Kolovesi	5–6	13–16	13–16	22–28
Haukivesi	32–37	41–55	44–54	48–58
Pihlajavesi	14–19	35–40	40–46	55–65
Tolvanselkä- Katosselkä	4–5	13–19	15–25	16–24
Lietvesi	8–9	13–16	8–12	7–10
Luonteri	1–2	2	2	2
Petranselkä	9–12	3–4	4–7	11–15
Ilkonselkä	4	4	3–4	2–3
Lake Saimaa	113–134	164–210	167–217	211–272

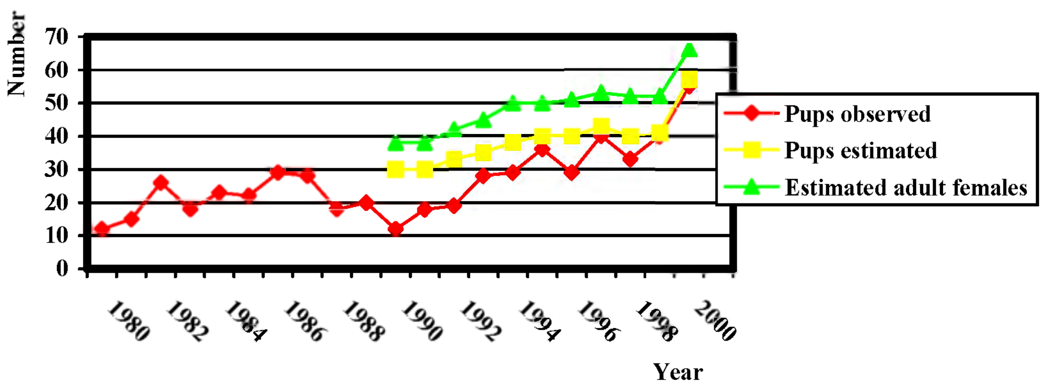


Figure 2.2.2.1. Numbers of Saimaa ringed seal pups observed and estimated, and mature females, 1980–2000 (Sipilä, 2003a).

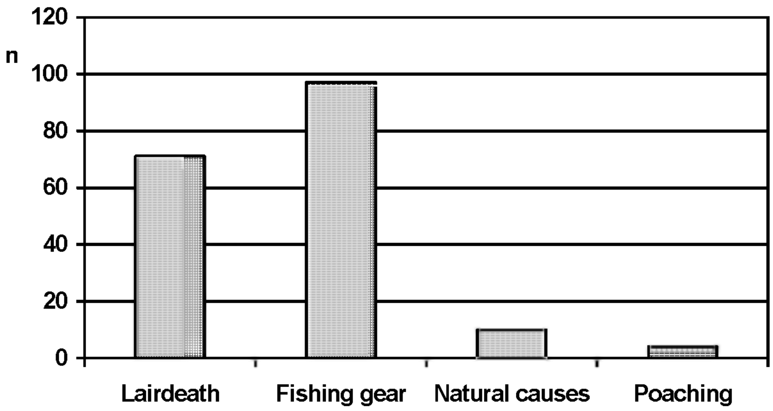


Figure 2.2.4.1. Main causes of death of the Saimaa ringed seal, 1977–2000. "Lair-death" includes premature birth, still-birth, and accidental death in the lair; "fishing gear" includes also deaths from suffocation without direct evidence of contact with fishing gear. "Natural causes" do not include lanugo-coated pups found dead (Sipilä *et al.*, 1999, 2002a; Sipilä and Koskela, 2003).

2.2.4 Interactions with commercial fisheries and intentional killing

A total of 182 carcasses were collected between 1977 and 2000. The most common causes of death of ringed seals in Lake Saimaa were drowning (or suffocation) in fishing gear (53.3%) and lair mortality (39.0%) (Figure 2.2.4.1).

The survival rate of weaned pups up to the age of two years is about 10% higher in the fishing restriction areas than in areas without restrictions. There is no record of intentional killing since the mid-1980s.

2.3 Ladoga ringed seal (*Phoca hispida ladogensis*)

2.3.1 Distribution and historical and current population size

The species is distributed throughout Lake Ladoga with the exception of the southernmost part. The northern part of the lake does not freeze during mild winters, and approximately 80% of the population can be found in the southern parts of the Ladoga under such conditions (Antoniuk, 1975; Filatov, 1990). Aerial surveys in 1994 revealed that the seals prefer closed and compact ice to open ice.

Numbers of Ladoga ringed seals decreased by about 50–75% during the 19th century as a consequence of hunting (Chapskii, 1932; Jääskeläinen, 1942; Sipilä *et al.*, 1996, 2002b; Sipilä and Hyvärinen, 1998). The current seal population in Lake Ladoga is estimated at about 5,000 individuals (Medvedev *et al.*, 1996); the population is thought to have remained stable over the past 30 years.

2.3.2 Reproductive capacity, effect of contaminants and health status, and interactions with commercial fisheries and intentional kill

The reproductive capacity of the species is not known, nor is there information on contaminants in these seals. In August 2001, skin lesions resembling seal pox were observed in about 40% of the seals in the Valaam Archipelago. There is no evidence of increased mortality or changes in behaviour of the affected seals. Current by-catches are unknown, but it is known that some seals are killed illegally.

2.4 Harbour seal (*Phoca vitulina*) (Kalmarsund stock)

There are two genetically separate harbour seal stocks in the Baltic Sea area (Figure 2.4.1). One is now confined to the Kalmarsund area in Sweden (and was described by Stanley *et al.* (1996) as the East Baltic stock), while another stock extends into the southwestern Baltic from the Kattegat (known as the West Baltic stock by Stanley *et al.*, 1996). The Kalmarsund stock numbers substantially less than the southwestern Baltic and Kattegat stock. Despite not being affected by the 2002 PDV epizootic, the Kalmarsund stock remains of greater management concern than the southwestern Baltic and Kattegat stock.

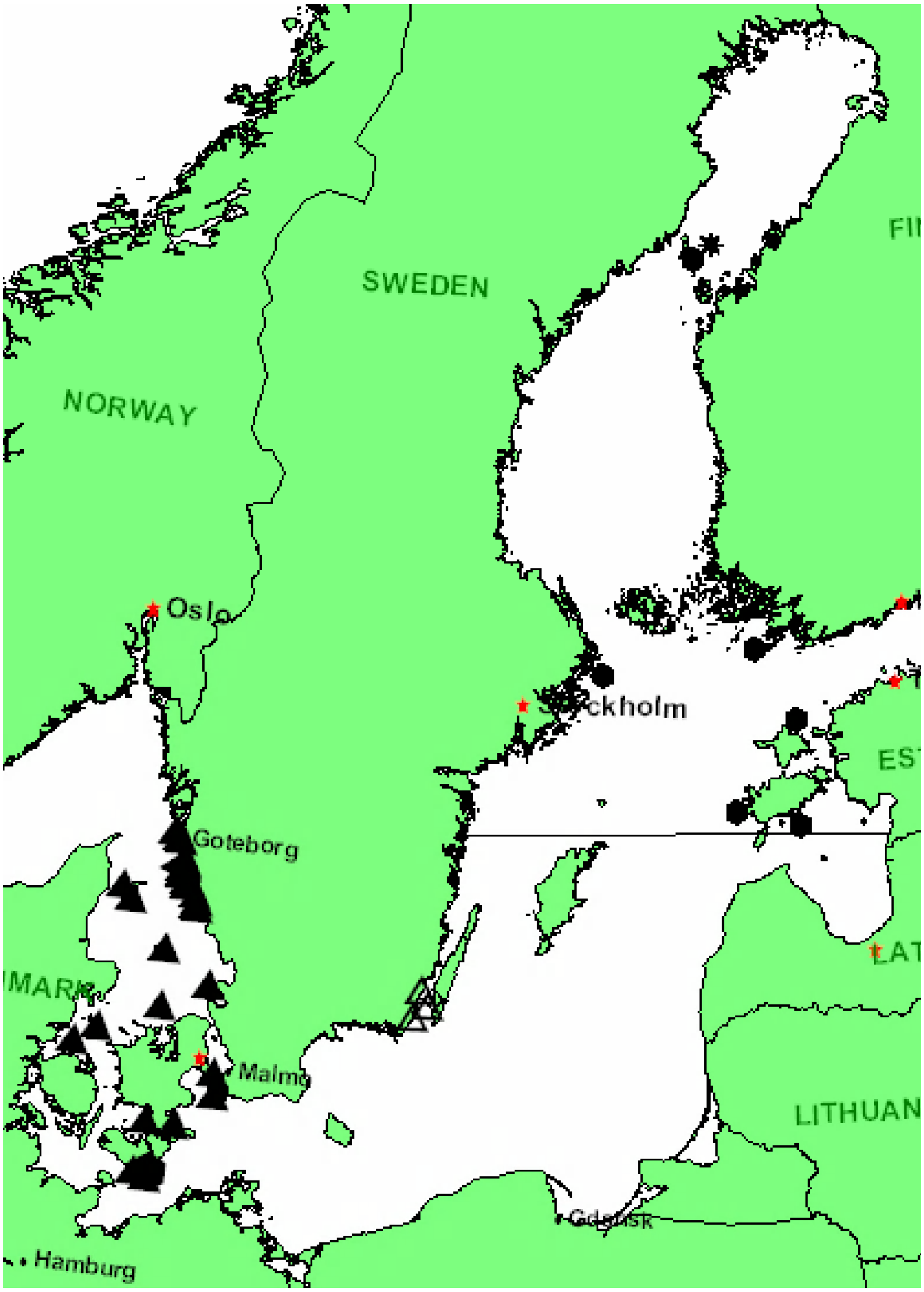


Figure 2.4.1. Breeding sites in the Baltic and Kattegat. Grey seals: filled circles (asterisks indicate breeding on ice). Harbour seals (Kalmarsund population): open triangles. Harbour seals (Southwest Baltic and Kattegat): filled triangles.

2.4.1 Distribution

During the Stone Age, harbour seals were distributed in the southern Baltic, south of a line north of Gotland (Sweden) to north of Saaremaa (Estonia). The present distribution is more restricted as compared with both archaeological data and hunting statistics. Harbour seals have disappeared from the southern part of Gotland, where reproducing animals were observed in the 1980s, and also from the east coast of Öland and the northern part of the Kalmarsund. The present distribution is limited to three localities in the Kalmarsund region in Sweden.

2.4.2 Historical and current population size

Model results show that the maximum abundance over the past 8,000 years occurred just prior to 1905, followed by a decline up to 1960 (Figure 2.4.2.1). Harbour seal numbers reached a minimum from 1960 to 1985, after

Figure 2.4.2.1. Model of population trends of harbour seals in the Baltic from 8,000 years ago, when it was formed, and up to the present. The maximum abundance is indicated to have occurred just prior to 1905, followed by a decline up to 1960. A severe bottleneck occurred during the period 1960 to 1985, after which seal numbers increased. Although numbers of seals are indicated to have been lower in earlier history compared with the latter half of the 19th century, an additional maximum abundance occurred 4,000 to 1,800 years ago (Härkönen and Harding, 2003).

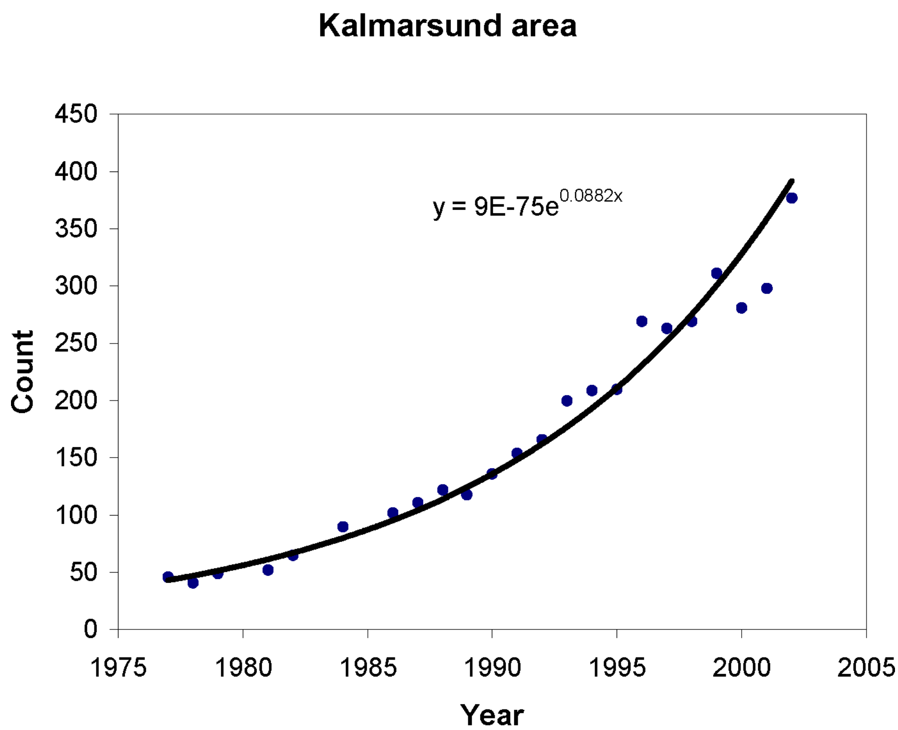


Figure 2.4.2.2. Counts of harbour seals in the Kalmarsund stock. The mean annual rate of increase was 9% in the period 1977–2002 (Härkönen and Harding, 2003).

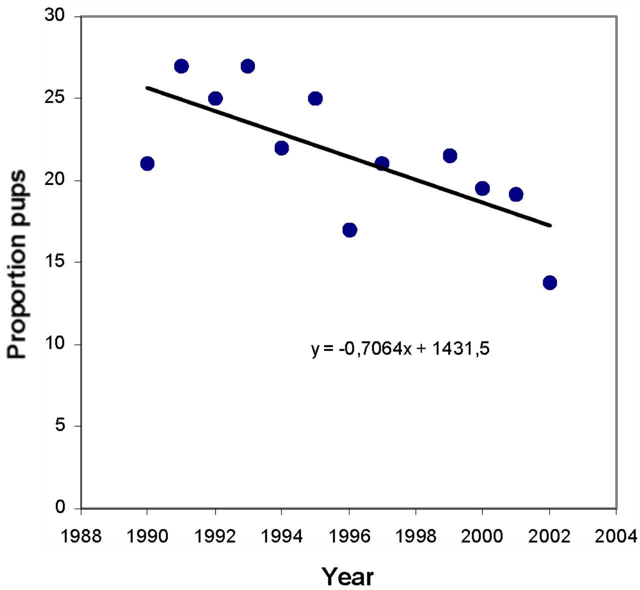


Figure 2.4.3.1. Proportion of pups in the harbour seal population in Kalmarsund. Pups were counted in June, while total population sizes were counted during the moult in August (Härkönen and Harding, 2003).

which seal numbers increased. Although numbers of seals are indicated to have been lower in earlier history compared with the latter half of the 19th century, another peak in abundance occurred 4,000 to 1,800 years ago (Härkönen and Harding, 2003).

The results of surveys along the Swedish coast in the mid-1970s showed 50 seals (Figure 2.4.2.2). An increase to 377 harbour seals had occurred by 2002, which corresponds to an annual rate of increase of 9%. There is a suggestion that the rate of increase may have slowed during the 1990s. This population was not affected by the PDV epizootics in 1988 or 2002 (T. Härkönen, pers. comm.).

2.4.3 Reproductive capacity

For the period 1990 to 2002, the number of pups as a percentage of the total counted population size showed a decreasing trend ($P<0.003$) (Figure 2.4.3.1).

2.4.4 Effect of contaminants and health status, and interactions with commercial fisheries and intentional killing

Organochlorines had negative effects on the reproductive capacity of both ringed seals and grey seals from the 1960s to the 1980s (Helle, 1986; Bergman and Olsson, 1986; Bergman, 1999). Such effects are also suggested for harbour seals in the period 1977–1989 (Härkönen *et al.*, 2002). There is no current information on health status or contaminants in this population.

Eel pound nets close to the major haul-out site caused high by-catches in the past, but they are now closed down. This has reduced the fishery mortality. Two individuals are known to have drowned in eel fyke nets in 2001, but there is no recent estimate of total by-catch. There is no intentional killing of these seals.

2.5 Harbour seal (*Phoca vitulina*) (southwest Baltic and Kattegat stock)

2.5.1 Distribution and historical and current population size

The species is distributed on all suitable sandbanks and islands of the southwestern Baltic and Kattegat. There are no seasonal changes in distribution.

Population dynamics prior to the 1988 seal epizootic are described in Heide-Jørgensen and Härkönen (1988). About 56% of harbour seals in the area died in the 1988 seal epizootic. After the 1988 epizootic, the population grew from approximately 5,000 to more than 10,000 seals just prior to the 2002 epizootic, when again more than half of the population died.

In the southwestern Baltic itself, the first surveys started in 1990, when 224 seals were counted. In 1998, numbers had increased to 315, which corresponds to a 4.8% annual rate of increase. In 2002 this population was hit by the PDV epizootic and reduced by 50%.

Current population estimates and trends cannot be determined before additional censuses that are scheduled for August 2003.

2.5.2 Reproductive capacity, effect of contaminants and health status, and interactions with commercial fisheries and intentional killing

The low rate of population increase in the area, compared to the Skagerrak, prior to the last epizootic is an indication of reduced reproductive capacity (Härkönen *et al*., 2002).

During the seal epizootic in 1988 more than 1,000 lower jaws were collected in the Kattegat, Skagerrak, and the Baltic. Subsequent analyses revealed a high prevalence of alveolar exostosis, not found at all in reference material collected from 1850–1930. Similar changes in Baltic grey seals were thought to be indicative of organochlorine pollution (Mortensen *et al*., 1992; Härkönen *et al*., 2002).

By-catches of harbour seals amounted to about 300 individuals in the Swedish fishery on the west coast in 2001 (Lunneryd *et al.*, 2003). A high proportion of the by-catch is in lumpsucker and flatfish bottom-set gillnets. No information on by-catch is available from Denmark. In 2002, licences for a total of six animals were issued in Sweden to kill harbour seals, three were shot, and three licences were issued for fourteen animals (but only five were shot) in Denmark in the same period.

2.6 Grey seal (*Halichoerus grypus*)

2.6.1 Distribution and historical and current population size

In the Baltic, the grey seal is migratory and distribution varies between seasons. During the breeding season, the distribution of the species is dependent on ice conditions in the central Baltic and the main breeding areas can be found from 57°N up to the ice edge. The largest concentrations of grey seals outside the breeding season are found in the northern Baltic. Studies of long-distance movements show that some individuals move throughout the Baltic during the ice-free period. Grey seals do not breed at present on the southern coast of the Baltic.

At the beginning of the 20th century, the minimum population size was about 100,000 individuals (Harding and Härkönen, 1999). Total population size estimated using a capture-recapture method based on photo-ID (Hiby *et al*., 2001, 2003) for the year 2000 was 12,053 (95% CI 8,073–14,051). Counts on the Swedish coast provide a sufficiently long time series for trend analysis. Here, the mean annual population increase was 7.8% for the period 1990 to 2002. The intrinsic rate of increase for east Atlantic grey seals was found to be about 10% (Harding *et al*., 2003).

2.6.2 Reproductive capacity and effect of contaminants and health status

Pathological studies (Bergman, 1999) and population trends suggest that the reproductive capacity of the species has improved since the 1970s. Pregnancy rates in the material collected over the period 1985–1996 were 60% (Bergman, 1999).

The general health status of grey seals in the Baltic has improved, but colonic ulcers caused by hookworms have increased in frequency and renal lesions persist. Colonic ulcers are the second most important cause of death after incidental catching and hunting (Bergman, 1999).

2.6.3 Interactions with commercial fisheries and intentional killing

In Estonia, 150 seals were estimated to have been by-caught in commercial fisheries in 2001, based on interviews with fishermen (I. Jüssi, pers. comm.). In Poland, seven seal corpses were delivered to the Hel marine station in 2001 (K. Skóra and I. Kuklik, pers. comm.). An interview survey of the Swedish fishery gave an estimated total by-catch of 430 grey seals in 2001 (Lunneryd *et al*., 2003). Approximately 2/3 of this was in the Baltic Proper, where most of the fishing activity is located. About half of the by-catches occur in bottom-set gillnets for turbot and cod. In the Gulf of Bothnia, the majority of incidental catches are in salmon traps. A similar estimate of grey seal by-catches was made in 1996 (Lunneryd and Westerberg, 1997). The number of by-catches has increased slightly, but the relative proportion of by-caught animals has decreased from approximately 14% of the counted population in Swedish waters to less than 10%. The reason for this is probably a change in fishing practice as an adaptation to increasing seal interaction. There are no recent data on by-caught grey seals from other Baltic countries.

The yearly hunting quota in 2002–2003 totals 430 grey seals for the Baltic Sea area (Sweden 200, Finland 230). In addition, permits to hunt grey seals are issued in Åland as mitigation to the seal-fisheries conflicts. No upper limit is set for those permits and 156 permits were granted in 2002–2003. Thus, the total allowable take by legal hunting adds up to 586 grey seals. The actual number of seals that were reported killed has been much less than half of the allowable catch, but hunting losses are unknown.

To decrease fishery interactions, Finland has established seven nature reserves, mainly for grey seals, but some for ringed seals also, in the Baltic Sea where fishing is banned (Sipilä, 2003b).

At least 35 grey seal pups were killed illegally in one grey seal breeding colony at the Estonian coast in the spring of 2002.

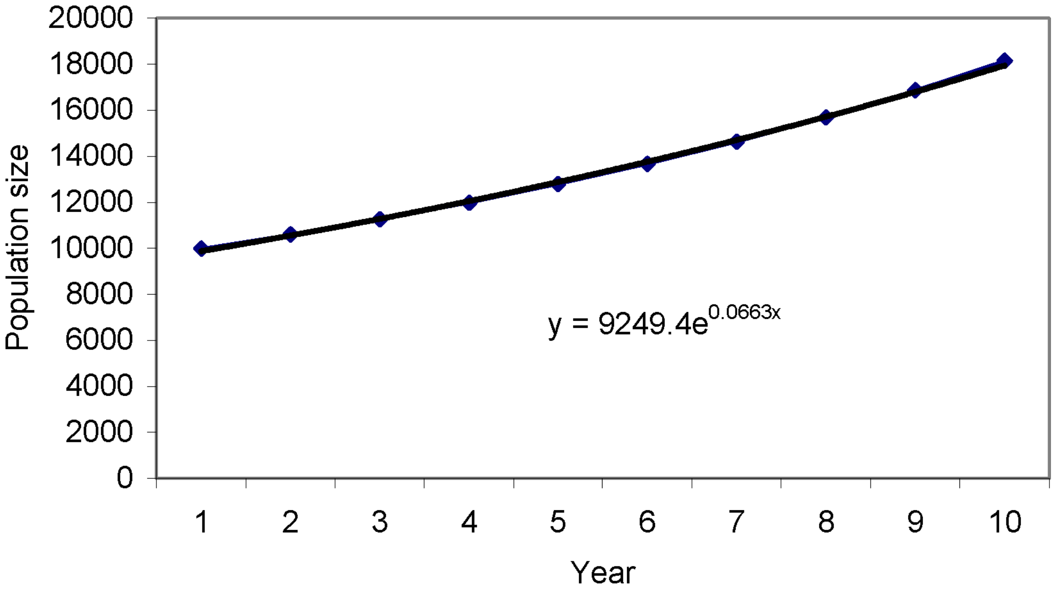


Figure 2.6.4.1. Modelled population growth of grey seals under the scenario where the initial population size is 10,000, the maximum rate of increase is 1.10, and the annual by-catch is 800 individuals where pups of the year comprise 2/3 or more of the by-catch.

2.6.4 Conclusions

A simple model was constructed to study whether combinations of observed data on population size, rate of increase, and size of by-catches are realistic for the Baltic grey seal population. Assuming a theoretical size of the initial population at 10,000 and an annual rate of 800 by-caught individuals, rates of population increase were modelled under conditions where the catch was according to the stable age distribution, where 33% were pups, and finally where 66% were pups. It was found that the data are realistic if 66% or more of the by-catch is of pups of the year (Figure 2.6.4.1).

2.7 Harbour porpoise (*Phocoena phocoena*)

2.7.1 Distribution, migration, and stock identity

The relative abundance of harbour porpoises in the Baltic decreases from west to east from the Belt Seas towards the Baltic Proper (Hammond *et al.*, 2002; Berggren *et al.*, 2002; Gillespie *et al.*, 2003). Occurrence in the eastern Baltic is occasionally reported (Karalius, pers. comm.; ASCOBANS, 2000).

Several studies of morphology, genetics, and contaminant loads indicate that harbour porpoises in the Baltic Sea are distinct from animals in the Skagerrak/Kattegat areas (Börjesson and Berggren, 1997; Wang and Berggren, 1997; Berggren *et al.*, 1999). Population-level differences have been found between porpoises from the Belt Seas and the North Sea (Kinze, 1985; Andersen, 1993), between the Kiel/Mecklenburg Bights and the North Sea (Tiedeman *et al.*, 1996; Huggenberger, 1997), and between the Skagerrak/Kattegat areas and the west coast of Norway (Wang and Berggren, 1997). Further, porpoises in the Kiel/Mecklenburg Bights and the Baltic Sea are distinct on genetic and morphological grounds (Tiedeman *et al.*, 1996; Huggenberger, 1997; Huggenberger *et al.*, 2002). There are indications of seasonal migrations of porpoises between Danish inner waters and the North Sea (Teilmann *et al.*, 2003).

In summary, two populations of harbour porpoise are considered to live in the area: one in the Baltic Proper and one in the eastern part of the Skagerrak, Kattegat, Belt Sea, Kiel Bight and Mecklenburg Bight to the Darss sill in the east.

2.7.2 Historical and current population size

Available information indicates declining abundances, but former population levels are not known.

The abundance of harbour porpoises in the Baltic Sea was estimated during a line-transect aerial survey in July 1995 (Hiby and Lovell, 1996). The survey covered a 43,000 km^2 area (corresponding to ICES Sub-divisions 24 and 25, but excluding a 22 km wide corridor along the Polish coast) and yielded an estimate of 599 animals (Table 2.7.2.1). The abundance estimate for the Baltic Sea was based on sightings of only three groups, each containing a single animal. The abundance estimate was inevitably accompanied by a large confidence interval. The same crew also covered the Kiel and Mecklenburg Bight area in July 1995 and the resultant estimate was 817 animals (Hiby and Lovell, 1996). A ship-based line-transect survey of Polish coastal waters in 2001 saw only one harbour porpoise, thus rejecting the idea that these waters hold a large population of harbour porpoises

(Berggren *et al.*, 2002.). Abundance estimates for other species are not available for this region.

An aerial survey of German and some southern Danish waters was undertaken from May to August 2002 (Figure 2.7.2.1). This survey found the highest relative abundance of porpoises in the Pomeranian Bight between the island of Rügen and the Polish border (Scheidat *et al.*, 2003). The maximum group sizes in this area were ten animals. Repeated flights in August, September, December, February, and March in the same area did not find a single porpoise (M. Scheidat, pers. comm.). This demonstrated that the overall density of porpoises was lower between the island of Rügen and the Polish border than indicated through the surveys in May and July.

No information is available for assessing any trend in abundance. Two aerial surveys were conducted in the Baltic Sea in 2002 by Germany and Sweden, but the data have not yet been published. Another large-scale abundance survey is planned for 2005.

2.7.3 Reproductive capacity, and effect of contaminants and health status

There is no new information available on reproductive capacity.

A large number of different lesions and pathological changes are reported from the Baltic Sea. Typical autopsy findings include heavy attack from parasites in the lung, liver, stomach, intestines and middle ear cavities, skin lesions, and pneumonia. Other findings are liver fibrosis, arthrosis, and abscesses in muscles, lungs, and other organs (Siebert *et al.*, 1999; Clausen and Andersen, 1988). Animals from the Baltic also had 41% to 254% higher mean levels of PCDD/Fs and PCBs than corresponding samples from the Kattegat and Skagerrak (Berggren *et al.*, 1999; Bruhn, *et al.*, 1999).

Table 2.7.2.1. Abundance estimates for harbour porpoises in the Baltic Sea, Belt Seas, Kiel and Mecklenburg Bights, Kattegat, and Skagerrak.

Year of estimate	ICES Area	Abundance estimate	95% Confidence limits	Method	Reference
1994	IIIa + b	36,046	20,276–64,083	Ship-based	Hammond *et al.*, 2002
	IIIc	588	(CV 0.48)		
1995	24+25	599	200–3,300	Aerial survey, line transect	Hiby and Lovell, 1996
	K&M Bights	817	300–2,400		

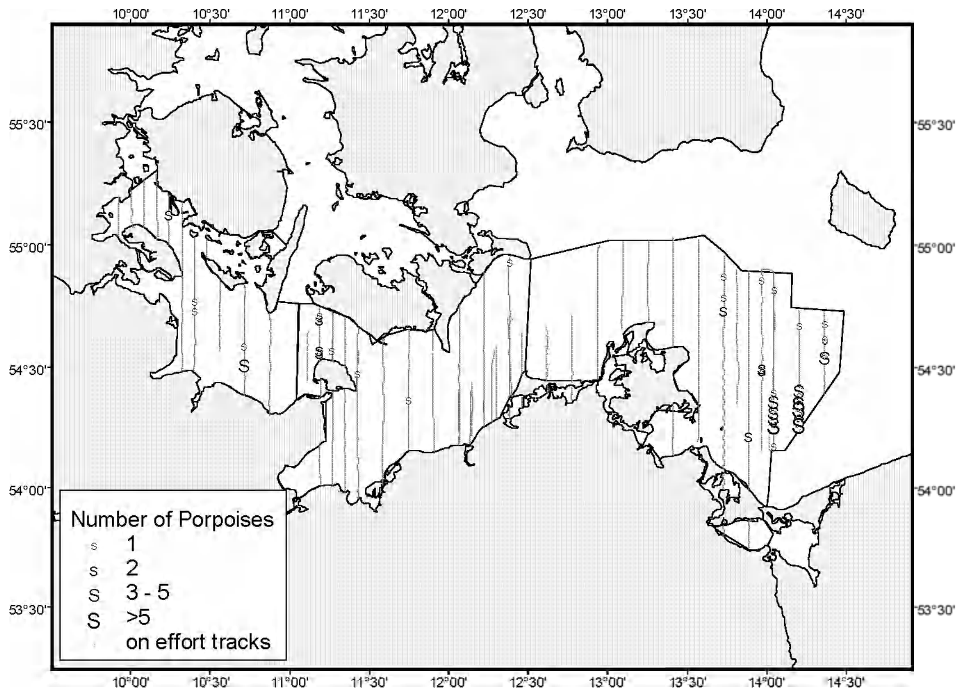


Figure 2.7.2.1. German aerial surveys in the Baltic (May to August 2002) (Scheidat *et al.*, 2003).

2.7.4 Interaction with commercial fisheries

Incidental mortality in fishing gear represents the most significant threat to porpoise populations (Teilmann and Lowry, 1996; ASCOBANS, 2002). By-catch is known to occur in different types of fisheries, but no reliable estimates are available for any large part of the Baltic, including the Kattegat (Koschinski, 2002).

As in other areas, harbour porpoises are believed to be subject to incidental takes in gillnet fisheries. Atlantic salmon driftnet fisheries were suggested to have taken substantial numbers of harbour porpoises in the past (Ropelewski, 1957; Lindroth, 1962). There have been very few studies in the Baltic to the east of the island of Rügen. No by-catches were reported by a Danish observer programme (350 km.days of net observed (less than 0.5% of total net days in this fishery)) between 1992 and 1998 (Vinther, 1999) or in more recent years (F. Larsen, pers. comm.). Berggren (1994) used reports from Swedish fishermen to estimate a minimum catch of about five harbour porpoises per year in the early 1990s. Most of these were taken in salmonid driftnets or cod gillnets. The scale of the fishery has declined over the past twenty years, so it is likely that the harbour porpoise by-catch has also declined. The Swedish turbot fishery has not reported a substantial by-catch (Berggren, 1994). A total of six nights were spent at sea by an observer on salmon driftnet vessels; no by-catch was recorded by this observer, but one was reported from a non-observed vessel (Harwood *et al.*, 1999).

In a study of the distribution of by-catch in Polish waters, Kuklik and Skóra (2003) reported that 45 dead harbour porpoises from by-catch were notified to Hel Marine Station over ten years, with nearly half of these from Puck Bay (Figure 2.7.4.1) and 40% in semi-driftnet fisheries for salmonids. A further third of the by-catch was in set-nets for cod (Table 2.7.4.1). Two by-catches were reported from Finland between 1986 and 1999 (ASCOBANS, 2000). In other countries' fisheries in the Baltic Proper, there is either no information (Lithuania) or no by-catch reported or believed to occur (Germany, Russia, Latvia, Estonia).

No by-catch was reported by the Danish fishery observer programme (193 km.days observed (less than 0.5% of total net days in this fishery)) in the Belt Seas between 1992 and 1998 (Vinther, 1999). Based on interviews with fishermen, K.-H. Kock (pers. comm.) estimated a catch of about 3–5 harbour porpoises per year in German fisheries in this area.

Studies on by-catches of harbour porpoises in set-net fisheries were conducted on the Swedish cod and pollack fisheries in the Kattegat in 1996–1997 (Harwood *et al.*, 1999). A total of 7,441 net km.hrs was observed over three seasons of the year in two ICES rectangles on the Skagerrak/Kattegat boundary. A total of twelve porpoises were seen as by-catch, while a further thirteen animals were reported as by-catch on unobserved vessels fishing in the same rectangles. Based on these figures, these authors extrapolated a catch of 105 animals per 10,000 net km.hrs in the Skagerrak/Kattegat combined. The Swedish fisheries targeting cod and pollack decreased by 59% between 1997 and 2000 due to the reduction in the stock size of cod. The overall effort in Swedish set-net fisheries decreased by 45% during this period (data from the Swedish National Board of Fisheries). Vinther (1999) reported observations of 329 net km.days between 1995 and 1998 on Danish set-net fisheries in the Kattegat and Skagerrak. A total of five porpoises were observed as by-catch in one ICES rectangle; four of these were caught in the lumpfish fishery. This equates to fifteen animals by-caught per 1,000 net km.days.

Table 2.7.4.1. By-catch of harbour porpoises in different types of fishing nets in 1990–1999 in Poland (Kuklik and Skóra, 2003).

Year	Total number of by-caught animals	Type of nets					
		Semi-driftnets (salmon)	Bottom-set gillnets		Herring gillnets	Herring trawl nets	Other set nets
			Cod	Others			
1990	1	1					
1991	7	3	1	2			1
1992	5		1	2			2
1993	7	4	1	2			
1994	3	1	1		1		
1995	5	4				1	
1996	10	4	5	1			
1997	2	1	1				
1998	3		3				
1999	2		2				
Total	45	18	15	7	1	1	3
%	100	40.0	33.3	15.5	2.2	2.2	6.8

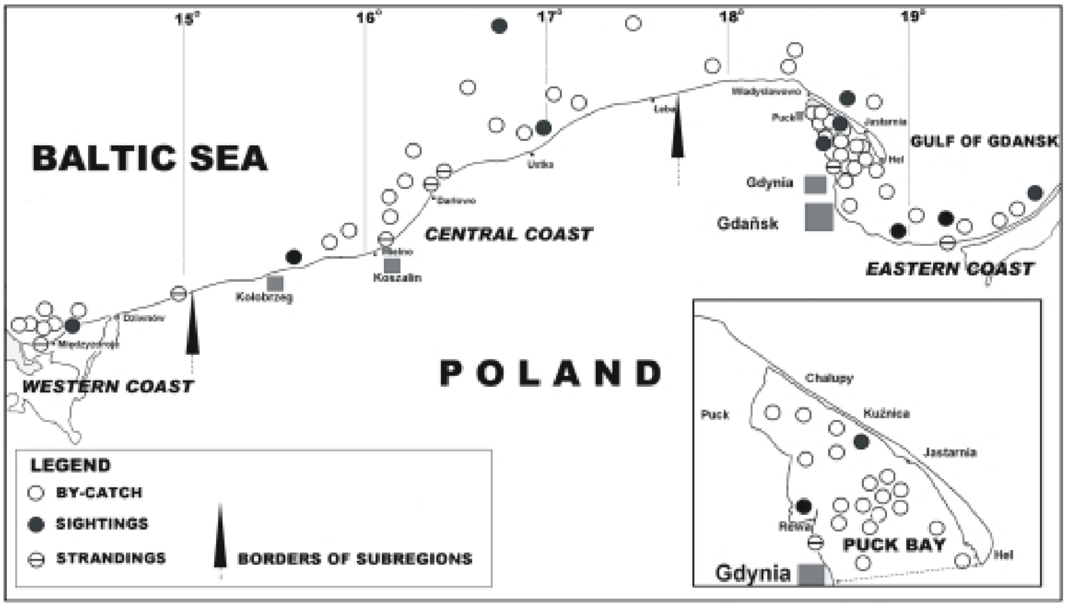


Figure 2.7.4.1. Places where harbour porpoises were sighted, by-caught, or washed ashore on the Polish coast between 1990 and 1999 (Kuklik and Skóra, 2003).

References

Andersen, L.W. 1993. The population structure of the harbour porpoise, *Phocoena phocoena*, in Danish waters and part of the North Atlantic. Marine Biology, 116: 1–7.

Anon. 1939. Eesti statistika 1931–1939, Vol. 17. Tallinn (Estonian Statistics, in Estonian).

Antoniuk, A.A. 1975. Estimation of the population size of the ringed seal in Lake Ladoga. Zoolicheschkii Zhurnal, 54: 1371–1377.

ASCOBANS. 2000. Fourth annual compilation of national reports. ASCOBANS Secretariat, Bonn.

ASCOBANS. 2002. ASCOBANS Recovery plan for Baltic harbour porpoises (Jastarnia plan). Agreement on the Conservation of Small Cetaceans of the Baltic and North Seas, Bonn.

Becker, P. 1984. Uhanalaisuuden syyt. *In* Saimaannorppa, Saimaannorppa, pp. 109–118. Ed. by P. Becker. Suomen Luonnonsuojelun Tuki Oy, Helsinki.

Bergman, A. 1999. Health condition of the Baltic grey seal (*Halichoerus grypus*) during two decades. Gynecological health improvement but increased prevalence of colonic ulcers. APMIS, 107: 270–282.

Bergman, A., and Olsson, M. 1986. Pathology of Baltic grey seal and ringed seal females with special reference to adrenocortical hyperplasia: is environmental pollution the cause of a widely distributed disease syndrome? Finnish Game Research, 44: 47–62.

Bergman, G. 1956. Sälbeståndet vid våra kuster. Nordenskiöld-samfundets tidskrift, 16: 50–65.

Berggren, P. 1994. By-catches of the harbour porpoise (*Phocoena phocoena*) in the Swedish Skagerrak, Kattegat and Baltic waters, 1973–93. Report of the International Whaling Commission, Special Issue, 15: 211–216.

Berggren, P., Brown, S., Gillespie, D., Kuklik, I., Lewis, T., Matthews, J., McLanaghan, R., Moscrop, A., and Tregenza, N. 2002. Passive acoustic and visual survey of harbour porpoises (*Phocoena phocoena*) in Polish coastal waters confirms endangered status of Baltic population. Paper at 16th annual European Cetacean Society Conference, Liege, 7–11 April 2002.

Berggren, P., Ishaq, R., Zebür, Y., Näf, C., Bandh, C., and Broman, D. 1999. Patterns and levels of organochlorine contaminants (DDTs, PCBs, non-ortho PCBs and PCDD/Fs) in male harbour porpoises (*Phocoena phocoena*) from the Baltic Sea, the Kattegat-Skagerrak Seas and the west coast of Norway. Marine Pollution Bulletin, 12: 1070–1084.

Börjesson, P., and Berggren, P. 1997. Morphometric comparisons of skulls of harbour porpoises (*Phocoena phocoena*) from the Baltic, Kattegat and Skagerrak seas. Canadian Journal of Zoology, 75: 280–287.

Bruhn, R., Kannan, N., Petrick, G., Schulz-Bull, D.E., and Duinker, J.C. 1999. Persistent chlorinated organic contaminants in harbour porpoises from North Sea, the Baltic Sea and Arctic waters. Science of the Total Environment, 237/238: 351–361.

Chapskii, K.K. 1932. Ladozhski tiulen i vozmozhnost ego promysla. Report of the Fisheries Research Institute of Leningrad, 13: 147–157.

Clausen, B., and Andersen, S.H. 1988. Evaluation of bycatch and health status of the harbour porpoise (*Phocoena phocoena*) in Danish waters. Danish Review of Game Biology, 13: 1–20.

Filatov, I.E. 1990. Ladozhkaja kolchataja nerpa. *In* Redkie izthezajushie vidy mlekopitajushih SSSR, pp. 57–64. Ed. by V.E. Sokolov and A.L. Janshin. Nauka, Moskva.

Gillespie, D., Berggren, P., Brown, S., Kuklik, I., Lacey, C., Lewis, T., Matthews, J., McLanaghan, R., Moscrop, A., and Tregenza, N. 2003. The relative abundance of harbour porpoises (*Phocoena phocoena*) from acoustic and visual surveys in German, Danish, Swedish and Polish waters during 2001 and 2002. Working paper presented at the 2003 ASCOBANS meeting, 7 pp.

Greve, K. 1909. Säugtiere Kur-Liv-Estlands. Riga. 184 pp.

Hammond, P.S., Berggren, P., Benke, H., Borchers, D.L., Collet, A., Heide-Jørgensen, M.P., Heimlich, S., Hiby, A.R., Leopold, M.F., and Øien, N. 2002. Abundance of the harbour porpoise and other cetaceans in the North Sea and adjacent waters. Journal of Applied Ecology, 39: 361–376.

Harding, K., and Härkönen, T. 1999. Development in the Baltic grey seal (*Halichoerus grypus*) and ringed seal (*Phoca hispida*) populations during the 20th century. Ambio, 28: 619–627.

Harding, K., Härkönen, T., and Helander, B. 2003. Ecological risk analysis of the Baltic gray seal population (*Halichoerus grypus*). Working paper presented to the 2003 meeting of the Working Group on Marine Mammal Ecology.

Härkönen, T., and Harding, K.C. 2003. Historical and present status of the harbour seals in the Baltic proper. Working paper presented to the 2003 meeting of the Working Group on Marine Mammal Ecology.

Härkönen, T., Harding, K., and Heide-Jørgensen, M-P. 2002. Rates of increase in age-structured populations: a lesson from the European harbour seals. Canadian Journal of Zoology, 80: 1498–1510.

Härkönen, T., and Heide-Jørgensen, M.-P. 1990. Density and distribution of the ringed seal in the Bothnian Bay. Holarctic Ecology, 13: 122–129.

Härkönen, T., Jüssi, M., Jüssi, I., Stenman, O., and Sagitov, R. 2003. Status of the Baltic ringed seal (*Phoca hispida botnica*). Working paper presented to the 2003 meeting of the Working Group on Marine Mammal Ecology.

Härkönen, T., and Lunneryd, S.G. 1992. Estimating abundance of ringed seals in the Bothnian Bay. Ambio, 21: 497–503.

Härkönen, T., Stenman, O., Jüssi, M., Jüssi, I., Sagitov, R., and Verevkin, M. 1998. Population size and distribution of the Baltic ringed seal (*Phoca hispida botnica*). *In* Ringed seals (*Phoca hispida*) in the North Atlantic. Ed. by M.-P Heide-Jørgensen and C. Lydersen. NAMMCO Scientific Publications, 1: 167–180.

Harwood, J., Andersen, L.W., Berggren, P., Carlström, J., Kinze, C.C., McGlade, J., Metuzals, K., Larsen, F., Lockyer, C.H., Northridge, S., Rogan, E., Walton, M., and Vinther, M. 1999. Assessment and reduction of the by-catch of small cetaceans (BY-CARE). Final report to the European Commission on FAIR-CT05-0523.

Heide-Jørgensen, M.-P., and Härkönen, T. 1988. Rebuilding seal stocks in the Kattegat-Skagerrak. Marine Mammal Science, 4: 231–246.

HELCOM. 2001. Conservation and management of seal population in the Baltic. Action Plan for the implementation of the HELCOM Project on Seals. November 2001.

Helle, E. 1980a. Aerial census of ringed seals *Pusa hispida* basking on the ice of the Bothnian Bay, Baltic. Holarctic Ecology, 3: 183–189.

Helle, E. 1980b. Lowered reproductive capacity in female ringed seals (*Pusa hispida*) in the Bothnian Bay, northern Baltic Sea, with special reference to uterine occlusions. Annals Zoologica Fennica, 17: 147–158.

Helle, E. 1985. Ympäristömyrkyt ja Suomen hylkeet. Suomen Riista, 32: 5–21.

Helle, E. 1986. The decrease in the ringed seal population of the Gulf of Bothnia in 1975–84. Finnish Game Research, 44: 28–32.

Helle, E. 1990. Norppakannan kehitus Perämerellä 1980-luvulla. Suomen Riista, 36: 31–36.

Helle, E., Hyvärinen, H., Pyysalo, H., and Wickström, K. 1983. Levels of organochlorine compounds in an inland seal population in eastern Finland. Marine Pollution Bulletin, 14: 256–260.

Helle, E., Hyvärinen, H., and Sipilä, T. 1981. Yhdestoista hetki saimaanhylkeen suojelussa. Suomen Luonto, 40: 423–425. (Summary: The Saimaa ringed seal - Symbol of successful nature protection measures, or a bitter defeat?)

Helle, E., Hyvärinen, H., and Stenman, O. 1985. PCB and DDT levels in the Baltic and Saimaa seal populations. Finnish Game Research, 44: 63–68.

Helle, E., Olsson, M., and Jensen, S. 1976. PCB levels correlated with pathological changes in seal uteri. Ambio, 5: 5–6.

Helle, E., and Stenman, O. 1990. (Baltic seal populations in 1986–1990) (In Finnish or Swedish with English summary). Maailman Luonnon Säätiön WWF Suomen Rahaston Raportteja 1. 76 pp.

Helminen, M., Karppanen, E., and Koivisto, I. 1968. Saimaannorpan elohopeapitoisuudesta 1967. Suomen Eläinlääkärilehti, 74: 87–89.

Henriksson, K., Karppanen, E., and Helminen, M. 1969. Mercury in inland and marine seals. Nordisk hygienisk tidskrift, 50: 54–59.

Hiby, L., and Lovell, P. 1996. Baltic/North Sea aerial surveys – final report. (Unpublished). 11 pp.

Hiby, L., Lundberg, T., Karlsson, O., and Helander, B. 2001. An estimate of the size of the Baltic grey seal population based on photo-id data. Report to project "Seals and fisheries" Länstyrelsen I Västernorrlands Län and Naturvårdsverket. 29-11-2001.

Hiby, L., Lundberg, T., Karlsson, O., Watkins, J., and Helander, B. 2003. An estimate of the size of the Baltic grey seal population based on photo-id data. Working paper presented to the 2003 meeting of the Working Group on Marine Mammal Ecology.

Huggenberger, S. 1997. Schweinswale (*Phocoena phocoena* L.) der deutschen Nord- und Ostsee. Diplomarbeit der Universitat zu Köln (in German).

Huggenberger, S., Benke, H., and Kinze, C.C. 2002. Geographical variation in harbour porpoise (*Phocoena phocoena*) skulls: support for a separate non-migratory population in the Baltic Proper. Ophelia, 56: 1–12.

Hyvärinen, H., and Sipilä, T. 1992. Saimaannorppa, *Phoca hispida saimensis*, pp. 68–71. Ed. by U. Elo. *In* Maailman Uhanalaiset Eläimet, Suomi, Weiling+ Göös, Vantaa.

Hyvärinen, H., Sipilä, T., Kunnasranta, M., and Koskela, J. 1998. Mercury pollution and the Saimaa ringed seal (*Phoca hispida saimensis*). Marine Pollution Bulletin, 36: 76–81.

Jääskeläinen, V. 1942. Laatokka kalavetenä, pp. 38–45. Ed. by V. Näsi and E. Ovaska. *In* Laatokan mainingit. Laatokan ja sen rannikon elämää sanoin ja kuvin. Otava, Helsinki.

Kinze, C.C. 1985. Intraspecific variation in Baltic and North Sea harbour porpoise (*Phocoena phocoena* (L., 1758)). Videnskabelige Meddelelser Dansk Naturhistorisk Forening, 146: 63–74.

Kokko, H., Helle, E., Lindstrom, J., Ranta, E., Sipilä, T., and Courchamp, F. 1999. Backcasting population sizes of ringed and grey seals in the Baltic and Lake Saimaa during the 20th century. Annals Zoologica Fennica, 36: 65–75.

Koschinski, S. 2002. Current knowledge on harbour porpoise (*Phocoena phocoena*) in the Baltic Sea. Ophelia, 55: 167–197.

Kostamo, A., Medvedev, N., Pellinen, J., and Kukkonen, J. 2000. Analysis of organochlorine compounds and extractable organic halogen in three subspecies of ringed seal in Northeast Europe. Environmental Toxicology and Chemistry, 194: 848–854.

Kuklik, I., and Skóra, K.E. 2003 (in press). By-catch as a potential threat for harbour porpoise (*Phocoena phocoena* L.) in the Polish Baltic Waters. NAMMCO Scientific Publications.

Lindroth, A. 1962. Baltic salmon fluctuations 2: porpoise and salmon. Report of the Institute of Freshwater Research, Drottningholm, 44: 105–112.

Lunneryd, S.G., and Westerberg, H. 1997. By-catch of and gear damages by grey seal (*Halichoerus grypus*) in Swedish waters. ICES CM 1997/Q:11. 10 pp.

Lunneryd, S-G., Königson, S., and Westerberg, H. 2003. Bycatch of seals in the Swedish fishery in 2001. Working paper to the 2003 meeting of the Working Group on Marine Mammal Ecology.

Marttinen, A. 1946. Mikä on Saimaan hylkeen kohtalo. Metsästys ja kalastus 1946: 116–118.

Mattson, M., and Helle, E. 1995. Reproductive recovery and PCBs in Baltic seal populations. Eleventh Biennial Conference on the Biology of Marine Mammals, 14–18 December 1995, Orlando, Florida, U.S.A. Abstracts, p. 74.

Medvedev, N., Sipilä, T., Hyvärinen, H., and Kunnasranta, M. 1996. Proposals for the protection of the Ladoga seal. *In* Proceedings of the second international Lake Ladoga Symposium 1996. Ed. by V. Simile, M. Viljanen, and T. Sepukhina. Univ. of Joensuu, publications of Karelian Institute, No. 117. Joensuu. 302 pp.

Mortensen, P., Bergman, A., Bignert, A., Hansen., H.J., Härkönen, T., and Olsson, M. 1992. Prevalence of skull lesions in harbour seals *Phoca vitulina* in Swedish and Danish museum collections during 1835–1988. Ambio, 21: 520–524.

Nyman, M., Raunio, H., Taavistainen, P., and Pelkonen, O. 2001. Characterization of xenobiotic metabolizing cytochrome P450 (CYP) forms in ringed and grey seals from the Baltic Sea and reference sites. Comparative Biochemistry and Physiology, Part C, Toxicology and Pharmacology, 128: 99–112.

Reeves, R.R. 1998. Distribution, abundance and biology of the ringed seals (*Phoca hispida*): an overview. *In* Ringed seals in the North Atlantic. Ed. by M.-P. Heide-Jørgensen and C. Lydersen. NAMMCO Scientific Publication, 1: 9–45.

Rezvov, G. 1975. On the distribution of pupping ringed seals in the Gulf of Finland and its dependence on the severity of winters. Marine Mammals, Kiev, 2: 73–74. (In Russian).

Ropelewski, A. 1957. The common porpoise (*Phocoena phocoena* L.) as a by-catch in Polish Baltic fisheries. Prace Morskiego Instytutu Rybackiego, 9: 427–437. (in Polish)

Scheidat, M., Kock, K.-H., and Siebert, U. 2003.Summer distribution of harbour porpoise (*Phocoena phocoena*) in the German North and Baltic Sea. Working paper presented at the 2003 ASCOBANS meeting. 13 pp.

Siebert, U., Joiris, C., Holsbeek, L., Benke, H., Failing, K., Frese, K., and Petzinger, E. 1999. Potential relation between mercury concentrations and necropsy findings in cetaceans from German waters of North and Baltic Seas. Marine Pollution Bulletin, 38: 285–295.

Sipilä, T. 1981 Saimaanhylkeen suku sammuu. Pohjois-Karjalan Luonto, 11(1): 21–23.

Sipilä, T. 1983. Saimaanhyljetutkimus, tutkimusraportti 1981–1982. Laudatur – Erikoistyö, Joensuun korkeakoulu, Biologian laitos. 68 pp.

Sipilä, T. 1990. Lair structure and breeding habitat of the Saimaa ringed seal (*Phoca hispida saimensis*). Finnish Game Research, 47: 11–20.

Sipilä, T. 1992. Saimaannorpan (*Phoca hispida saimensis* Nordq.) pesintä-, populaatio- ja suojelubiologiasta. Lisensiaatintutkielma, Biologian laitos, Joensuun yliopisto. 45 pp.

Sipilä, T. 1994. Saimaannorppa, alkuperäinen savolainen. *In* Ympäristön tila Mikkelin läänissä, pp. 91–94. Ed. by P. Tahvanainen. Vesi- ja ympäristöhallitus, Helsinki. (in Finnish)

Sipilä, T. 2003a. Conservation biology of Saimaa ringed seal (*Phoca hispida saimensis*). PhD Thesis.

Sipilä, T. 2003b. Nature protection areas for seals in Baltic Sea, Finland. Working paper presented to the 2003 meeting of the Working Group on Marine Mammal Ecology.

Sipilä, T., Helle, E., and Hyvärinen, H. 1990. Distribution, population size, and reproductivity of the Saimaa ringed seal (*Phoca hispida saimensis* Nordq.) in Finland 1980–84. Finnish Game Research, 47: 3–10.

Sipilä, T., and Hyvärinen, H. 1998. Status and biology of Saimaa (*Phoca hispida saimensis*) and Ladoga (*Phoca hispida ladogensis*) ringed seals. *In* Ringed seals in the North Atlantic. Ed. by M.-P. Heide-Jørgensen and C. Lydersen. NAMMCO Scientific Publication, 1: 83–99.

Sipilä, T., Hyvärinen, H., and Medvedev, N. 1996. The Ladoga seal (*Phoca hispida ladogensis* Nordq.). Hydrobiologia, 322: 192–198.

Sipilä, T., and Koskela, J. 2003. Mortality of Saimaa seal 1994–2002. Working paper presented to the 2003 meeting of the Working Group on Marine Mammal Ecology.

Sipilä, T., Koskela, J.T., and Hyvärinen, H. 1999. Mortality of the Saimaa seal (*Phoca hispida saimensis*). *In* Abstracts of the 13th Biennnial Conference on Biology of Marine Mammals, Wailea, HI, 28 Nov–28 Dec., p. 174. The Society for Marine Mammalology.

Sipilä, T., Koskela, J.T., and Hyvärinen, H. 2002a. Mortality in the Saimaa ringed seal (*Phoca hispida saimensis*) population. Unpublished manuscript, available from T. Sipilä.

Sipilä, T., Medvedev, N., Kunnasranta, M., Bogdanov, V., and Hyvärinen, H. 2002b. Present status and recommended conservation actions for the Ladoga seal (*Phoca hispida ladogensis*) population. WWF Suomen Rahaston Raportti No. 15. WWF, Helsinki. 30 pp.

Smith, T.G. 1987. The ringed seal, *Phoca hispida*, of the Canadian Western Arctic. Canadian Bulletin of Fisheries and Aquatic Science, 216. 81 pp.

Stanley, H.F., Casey, S., Carnahan, J.M., Goodman, S., Harwood, J., and Wayne, R.K. 1996. Worldwide patterns of mitochondrial DNA differentiation in the harbour seal (*Phoca vitulina*). Molecular Biology and Evolution, 13: 368–382.

Teilmann, J., Dietz, R., Larsen, F., Desportes, G., and Geertsen, B. 2003. Seasonal migrations and population structure of harbour porpoises (*Phocoena phocoena*) in the North Sea and Inner Danish waters based on satellite telemetry. European Cetacean Society Conference, Las Palmas de Gran Canaria, March, 2003. Abstract.

Teilmann, J., and. Lowry, N. 1996. Status of the harbour porpoise (*Phocoena phocoena*) in Danish waters. Report of the International Whaling Commission, 46: 619–625.

Tiedeman, R., Harder, J., Gmeiner, C., and Haase, E. 1996. Mitochondrial DNA patterns of harbour porpoises (*Phocoena phocoena*) from the North and Baltic Sea. Zeitschrift für Säugetierkunde, 61: 104–111.

Tormosov, D.D., and Rezvov, G.V. 1978. Information on the distribution, number and feeding habits of ringed and grey seals in the Gulfs of Finland and Riga in the Baltic Sea. Finnish Game Research, 37: 14–17.

Tormosov *et al*. unpublished field notes available to: Härkönen, T., Stenman, O., Jüssi, M., Jüssi,, I, Sagitov, R., and Verevkin, M. 1998. Population size and distribution of the Baltic ringed seal (*Phoca hispida botnica*). *In* Ringed seals (*Phoca hispida*) in the North Atlantic. Ed. by M.-P. Heide-Jørgensen and C. Lydersen. NAMMCO Scientific Publications, 1: 167–180.

Vinther, M. 1999. Bycatches of harbour porpoises (*Phocoena phocoena* L.) in Danish set-net fisheries. Journal of Cetacean Research and Management, 1: 123–135.

Wang, J.Y., and Berggren, P. 1997. Mitochondrial DNA analysis of harbour porpoises (*Phocoena phocoena*) in the Baltic Sea, the Kattegat-Skagerrak Seas and off the west coast of Norway. Marine Biology, 127: 531–537

Westerling, B., and Stenman, O. 1992. Hälsotillståndet hos sälbestånden i Östersjön. Veterinären och den yttre miljön. Nordisk kommitte för veterinärvetenskapligt samarbete 7:e symposiet 1–2 oktober 1992, Wik, Uppsala, pp. 77–83. (In Swedish).

3 MONITORING PROGRAMME FOR ESTIMATING THE ABUNDANCE OF SEALS AND OTHER MARINE MAMMAL POPULATIONS IN THE BALTIC SEA

Request

Item 6 of the 2003 requests from the Helsinki Commission, which states as follows:

In connection with the standing request, HELCOM HABITAT additionally requests a monitoring programme for estimation of the abundance of seal and other mammal populations.

There is a need to develop a monitoring programme for improved estimation of marine mammal species abundance. Countries use different methods for estimation of the number of individual mammals. Therefore surveys of different species should be standardized and synchronized in all commonly accepted methods for monitoring of species abundance.

Source of information

The 2003 Report of the Working Group on Marine Mammal Ecology (ICES CM 2003/ACE:03).

Summary

Risk analysis for grey seals and power analysis for harbour seals have proved to be powerful tools to clarify the power of monitoring programmes to detect changes. With the current survey methodology for grey seals, it would take more than nine years to detect a 5% change in annual rate of increase. The standard aerial survey methodology used for grey seal surveys during the pupping season is less suitable in the Baltic since pupping occurs on both land and ice. Three methods are proposed for monitoring grey seals in the Baltic: 1) photo-identification to derive population estimates; 2) counts during the peak moulting season to study population trends; and 3) pup counts under conditions when a majority of pups are born on land. To be useful, it is important that the project is coordinated and implemented on a permanent basis for the whole Baltic region.

For harbour seals, power analysis has shown that it will take seven years to detect a 5% change in annual rate of increase, when using the current approach of three replicate flights annually. Thus, annual replicate surveys of seals in the entire area are more effective to detect changes in trends within a reasonable time. Given a choice between replicated flights at longer intervals and single annual flights, ICES recommends the former.

For ringed seals, the strip survey technique used since the mid-1990s is recommended for continuation, since the results are less affected by subjective factors. The method is also suitable for preparing detailed ringed seal distribution maps. The continuation of lair counts is also recommended for Lake Saimaa. For Lake Lagoda, both types of surveys should be implemented.

For harbour porpoise, the standard method for estimating the abundance of cetaceans is the line-transect distance sampling methodology. This is recommended for continuation, supported by the development of new methods of analysing line-transect data to estimate population sizes. For the Baltic Proper, the very low density of harbour porpoises makes it difficult for line-transect methods to obtain a sufficient number of sightings to obtain a reliable abundance estimate with an acceptable confidence interval. The development of additional methods for areas of low density is urgently needed. One approach is to use acoustic methods to detect porpoises. A seven-step procedure to move forward in developing possible methods is proposed here.

Recommendations and advice

For monitoring grey seals in the Baltic, ICES recommends three methods: 1) photo-identification to derive population estimates; 2) counts during the peak moulting season to study population trends; and 3) pup counts under conditions when a majority of pups are born on land.

For harbour seals, ICES recommends annual replicate surveys in the entire area as the most effective method to detect changes in trends within a reasonable time. If annual surveys are not possible, ICES recommends replicate surveys at longer intervals rather than a single annual survey.

For ringed seals, ICES recommends continuation of the strip survey technique used since the mid-1990s. The continuation of lair counts is also recommended for Lake Saimaa. For Lake Lagoda, both types of surveys should be implemented.

For harbour porpoises in the Skagerrak and Kattegat, ICES recommends continuation of the standard line-transect distance sampling methodology, with the development of new methods of analysing line-transect data to estimate population sizes. To obtain information on the abundance of harbour porpoises in the Baltic Proper, the following step-wise process is proposed:

1) Review information on the historical and current distribution of harbour porpoises in the Baltic Proper, including strandings, by-catches, and sightings.
2) Develop an automated harbour porpoise click detector that can be deployed from various platforms of opportunity.

3) Find appropriate platforms of opportunity from which the acoustic detectors will be deployed. This could, for example, include ferries or research vessels.
4) Use stationary click detectors (POD—porpoise detectors) on sites chosen based on information provided by step 1 in order to confirm the presence or absence of porpoises in particular areas.
5) Deploy satellite tags on harbour porpoises of the Baltic Proper to obtain information on the movement of these animals in the Baltic Sea. A suitable area for this project could be the area around Hel, Poland, where it is possible to deploy pound nets to catch animals.
6) Analyse the data collected from step 1 to step 5 in terms of occurrence and distribution of harbour porpoises, including seasonal changes.
7) Use the available data to determine the best temporal and spatial scale for conducting a dedicated line-transect survey to determine abundance in a substratum of the Baltic Proper, using combined visual and acoustic ship-board methodology.

Scientific background

3.1 Introduction

The growing population of grey seals has led to increased fishery interactions and subsequent demands for the re-introduction of hunting. Although hunting increases the risk of quasi-extinction, the risk can be significantly reduced by the choice of a cautious hunting regime. A demographic analysis and a risk assessment of the population have shown how the risk of quasi-extinction by over-exploitation can be reduced. The best hunting regimes allow no hunting below a "security level" in population size. Obviously, to implement such a hunting regime, knowledge of the population size and growth rate is required. Risk analysis for grey seals and power analysis for harbour seals have proved to be powerful tools to clarify the power of monitoring programmes to detect changes. Risk analysis has been conducted for grey seals. With the current survey methodology, it would take more than nine years to detect a 5% change in the annual rate of increase. A hunt exceeding 300 females increases the risk for quasi-extinction substantially, but also the age and sex composition of the animals killed influences the "cost of the hunt", and thereby the risk for quasi-extinction (Harding *et al.*, 2003).

3.2 Grey seals

3.2.1 Currently used methodology

Grey seal abundance has been estimated in the Baltic Sea using photo-identification mark-recapture methodology (Hiby *et al.*, 2001, 2003). Trends in relative abundance have been monitored using direct counts of moulting animals (Jüssi and Jüssi, 2001; Helander and Karlsson, 2002).

3.2.2 Proposed monitoring programme

The standard aerial survey methodology used for grey seal surveys during the pupping season (Duck, 2002) is less suitable in the Baltic since pupping occurs on both land and ice. The proportion of seals pupping on land and on ice varies depending on ice conditions. Ice breeders are distributed over vast unstable areas which are difficult to survey. However, the standard method of monitoring numbers of pups born would be feasible in years when the ice coverage is limited and a majority of pups are born on land. In contrast to Atlantic populations, Baltic grey seals change land-breeding sites frequently. For estimates of pup production and mortality rates, ground counts involving assessment of moulting stages are necessary. Therefore, three methods are proposed for monitoring grey seals in the Baltic: 1) photo-identification to derive population estimates; 2) counts during the peak moulting season to study population trends; and 3) pup counts under conditions when a majority of pups are born on land.

Photo-identification should be carried out at least once every second year. To be useful, it is important that the project is coordinated and implemented on a permanent basis for the whole Baltic region. The ideal situation is that the work is performed by a small number of teams to ensure consistency in data collection. However, photo-identification is less suitable to detect changes in abundance quickly, since the process to derive the estimates is slow. Therefore, internationally coordinated counts during the peak moulting season are proposed to derive trends in abundance and to be able to detect changes in trends. It is also important that the survey design is coordinated internationally to ensure consistency in data collection, including environmental covariates that may affect the number of seals hauled out. A periodic international workshop is needed to ensure that the photo-identification methodology will be standardized and subsequently deployed in all countries involved.

3.3 Harbour seals

3.3.1 Currently used methodology

A minimum of three aerial surveys during the peak moulting season in the end of August are used for estimates of trends in harbour seals in the entire area of distribution (Härkönen *et al.*, 2002). At specific monitoring sites, ground counts of pups are conducted for estimating changes in reproductive capacity (Härkönen *et al.*, 2002).

3.3.2 Proposed monitoring programme

Power analysis has shown that it will take seven years to detect a 5% change in the annual rate of increase, when using three replicate flights. Thus, annual replicate surveys of seals in the entire area are more effective to detect changes in trends within a reasonable time. Monitoring carried out at long time intervals reduces the possibilities to evaluate the effects of catastrophic events. Furthermore, more frequent surveys increase the likelihood to detect annual variations in rates of increase. However, given a choice between replicated flights at longer intervals and single annual flights, ACE recommends the former.

3.4 Ringed seals

3.4.1 Currently used methodology

For estimates of the distribution and abundance of ringed seals on ice, a strip survey technique has been used as described by Härkönen and Heide-Jørgensen (1990) and Härkönen and Lunneryd (1992). In this method, strips are placed in a systematic manner to evenly cover the study area. The surveyed strips extended to 400 m to either side of the aircraft, flying at an altitude of 90 m. Observations of seals were noted at two-minute intervals, which permits the positioning of observations within segments (Härkönen and Lunneryd, 1992). When seal density is calculated for each segment, detailed mapping of ringed seal distribution is possible. The method has been implemented in all surveys in the Bothnian Bay from 1988–2002, and in 1996 also in surveys in the Gulf of Riga and the Gulf of Finland. In surveys before 1996, different or modified methods were used in the latter two areas due to logistic problems and changing ice conditions.

In earlier surveys, a low-winged single-engine aircraft was flown at an altitude of about 30 m over the ice where conditions were judged to be most suitable for seals. The distance and angle to each sighted seal were measured and recorded, and the strip width was calculated retrospectively based on these measurements (Helle, 1980). The length of the strip was calculated from the air speed of the aircraft.

Lair counts and monitoring of mortality are used to assess population size in Lake Saimaa. In Lake Ladoga there is no regular monitoring, but considering the biology of the species, both lair counts and aerial surveys have been developed for the region.

3.4.2 Proposed monitoring programme

It is proposed to use the strip survey technique according to Härkönen and Heide-Jørgensen (1990) and Härkönen and Lunneryd (1992) described above, since the results are less affected by subjective factors. The method is also suitable for preparing detailed ringed seal distribution maps.

For Lake Saimaa, continuation of annual lair counts and mortality monitoring is recommended, while in Lake Ladoga both ringed seal survey methods should be employed according to the alternative breeding behaviour of the seals.

3.5 Harbour porpoises

3.5.1 Currently used methodology

The standard method for estimating the abundance of cetaceans is the line-transect distance sampling methodology described in detail in Buckland *et al.* (1993). These surveys can be conducted using either aircraft or ships and need to include some calculation of correction factors (g(0)). A minimum number of sightings is needed to estimate the effective strip width. Therefore, a problem in using line-transect methodology in areas of very low density such as the Baltic Proper is the difficulty of having a sufficient number of sightings to obtain a reliable abundance estimate with an acceptable confidence interval.

3.5.2 New methodology

The development of new methods of analysing line-transect data, such as modelling used by Hedley *et al.* (1999), can be used to estimate population sizes. This can also be applied to old data which could be re-analysed.

Also, the development of additional methods for areas of low density is urgently needed. One approach is to use acoustic methods to detect porpoises. Towed hydrophone click detectors have been used in surveys in the Baltic Sea (Gillespie *et al.*, 2002), but no abundance estimates have been derived from these methods. Acoustics have been used together with visual surveys for sperm whales to estimate abundance (Barlow, 1999). The planned SCANS II survey for 2005 should incorporate the development and use of click detectors and provide data to calibrate this methodology with the line-transect distance sampling results.

The use of stationary hydrophones such as PODs (porpoise detectors) can be employed to monitor small areas. They are especially useful in areas with a low density of porpoises.

3.5.3 Proposed monitoring programme

Because of the different populations and their very different status within the larger Baltic Sea area, the two harbour porpoise populations are treated separately.

3.5.3.1 Harbour porpoises in the Kattegat/Skagerrak/Belt Sea area

To estimate the abundance of harbour porpoises in the Kattegat/Skagerrak/Belt Sea area, standard line-transect distance sampling, either by plane or by boat, should be used. A large-scale aerial and/or ship-based line-transect distance sampling survey should be conducted annually (ideally), or at least every three years. Detailed surveys will probably provide smaller confidence intervals for the population sizes. The more frequently a survey is conducted, the higher the probability of detecting changes in trends in population size.

3.5.3.2 Harbour porpoises in the Baltic Proper

Due to the low density of porpoises in the Baltic Proper, standard line-transect methodology is not applicable. Furthermore, very little information on the current distribution of porpoises in the Baltic Proper is available. Accordingly, ICES proposes that the step-wise process to obtain information on the abundance of harbour porpoises in the Baltic Proper, as described in "Recommendations and advice", above, be carried out.

3.6 Advice on harmonization and synchronization of methods

3.6.1 Seals

Workshops should be held annually to ensure the compatibility of methods and analyses of grey seal abundance. Training of field personnel should include evaluations of observer variability.

3.6.2 Harbour porpoises

Aerial and shipboard surveys using line-transect distance sampling methodology should be standardized to ensure comparable results.

To make the data collection and analyses from stationary click detectors comparable between countries, an international workshop should be conducted.

3.7 References

Barlow, J. 1999. Trackline detection probability for long diving whales. *In* Marine Mammal Survey and Assessment Methods, pp. 209–221. Ed. by G.W. Garner, S.C. Amstrup, J.L. Laake, B.F.J. Manly, L.L. McDonald, and D.G. Robertson. A.A. Balkema, Rotterdam, the Netherlands.

Buckland, S.T., Anderson, D.R., Burnham, K.P., and Laake, J. 1993. Distance Sampling: Estimating Abundance of Biological Populations. Chapman and Hall, London, UK.

CEC. 2002. Report of the Subgroup on Fishery and Environment (SFGEN), Scientific, Technical and Economic Committee for Fisheries (STECF), Incidental Catches of Small Cetaceans, Brussels, 10–14 December 2001. SEC (2002) 376. 83 pp.

Duck, C. 2002. Pup production in the British grey seal population. Annex II in Scientific advice on matters relating to British seal populations. 2002. Available through: http://smru.st-and.ac.uk.

Gillespie, D., Brown, S., Lewis, T., Matthews, J., McLanaghan, R., and Moscrop, A. 2002. Preliminary results on the relative abundance of harbour porpoises (*Phocoena phocoena*) from acoustic and visual surveys in German, Danish, Swedish and Polish waters during summer 2002. IFAW Preliminary Report (unpublished). Song of the Whale Research Team, International Fund for Animal Welfare, 87–90 Albert Embankment, London SE1 7UD, UK. 10 pp.

Harding, K., Härkönen, T., and Helander, B. 2003. Ecological risk analysis of the Baltic gray seal population (*Halichoerus grypus*). Working paper presented to the 2003 meeting of the Working Group on Marine Mammal Ecology.

Härkönen, T., Harding, K., and Heide-Jørgensen, M.-P. 2002. Rates of increase in age-structured populations: a lesson from the European harbour seals. Canadian Journal of Zoology, 80: 1498–1510.

Härkönen, T., and Heide-Jørgensen, M.-P. 1990. Density and distribution of the ringed seal in the Bothnian Bay. Holarctic Ecology, 13: 122–129.

Härkönen, T., and Lunneryd, S.G. 1992. Estimating abundance of ringed seals in the Bothnian Bay. Ambio, 21: 497–503.

Hedley, S.L., Buckland, S.T., and Borchers, D.L. 1999. Spatial modelling from line transect data. Journal of Cetacean Research and Management 1, 255: 64.

Helander, B., and Karlsson, O. 2002. Inventering av gråsäl vid svenska Östersjökusten 2001. Sälinformation, Naturhistoriska riksmuseet, 2002:1.

Helle, E. 1980. Lowered reproductive capacity in female ringed seals (*Pusa hispida*) in the Bothnian Bay, northern Baltic Sea, with special reference to uterine occlusions. Annals Zoologica Fennica, 17: 147–158.

Hiby, L., Lundberg, T., Karlsson, O., and Helander, B. 2001. An estimate of the size of the Baltic grey seal population based on photo-id data. Report to project "Seals and fisheries". Länstyrelsen i Västernorrlands Län and Naturvårdsverket. 29-11-2001.

Hiby, L., Lundberg, T., Karlsson, O., Watkins, J., and Helander, B. 2003. An estimate of the size of the Baltic grey seal population based on photo-id data. Working paper presented at the 2003 meeting of the Working Group on Marine Mammal Ecology.

Jüssi, I., and Jüssi, M. 2001. Action plan for grey seals in Estonia 2001–2005. Eesti ulukid/Estonian Game 7. 88 pp.

Request

The request from the European Commission, Directorate General for Fisheries, in February 2002 concerning by-catch of cetaceans states:

Develop further the basis for advice to the European Commission on cetacean by-catch and mitigation measures in EU Fisheries [EC DG FISH]

i) *Update information on by-catches of cetaceans by species, gear, and area.*
ii) *Update information on sizes and distribution of cetacean populations against which by-catches can be counted.*
iii) *Details of gears, areas, and times associated with effective closures. Potential advantages and disadvantages of a generalized use of pingers in fixed gear; technical specifications affecting the effectiveness of pingers.*
iv) *Potential advantages and disadvantages of a generalized use of pingers or other deterrents in pelagic trawls; updated information and technical specifications.*
v) *Technical details of any other possible mitigation measure.*

This request was previously considered by the 2002 ACE meeting, which prepared an extensive response and advice (ICES, 2002). Subsequently, information on cetacean by-catches in European waters was reviewed in 2003 by the Working Group on Marine Mammal Ecology (WGMME) and also by two earlier meetings of the Subgroup on Fisheries and Environment (SGFEN) of the Scientific, Technical and Economic Committee for Fisheries (STECF) (CEC, 2002a, 2002b). There has been rather limited new information available since the 2002 ACE meeting.

Source of information

The 2003 Report of the Working Group on Marine Mammal Ecology (WGMME) (ICES CM 2003/ ACE:03).

Summary

New information on small cetacean by-catches includes figures on: the by-catch of common dolphins in UK pelagic trawls for sea bass in the English Channel in 2002; harbour porpoises from by-catch in Polish waters reported over ten years; by-catch of dolphins in Spanish trawl fisheries in the Bay of Biscay in 1996–2001; by-catch of minke whales in trap lines in lobster/crab fisheries including three records in 2000 and one in 2001 for the UK and one for Spain in 2000. No by-catch was reported in 45 days of longline fishing in Sub-area VIII during 1998–2001; in a limited observation scheme for the Spanish Basque Country shelf set-net fisheries for 1998–2001; nor in an observer scheme for the Spanish tuna purse seine fisheries in the Central-Eastern Atlantic covering 3,113 days at sea.

Aerial surveys in May to August 2002 gave new information on the distribution of harbour porpoises in German North Sea and Baltic waters. In the North Sea, densities were highest in the northeastern part of the survey area, closest to the Danish border. In the Baltic Sea, the highest relative abundance of porpoises was found in the Pomeranian Bight between the island of Rügen and the Polish border.

The UK has recently launched a consultation document with a proposed national by-catch reduction strategy that will entail using pingers. In Sweden, pingers have been deployed in the salmon driftnet fishery in a feasibility study to examine their handling characteristics and any potential disruption of fishing activity.

Trials were conducted during the summer of 2002 in the Irish pelagic pair trawl fishery for albacore using acoustic deterrent devices. A paired trial using standard pingers in one net and a control net without pingers was inconclusive. A paired trial using a remotely triggered louder acoustic deterrent seemed promising, but further work is needed. Work on the development of a dolphin exclusion grid for use in the bass pelagic pair trawl fishery continues in the UK. Two trials have been conducted, in 2002 and 2003, largely aimed at addressing fish loss.

Recommendations and advice

Due to the limited amount of new information at this ACE meeting, there is no new advice given this year; however, ICES reiterates its 2002 recommendations (ICES, 2002) concerning cetacean by-catch.

In the 2002 ACE report, ACE referred to recommendations from the draft ASCOBANS recovery plan for harbour porpoise in the Baltic. No specific recommendation concerning the Baltic was, however, given by ACE. For clarification, ICES endorses the (now completed) ASCOBANS "Jastarnia Plan" for recovery of the harbour porpoise in the Baltic Sea.

Scientific background

4.1 Information on by-catch of cetaceans

4.1.1 Gillnets

Estimates for the by-catch of harbour porpoises (*Phocoena phocoena*) in Danish and UK North Sea gillnet fisheries are summarized in the 2002 ACE report (ICES, 2002). There are still no estimates of by-catch for any existing Norwegian gillnet fisheries, though there is now a limited marine mammal by-catch observer scheme on vessels larger than 21 m working north of 62 °N.

A one-year programme is under way in Germany to estimate by-catch in gillnet fisheries in the North Sea. Three vessels are currently taking part in this fishery. Only one vessel fished with gillnets during winter-early spring 2002–2003, primarily in the southern North Sea off the Dutch coast. No by-catch was observed. Since the sole season started in April, all three vessels have been engaged in sole fishing. Previous investigations in this fishery in 1996 have shown that no by-catch occurs in this fishery, probably due to the low net height.

In a study of the distribution of by-catch in Polish waters, Kuklik and Skóra (2003) reported that 45 dead harbour porpoises from by-catch were notified to Hel Marine Station over ten years, with nearly half of these from Puck Bay (Figure 4.1.1.1) and 40% in semi-driftnet fisheries for salmonids. A further third of the by-catch was in set-nets for cod (Table 4.1.1.1).

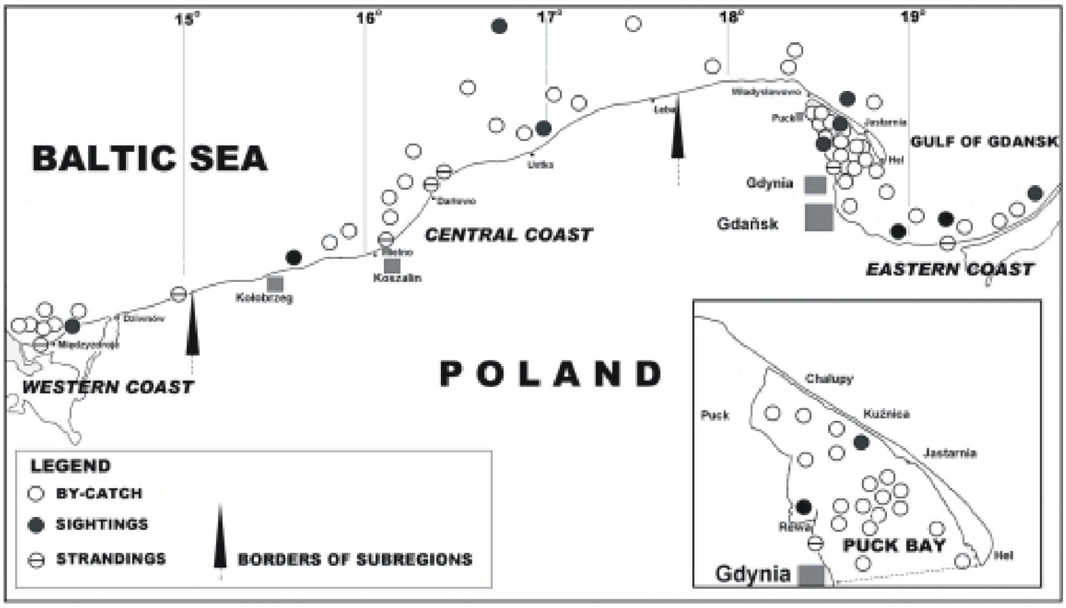


Figure 4.1.1.1. Places where harbour porpoises were sighted, by-caught, or washed ashore on the Polish coast between 1990 and 1999 (Kuklik and Skóra, 2003).

Table 4.1.1.1. By-catch of harbour porpoises in different types of fishing nets in 1990–1999 in Poland (Kuklik and Skóra, 2003).

Year	Total number of by-caught animals	Type of nets					
		Semi-driftnets (salmon)	Bottom-set gillnets		Herring gillnets	Herring trawl nets	Other set nets
			Cod	Others			
1990	1	1					
1991	7	3	1	2			1
1992	5		1	2			2
1993	7	4	1	2			
1994	3	1	1		1		
1995	5	4				1	
1996	10	4	5	1			
1997	2	1	1				
1998	3		3				
1999	2		2				
Total	45	18	15	7	1	1	3
%	100	40.0	33.3	15.5	2.2	2.2	6.8

Estimates of by-catch for Irish gillnet fisheries are limited to those for the now-terminated albacore driftnet fishery and those for the Irish hake gillnet fishery in the Celtic Sea. Tregenza and Collet (1998) provided an estimate of around 200 common dolphins (*Delphinus delphis*) taken annually in the Celtic Sea hake gillnet fishery (UK and Ireland). Current levels of this interaction are unknown. There are as yet no estimates of by-catch for UK and Irish gillnet and tangle net fisheries in the Celtic Sea other than for the hake fishery.

Information on Dutch by-catches is limited to inferences from strandings and was summarized by ACE in 2002. Only very few by-catches, especially from recreational beach set-net gillnet fisheries, are reported from Belgium each year (national reports to ASCOBANS). There are three Belgian gillnet vessels operating on a regular basis.

There are no estimates of total by-catch in gillnet fisheries for France, Spain or Portugal, though some observations have been made in all three countries. A rather limited set-net effort has been observed with no or very few by-caught cetaceans. Further observations have been made by AZTI (Fisheries and Food Technological Institute) in gillnet fisheries in the Spanish Basque Country. Observers on board commercial fishing vessels collected data in the period 1998–2001 in several surveys. These included discard and other survey types, during which marine mammal by-catch data were also collected. So far, only the number of dolphins has been recorded, and there has been no species identification. Observations have been carried out on board the artisanal gillnet Basque fleet (with tangle and trammel nets) working on depths less than 150 m, in the Basque Country shelf. Despite scattered sightings of dolphins made during some of the trips, no cetacean by-catches have been recorded in fourteen tangle net hauls between 1998 and 2001 and 34 trammel net hauls between 1999 and 2000.

4.1.2 Pelagic trawls

There are no reliable estimates of total by-catch for pelagic trawl fisheries, though observations have been made and by-catch rates have been established for several fisheries.

Kuklik and Skóra (2003) refer to a single record of a harbour porpoise by-caught in a herring trawl in Polish Baltic waters in 1995. There have been some limited discard observations in other pelagic trawl fisheries in the Baltic, but no records of porpoise by-catch are known.

Information on the by-catch of common dolphins, striped dolphins (*Stenella coeruleoalba*), white-sided dolphins (*Lagenorhynchus acutus*), and long-finned pilot whales (*Globicephala macrorhynchus*) in the Irish pelagic trawl fishery for albacore was summarized in the 2002 ACE report. A summary of other reported cetacean by-catches in pelagic trawls is given in the 2001 ACE report (ICES, 2001).

There is an ongoing observer programme in the UK to monitor cetacean by-catch rates in pelagic trawl fisheries. In 2001, 53 common dolphins were recorded from twelve of 116 observed tows in the pelagic pair trawl fishery for bass in the English Channel, while in 2002 there were eight common dolphins recorded in two hauls among 66 observed tows. There have been no observed cetacean by-catches in other UK pelagic trawl fisheries.

Spanish trawl fisheries in the Bay of Biscay (Divisions VIIIa,b,c,d) have been monitored by observers in several fishery-related surveys over the period 1997 to 2001. Observations have mostly been made in just three fisheries: the "Baka" bottom trawl, targeting mixed species (in Divisions VIIIa,b,d); the bottom pair trawl working with VHVO (very high vertical opening) nets, targeting hake in Divisions VIIIa,b,d; and the bottom pair trawl working with VHVO nets, targeting mainly blue whiting and other pelagic species in Division VIIIc along the edge of the shelf of the Spanish Basque Country. The number of hauls observed and numbers of dolphins recorded are summarized in Table 4.1.2.1.

No species identifications are available, but this information will be collected in future. The by-catches observed in these trawl fisheries show considerable variability between years and months. No clear patterns of by-catch have been identified, and thus these data have not been used to extrapolate to the total by-catch for the whole fleet. The species composition is also unknown, and as in many other places, there is a lack of information on the population size of any of the candidate species in the Bay of Biscay, making it difficult to assess whether or not the observed levels of by-catch are likely to represent a risk to any populations there.

It is worth recalling here the fact that hundreds of dead common and striped dolphins have been washed ashore in the early part of every year in southern England and the Biscay coast of France for more than fifteen years. In January of 2003, a total of 116 dead cetaceans (58% of them common dolphins) were recorded stranded in the southwest of England (Cornwall, Devon, and Dorset) (Ross, 2003). A further 36 cetaceans stranded in December 2002. In the first four months of 2002, 195 stranded cetaceans were recorded for that area. In the same period of 2001, 107 cetaceans were recorded (61% common dolphins). More than 300 cetaceans stranded along the Atlantic coast of France, south of Brittany, in a period of ten days (from about 20–30 January). By far the dominant species was common dolphin, with a few striped dolphins and very few harbour porpoises (Ross, 2003). Many of the stranded animals showed clear evidence of having died in fishing operations. Pelagic trawl fisheries have been blamed for these deaths, but the limited extent of observer coverage in the many fisheries operating in this region makes it difficult to attribute

Table 4.1.2.1. Observations of marine mammals by-caught in different bottom trawl metiers in the Spanish Basque Country trawl fleet in the period 1996–2001.

Year	Fishing gear	ICES Division	Days fishing	Hauls observed	Dolphin by-catch
1996	VHVO BT*	VIIIa,b,d	69	207	3
	VHVO BT	VIIIc	55	187	0
	Baka	VIIIa,b,d	8	48	0
1997	VHVO BT	VIIIa,b,d	28	43	5
	VHVO BT	VIIIc	4	5	0
	Baka	VIIIa,b,d	7	29	0
1998	VHVO BT	VIIIa,b,d	34	39	4
	Baka	VIIIa,b,d	106	488	0
	VHVO BT	VIIIc	17	52	0
1999	VHVO BT	VIIIa,b,d	61	128	12
	Baka	VIIIa,b,d	110	282	0
	VHVO BT	VIIIc	2	7	0
2000	VHVO BT	VIIIa,b,d	101	167	0
2001	VHVO BT	VII**	38	76	0

* Very High Vertical Opening Bottom Trawl.
** This ICES Sub-area is out of the Bay of Biscay.

these mortalities to specific fisheries with any degree of certainty. Clearly, if these mortalities are to be reduced, independent observations will first need to be made among all the fisheries operating in this area at this time of year to identify the source of these by-catch mortalities.

4.1.3 Other fisheries

AZTI has sampled longline and purse seine effort. Longlining was widely used in the past, but its use is now in decline in Spain. Observations were made in Divisions VIIIa,b,c,d on vessels targeting mainly hake between 1998 and 2001. A total of 45 days and 45 hauls were observed with no marine mammal by-catch recorded.

Spanish purse seine vessels targeting tuna in the Indian and Central-Eastern Atlantic Oceans have also been observed. In a programme funded by the ship owners, 63 observers were placed on board 63 purse seiners, 23 in the Central-Eastern Atlantic and 40 in the Indian Ocean in 1998 and 1999. There were 2,459 days at sea observed in the Indian Ocean and 3,113 days at sea observed in the Atlantic without any marine mammal by-catch.

There have been a few recorded incidents of minke whales being drowned in trap lines in lobster/crab fisheries. These were summarized by the IWC in 2001 (IWC, 2002) and include three records in 2000 and one in 2001 for the UK, and one for Spain in 2000.

4.2 Information on sizes and distribution of cetacean populations

Small cetacean abundance estimates have been summarized by ACE (ICES, 2002). Although abundance estimates have been made for the North Sea and part of the Baltic Sea and Celtic Sea, and for a few other small areas, it is clear that for most other parts of the ICES region and for most species, there are as yet no satisfactory abundance estimates.

Clearly, it is not possible to assess the impact of any by-catch without knowing anything about the abundance of the animals concerned. A planning committee is currently drawing up a proposal to repeat the SCANS survey over a wider area.

4.2.1 Harbour porpoise in the North Sea

Scheidat *et al.* (2003) conducted aerial surveys in May to August 2002 to examine the distribution of harbour porpoises in the German part of the North Sea. The highest densities were found in the northeastern part of the survey area, closest to the Danish border.

4.2.2 Harbour porpoise in the Baltic Sea

A ship-based line-transect survey of Polish coastal waters in 2001 saw only one harbour porpoise. An aerial survey of German and some southern Danish waters was undertaken from May to August 2002 (Figure 4.2.2.1).

This survey found the highest relative abundance of porpoises in the Pomeranian Bight between the island of Rügen and the Polish border (Scheidat *et al.*, 2003). The maximum group sizes in this area were ten animals. Repeated flights in August, September, December, February, and March in the same area did not find a single porpoise (M. Scheidat, pers. comm.). This demonstrated that the overall density of porpoises was lower between the island of Rügen and the Polish border than indicated through the surveys in May and July.

No information is available for assessing any trend in abundance. Two aerial surveys have been conducted in the Baltic Sea in 2002 from Germany and Sweden, but the data have not yet been published. Another large-scale abundance survey is planned for 2005.

4.3 Mitigation measures

4.3.1 Possible limitations on use of gear: time/area closures

There has been no new work on time/area closures with respect to limiting cetacean by-catch.

4.3.2 Use of pingers in gillnets

The use of pingers has been mandated by domestic Danish regulations since 2001 in North Sea cod wreck net fisheries between August and October. The UK has recently launched a consultation document with a proposed national by-catch reduction strategy that will entail using pingers on all nets of more than 220 mm (stretched mesh) in ICES Divisions IVb and IVc, as well as on offshore cod wreck nets, and on all gillnets and tangle nets fished outside of 6 nm in ICES Divisions VIIe,f,g,h, and j (DEFRA, 2003).

In Sweden, pingers have been deployed in the salmon driftnet fishery in a feasibility study to examine their handling characteristics and any potential disruption of fishing activity.

4.3.3 Acoustic deterrents in pelagic trawl fisheries

BIM (2003) conducted trials during the summer of 2002 in the Irish pelagic pair trawl fishery for albacore using acoustic deterrent devices. A paired trial using "Aquamark 2000" pingers in one trawl and a control

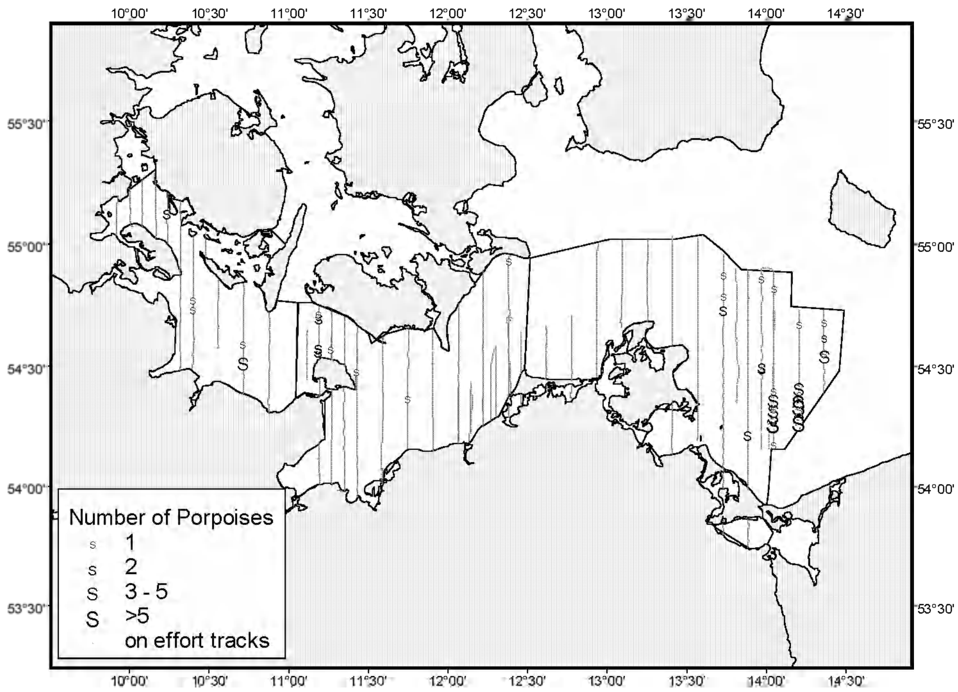


Figure 4.2.2.1. German aerial surveys in the Baltic (May to August 2002) (Scheidat *et al.*, 2003).

trawl without pingers was inconclusive. Another paired trial using a remotely triggered louder acoustic deterrent (prototype) seemed promising, but further work is needed.

4.3.4 Exclusion devices

Work on the development of a dolphin exclusion grid for use in the bass pelagic pair trawl fishery continues in the UK. Two trials have been conducted, in 2002 and 2003, largely aimed at addressing fish loss; the report of the most recent trial has not yet been completed.

References

BIM. 2003. Bord Iascaigh Mhara 2002 Annual Review Irish Sea Fisheries Board, Dun Laoghaire, Ireland. p. 16.

CEC. 2002a. Incidental catches of small cetaceans. Report of the meeting of the Subgroup on Fishery and the Environment (SGFEN) of the Scientific, Technical and Economic Committee for Fisheries (STECF), Brussels, 10–14 December 2001. SEC (2002) 376.

CEC. 2002b. Incidental catches of small cetaceans. Report of the second meeting of the Subgroup on Fishery and the Environment (SGFEN) of the Scientific, Technical and Economic Committee for Fisheries (STECF), Brussels, 11–14 June 2002. SEC (2002) 1134.

DEFRA. 2003. UK small cetacean bycatch response strategy – consultation document. Department for Environment, Food and Rural Affairs, Bristol.

ICES. 2001. Report of the ICES Advisory Committee on Ecosystems, 2001. ICES Cooperative Research Report, 249: 3–14.

ICES. 2002, Report of the ICES Advisory Committee on Ecosystems, 2002. ICES Cooperative Research Report, 254: 2–17.

IWC. 2002. Annex O: Summary of information from progress reports. Journal of Cetacean Research and Management, V. 4 Supplement, April 2002.

Kuklik, I., and Skóra, K.E. 2003 (in press). By-catch as a potential threat for harbour porpoise (*Phocoena phocoena* L.) in the Polish Baltic Waters. NAMMCO Scientific Publications.

Morizur, Y., Berrow, S.D., Tregenza, N.J.C., Couperus, A.S., and Pouvreau, S. 1999. Incidental catches of marine-mammals in pelagic trawl fisheries of the Northeast Atlantic. Fisheries Research, 41: 297–307.

Scheidat, M., Kock, K.-H., and Siebert, U. 2003. Summer distribution of harbour porpoise (*Phocoena phocoena*) in the German North and Baltic Sea. Working paper presented at the 2003 ASCOBANS meeting. 13 pp.

Ross, A. 2003 Cetacean by-catch in pelagic trawl fisheries in the Celtic Sea, Biscay, Channel area – a case for emergency action. Working paper presented at the 2003 ASCOBANS meeting. 10 pp.

Tregenza, N.J.C., and Collet. A. 1998. Common dolphin (*Delphinus delphis*) bycatch in pelagic trawl and other fisheries in the northeast Atlantic. Reports of the International Whaling Commission, 48: 453–459.

5 DRAFT OSPAR LIST OF THREATENED AND DECLINING SPECIES AND HABITATS

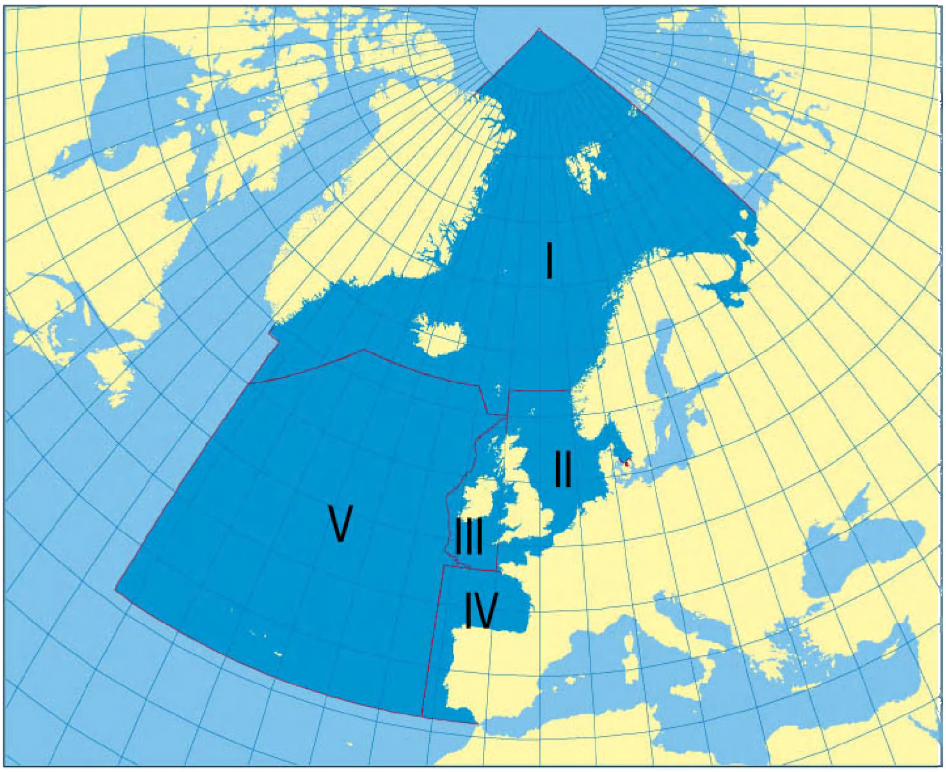
Request

In November 2001, the OSPAR Biodiversity Committee reviewed a draft list of threatened and declining species and habitats and agreed that this list should be further developed for approval by the Committee at its meeting in early 2003. This list must be supported by a justification of how and why the species and habitats were selected, and the Biodiversity Committee noted that Quality Assurance of the data used in identifying threatened species or habitats is very important. Hence, the OSPAR Commission requested ICES to contribute to the peer-review process.

The OSPAR request, transmitted in January 2002, was "*for the assessment by ICES by the early autumn 2002 of the data on which the justification of the OSPAR Priority List of Threatened and Declining Species will be based. The purpose of the assessment would be to ensure that the data used for producing the justification are sufficiently reliable and adequate to serve as a basis for conclusions that the species and habitats concerned can be identified, consistently with the Texel-Faial criteria, as requiring action in accordance with the OSPAR Strategy on the Protection and Conservation of the Ecosystems and Biological Diversity of the Maritime Area.*"

ICES provided advice (ICES, 2002) on most of the species and habitats on the draft list, but did not have sufficient time to comment on fourteen species on the draft list. This advice was reviewed by the OSPAR Biodiversity Committee in January 2003 alongside a set of papers providing considerable further evidence to support the list. This further evidence was, in many cases, based on the ICES advice. The OSPAR Biodiversity Committee asked that ICES should advise on the evidence concerning those species that had not previously been reviewed. This section provides this information and also updates advice from 2002 in the light of new evidence and to ensure consistency.

Figure 5.1. Map showing the OSPAR regions.



Region	Lead country / countries	Other participating countries
I – Arctic Waters	Norway	Denmark (including Faroe Islands) and Iceland
II – Greater North Sea	Netherlands	Belgium, Denmark, France, Germany, Norway, Sweden and UK
III – Celtic Seas	Ireland and UK	
IV – Bay of Biscay and Iberian Coast	France and Spain	Portugal
V – Wider Atlantic	Iceland and Portugal	Belgium, Denmark, France,Germany, Ireland, Netherlands, Norway, Spain, Sweden and UK

The work of the Regional Task Teams was coordinated by the Assessment Coordination Group under the Environmental Assessment and Monitoring Committee.

Source of information

The 2002 Report of the Study Group on Elasmobranch Fishes (SGEF) (ICES CM 2002/G:08).

The 2002 Report of the Benthos Ecology Working Group (BEWG) (ICES CM 2002/E:07).

The 2001 Report of the Working Group on the Biology and Assessment of Deep-Sea Fisheries Resources (WGDEEP) (ICES CM 2001/ACFM:23).

The 2002 Report of the Working Group on Seabird Ecology (WGSE) (ICES CM 2002/C:04).

The 2002 Report of the Working Group on Marine Habitat Mapping (WGMHM) (ICES CM 2002/E:05).

The 2002 Report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO) (ICES CM 2002/ACE:03).

The 2003 Report of the Working Group on Fish Ecology (WGFE) (ICES CM 2003/G:04).

Documents BDC 03/3/2a–h from the OSPAR Biodiversity Committee meeting of January 2003, and some of the references therein.

Summary

This section completes ICES advice on the evidence used to support the draft OSPAR Priority List of Threatened and Declining Species and Habitats. Partial advice on this topic was provided in 2002. The 2003 advice is summarized in Table 5.1, which also includes updated advice concerning the species and habitats considered in 2002. Certain habitats that were included in the 2002 advice had not been on the draft OSPAR list, and advice relating to them has now been removed from this table. Updates relate mostly to consistency of the advice, but in some cases new information has become available since 2002. Detailed information on the species included for the first time in this ICES advice is available at Annex 1.

Recommendations and advice

Table 5.1 summarizes the ICES advice as to the adequacy of the evidence on the existence of actual declines and threats to the species and habitats on the OSPAR list. It also comments on the spatial extent of the evidence. The scientific details of the evaluation in 2003 are contained in Annex 1. Details for species and habitats assessed in 2002 are reported in Annex 1 of the 2002 report of the ICES Advisory Committee on Ecosystems (ICES, 2002).

Scientific background

OSPAR is in the process of preparing a list of threatened and declining species and habitats, to contribute to the requirements of Annex V (on the protection and conservation of the ecosystems and biological diversity of the maritime area) of the OSPAR Convention. In parallel with the process of preparing and refining a set of robust selection criteria (the Texel-Faial criteria), the Contracting Parties to OSPAR were asked to submit proposals for species and habitats which they felt were already under threat or in decline, and which therefore needed immediate management action. The evaluation of these submissions and the preparation of this list were ultimately considered by a workshop in Leiden in September 2001.

The draft OSPAR Priority List of Threatened and Declining Species and Habitats currently contains 29 species and habitats identified as of concern across the whole of the OSPAR maritime area, and ten identified as of concern in one or more of the five OSPAR Regions. These regions are as follows: Region I—Arctic Waters; Region II—Greater North Sea; Region III—Celtic Seas; Region IV—Bay of Biscay and Iberian Waters; and Region V—Wider Atlantic (Figure 5.1).

In January 2002, OSPAR requested ICES to evaluate the data that were used to justify the inclusion of each species and habitat on the list. Due to the timing of the request and the lack of working group meetings relevant to several types of species, the evaluation of the information on status and threat for several species was not completed in 2002. In early 2003, ICES Working Groups examined the data that were used to justify the inclusion of the remaining fish, reptile, and marine mammal species on the list. These assessments have been reviewed and are included as Annex 1 to this report.

It must be emphasized that these Working Groups were only asked to assess the data used to produce the list of species and habitats submitted to OSPAR. ICES was not asked to provide comment on the suitability, or otherwise, of the criteria used to generate that list, nor of the list of species that are under consideration. However, ICES noted that the species-based listings adopted by OSPAR were not consistent with the stock-based units that ICES uses for i) the assessment of commercial fish stocks, and ii) the implementation of fish stock recovery plans when the abundance has declined below specified reference points. ICES was also not asked to provide comments or suggestions for mitigation measures which may be necessary if these species and habitats are finally selected for management action.

Wherever possible, the preparation of a list of species, such as now under consideration, should be based on extensive biogeographical information on the respective species. Good knowledge of the geographical distribution of the species, and of the areas where it is threatened or declining, and where it is not, is fundamental to sound decisions about the risk to various species. Also needed is a good long-term documentation of any decline; a conclusion on decline should not be based on limited information regarding spatial and temporal dynamics of the populations involved. Special care is required when drawing conclusions regarding a decline of a species in areas at the border of its geographical distribution range.

ICES notes that complete information may be lacking on some species that are nonetheless potential candidates for designation. This may be because they are rare and, therefore, both potentially vulnerable to threats and difficult to study. It may be appropriate to evaluate such species for potential designation despite the imperfect information base. However, in those cases it is even more important to ensure that the best scientific advice possible is used in the designation. All potential information sources must be examined thoroughly and objective standards of interpretation applied during the designation. It is also important that the weaknesses in the information used in coming to the decision regarding risk are communicated clearly along with the final designation.

Reference

ICES. 2002. Report of the ICES Advisory Committee on Ecosystems, 2002. ICES Cooperative Research Report, 254: 42–46, 89–126.

Table 5.1. Summary of the adequacy of the evidence for declines in the OSPAR area and threats to the species and habitats listed in the draft OSPAR Priority List of Threatened and Declining Species and Habitats. Species shown in bold type concern wholly new advice in 2003. Small changes to the 2002 ICES advice have been undertaken to take account of new information and to achieve greater consistency. Some habitats that had been included in the 2002 ICES advice, but were not present on the draft OSPAR list, have been removed.

Threatened and/or declining species and habitats	Adequacy of evidence of decline	Adequacy of evidence of threat	OSPAR regions where the species occurs	OSPAR regions where the species is under threat and/or in decline
INVERTEBRATES				
Ocean quahog (*Arctica islandica*)	Sound evidence of decline in (southern) Region II.	Sound evidence for impact by trawling.	I, II, III, IV	II
Barnacle (*Megabalanus azoricus*)	Limited evidence of decline.	Limited evidence of unregulated harvesting.	V	V
Dogwhelk (*Nucella lapillus*)	Sound evidence of decline in several OSPAR regions, and limited evidence of recent recovery.	Sound evidence for TBT leading to imposex.	All	I, II, III, IV
Flat oyster (*Ostrea edulis*)	Sound evidence of widespread decline.	Sound evidence for overexploitation and also for introduction of other (warm-water) races and other oyster species. Sound evidence that disease and severe winters caused at least part of decline.	II, III, IV	II, III, IV
Limpet (*Patella ulyssiponensis aspera*)	Sound evidence of decline, probably mostly due to overexploitation.	Sound evidence of continuing threats.	V	V
BIRDS				
Lesser black-backed gull, *fuscus* subspecies (*Larus fuscus fuscus*)	Sound evidence of decline, for largely unknown reasons.	Limited evidence of continuing threats.	I	I
Steller's eider (*Polystica stelleri*)	No evidence of decline in OSPAR area, but sound evidence outside OSPAR area.	Oil pollution has killed birds in recent past, but no evidence of population-level threat.	I	

Table 5.1. Continued.

Threatened and/or declining species and habitats	Adequacy of evidence of decline	Adequacy of evidence of threat	OSPAR regions where the species occurs	OSPAR regions where the species is under threat and/or in decline
BIRDS (continued)				
Little shearwater (*Puffinus assimilis baroli*)	Sound evidence of probable decline in past in OSPAR area, but presently stable. Declines continue immediately outside OSPAR area.	Sound evidence of continuing threats, mostly from further introductions of mammals.	V	
Roseate tern (*Sterna dougallii*)	Sound evidence of decline.	Sound evidence of persistent threats, some outside OSPAR area.	II, III, IV, V	II, III, IV, V
Guillemot, Iberian population (*Uria aalge ibericus*)	Iberian subspecies may not be a valid taxon, but the population on Iberia is either extinct or near extinction.	Sound evidence that threats (oil pollution, by-catch in nets), that probably caused decline, persist.	IV	IV
FISH				
Sturgeon (*Acipenser sturio*)	Sound evidence of decline and local extirpation. Only one remaining spawning population (Region IV).	Sound evidence of persistent threats.	I, II, III, IV	I, II, III, IV
Allis shad (*Alosa alosa*)	Sound evidence of decline. No remaining self-sustaining populations outside OSPAR Regions II, III, and IV.	Sound evidence of persistent threats.	II, III, IV	II, III, IV
Basking shark (*Cetorhinus maximus*)	Sound evidence suggests widespread declines in the past.	Past declines caused by directed fishing. Limited evidence for current threats, but by-catch in fisheries is documented. Vulnerability is well documented.	All	All
Houting (*Coregonus lavaretus oxyrhinchus*)	Sound evidence of decline.	Sound evidence of persistent threats from changes in river habitats.	II	II
Common skate (*Dipturus* (*Raja*) *batis*)	Sound evidence of widespread decline.	Sound evidence of continuing threats from directed fisheries and by-catches.	All	All
Spotted ray (*Dipturus* (*Raja*) *montagui*)	Sound evidence of declines in southern and eastern North Sea, but poor evidence of decline in the western North Sea.	Sound evidence of threat from by-catch.	II, III, IV, V	II
Cod (*Gadus morhua*)	Sound evidence of declines in all ICES areas.	Management plans in place or being developed for all stocks, and recovery plans in place or being developed for stocks showing greatest decline.	All	All

Table 5.1. Continued.

Threatened and/or declining species and habitats	Adequacy of evidence of decline	Adequacy of evidence of threat	OSPAR regions where the species occurs	OSPAR regions where the species is under threat and/or in decline
FISH (continued)				
Couch's goby (*Gobius couchi*)	No evidence for decline; species either little known or naturally rare.	No evidence of significant threats.	II, III, IV	
Short-snouted seahorse (*Hippocampus hippocampus*)	No evidence for decline, though extent of seagrass habitat has decreased.	Sound evidence of threat to seagrass habitats, otherwise no evidence of threats to this species of seahorse. Vulnerability is well-documented for the genus.	II, III, IV	
Long-snouted seahorse (*Hippocampus guttulatus* formerly *H. ramulosus*)	No evidence for decline, though extent of seagrass habitat has decreased.	Sound evidence of threat to seagrass habitats, otherwise no evidence of threats to this species of seahorse. Vulnerability is well-documented for the genus.	II, III, IV	
Orange roughy (*Hoplostethus atlanticus*)	Sound evidence of severe decline in Region V, limited evidence of decline elsewhere.	Sound evidence of continuing threat from fisheries.	I, V	I, V
Sea lamprey (*Petromyzon marinus*)	Sound evidence of decline in Regions II and III, limited in other parts of range.	Sound evidence of continuing threat (from changes in river habitats and water quality).	I, II, III, IV	I, II, III, IV
Salmon (*Salmo salar*)	Sound evidence of decline in all regions. Much more serious declines in southern regions (France, Ireland, UK) than in northerly regions, where recent trends are upward.	Sound evidence of continuing low survival in marine phase.	I, II, III, IV	I, II, III, IV
Bluefin tuna (*Thunnus thynnus*)	ICCAT should be used as the primary source of advice on the status and trends of bluefin tuna, and of threats. ICES could review ICCAT information and advise in the context of consistent application of the Texel-Faial criteria.			
REPTILES				
Loggerhead turtle (*Caretta caretta*)	Sound evidence of large, widespread declines.	Sound evidence of continuing threats from by-catches in fishing gear, ship collisions, and marine debris.	IV, V	IV, V
Leatherback turtle (*Dermochelys coriacea*)	Sound evidence of large, widespread declines.	Sound evidence of continuing threats from by-catches in fishing gear, ship collisions, and marine debris.	All	All

Table 5.1. Continued.

Threatened and/or declining species and habitats	Adequacy of evidence of decline	Adequacy of evidence of threat	OSPAR regions where the species occurs	OSPAR regions where the species is under threat and/or in decline
MAMMALS				
Bowhead whale (*Balaena mysticetus*)	Sound evidence of large declines in past whaling era. Less than 100 individuals left in Spitzbergen stock.	No evidence of current direct threat, but indirect threats such as pollutant effects may be present.	I	I
Blue whale (*Balaenoptera musculus*)	Sound evidence of large declines in past whaling era; thought now to be recovering in North Atlantic, but remains depleted.	Limited evidence of insignificant direct threat in OSPAR area.	All, but Region II peripheral to natural range.	All
Northern right whale (*Eubalaena glacialis*)	Sound evidence of near extinction following hunting in whaling era; only 2–3 individuals sighted in OSPAR area in the past 20 years.	Threats impossible to evaluate in OSPAR area, but in western Atlantic ship strikes and entanglement in fishing gear continue to threaten possibly the same stock.	All, but Region II peripheral to natural range.	All
Harbour porpoise (*Phocoena phocoena*)	Sound evidence of declines in past in Channel/southern North Sea (and more recently in Baltic). Trends in other areas unknown.	Sound evidence that main threat is by-catch, particularly in bottom-set gillnets. By-catch likely to be unsustain-able on Celtic shelf, (Baltic), and probably parts of the North Sea.	Occurs in all regions, but the core of the range is Regions II and III. Population structure in OSPAR area is complex.	II, III
HABITATS				
Carbonate mounds	No clear evidence of declines of the mounds themselves.	Sound evidence of threat to mound biota from orange roughy fishery; no clear evidence of threats to the mounds themselves.	I, V	I, V
Deep-sea sponge aggregations	Sound evidence of decline in Region I, limited evidence elsewhere.	Limited evidence of continued threat from towed bottom gears and other physical disturbances to the sea floor.	I, III, IV, V	I, III, IV, V
Estuarine intertidal mudflats	Sound evidence of declines.	Sound evidence of continuing threats, e.g., land claim and coastal defenses.	I, II, III, IV	I, II, III, IV
Littoral chalk communities	Sound evidence of declines.	Sound evidence of threats from, e.g., coastal defenses.	II	II
Lophelia pertusa reefs	Sound evidence of declines.	Sound evidence of continuing threats from towed bottom gear.	All	All
Oceanic ridges with hydrothermal effects	No evidence of decline.	No evidence of current threats.	I, V	

Table 5.1. Continued.

Threatened and/or declining species and habitats	Adequacy of evidence of decline	Adequacy of evidence of threat	OSPAR regions where the species occurs	OSPAR regions where the species is under threat and/or in decline
HABITATS (continued)				
Ostrea edulis beds	Sound evidence of decline.	Sound evidence of continuing threats from overexploitation, introduction of other (warm-water) races and other oyster species.	II, III, IV	II, III, IV
Seamounts	No evidence of declines in habitat, but limited evidence of declines in associated biota.	Likely threats exist from towed bottom fishing gear, but no evidence of actual impact on habitat features.	I, IV, V	
Sublittoral mud with seapens and burrowing megafauna	Poor evidence of decline.	Sound evidence of threat by towed bottom gears causing physical disturbances to the sea floor across whole region.	I, II, III, IV	II, III
Zostera beds	Sound evidence of decline in Regions II and III.	Sound evidence of threats, e.g., nutrient inputs and turbidity in Regions II and III.	I, II, III, IV	II, III

6 ECOLOGICAL QUALITY OBJECTIVES

6.1 Overall consideration of the approach to and framework for the response to the OSPAR requests on ecological quality objectives

Request

Item 3 of the 2003 Work Programme from the OSPAR Commission is quoted below. The Work Programme refers to the EcoQs or EcoQOs using Roman numerals, while the text of the advice employs the letter designations for the EcoQ elements given in the Bergen Declaration. For clarity, these letters have been inserted in square brackets after the numerals in the request.

3. *Further work to develop Ecological Quality Objectives*

This activity relates to the issues, Ecological Quality Elements and Ecological Quality Objectives agreed in the Ministerial Declaration of the Fifth International Conference on the Protection of the North Sea (the Bergen Declaration).

3.1 *For the EcoQOs relating to (i)* [(a)] *spawning stock biomass of North Sea commercial fish species, (ii)* [(c)] *seal population trends in the North Sea, (iii)* [(e)] *by-catch in the North Sea of harbour porpoises, and (iv)* [(f)] *proportion of oiled common guillemots among those found dead or dying on beaches:*

- a. *develop draft guidelines (taking into account MON 01/9/1, Annex 6), including monitoring protocols and assessment methods, for evaluating the status of, and compliance with, those EcoQOs;*
- b. *for EcoQOs (i) and (iii), propose a list of species to be covered by these EcoQOs; and*
- c. *for all four EcoQOs, provide reference points (for commercial fish stocks) or current levels (for other species or populations), on an appropriate geographical basis, to be used as baselines against which progress can be measured;*
- d. *reconstruct the historic trajectory of these metrics and determine their historic performance (hit, miss or false alarm) relative to the objective being measured, as a basis for deciding their relationship to management; and*
- e. *provide advice on what management measures could be taken to help meet the EcoQOs.*

3.2 *Commence development, on the basis of the criteria for sound EcoQOs established by ICES in 2001, of related metrics, objectives, and reference levels for the EcoQOs relating to (v)* [(d)] *utilisation of North Sea breeding sites of seals, (vi)* [(g)] *mercury concentrations in eggs and feathers of North Sea seabirds, (vii)* [(h)] *organochlorine concentrations in seabird eggs, (viii)* [(i)] *plastic particles in the stomachs of North Sea seabirds, (ix)* [(j)] *local availability in the North Sea of sandeels for black-legged kittiwakes, (x)* [(k)] *seabird population trends in the North Sea as an index of seabird community health, (xi)* [(l)] *changes in the proportion of large fish and hence the average weight and average maximum length of the fish community, (xii)* [(o)] *density of sensitive (e.g., fragile) species, (xiii)* [(p)] *density of opportunistic species, and (xiv)* [(b)] *presence and extent of threatened and declining species in the North Sea.*

- a. *for EcoQ element (xi), develop draft guidelines (taking into account MON 01/9/1, Annex 6), including monitoring protocols and assessment methods, for evaluating the status of, and compliance with, those EcoQOs;*
- b. *for EcoQ elements (xii) and (xiii), identify possible species in the respective categories, consider further the spatial scale requirements of sampling and the adequacy of existing monitoring activities to determine their status and trends, and provide further advice based on scenario considerations on the applications of possible EcoQOs;*
- c. *for EcoQ element (xiv), consider the invertebrate and fish species and the habitats on the draft OSPAR list of threatened and declining species for their relevance and usefulness as a basis for EcoQOs for the North Sea; and*
- d. *where possible and appropriate for EcoQ elements (xi), (xii), (xiii) and (xiv), reconstruct the historic trajectory of the metrics and determine their historic performance (hit, miss or false alarm) relative to the objective being measured, as a basis for deciding their relationship to management.*

Source of information

The 2003 Report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO) (ICES CM 2003/ ACE:05).

The 2003 Report of the Working Group on Marine Mammal Ecology (WGMME) (ICES CM 2003/ ACE:03).

The 2003 Report of the Working Group on Seabird Ecology (WGSE) (ICES CM 2003/C:03).

The 2003 Report of the Working Group on Fish Ecology (WGFE) (ICES CM 2003/G:04).

2003–2005 Work Programme for the North Sea Pilot Project on Ecological Quality Objectives: General Framework. Summary Record of the OSPAR Biodiversity Committee Meeting, Dublin, 20–24 January 2003, Annex 4.

Summary

This section considers some general aspects of the Ecological Quality–Ecological Quality Objective (EcoQ–EcoQO) framework and then provides the information needed to address the request for advice on specific EcoQ elements from OSPAR.

In preparation for the review of the pilot project on EcoQOs in the North Sea, a framework for reporting on the consideration of the proposed EcoQOs has been produced. This framework is codified in a template that will be employed in 2004 by ICES Working Groups tasked with this work. This will include consideration of the reference level and historic trajectory, the robustness and ecological basis of the metric, monitoring programmes and their power and cost, and the geographical bounds over which the EcoQO is relevant.

The consideration of the individual EcoQ elements, summarized below, highlights the diversity in the degree of scientific support available for them. In response to the continuing request for advice from OSPAR, ICES will, using the standardized format provided by the template, consolidate the scientific background for each of the EcoQ elements in 2004. This will highlight the areas of scientific uncertainty and the data gaps. The EcoQ–EcoQO framework is explicitly an objective-based management system based on science. For many sectors of environmental management, this is a major cultural shift. Quantitative targets now supplement vague aspirations, such as "a healthy ecosystem", as management goals. There is a major science requirement here and the information below shows the heterogeneity in our current ability to deliver this. There is also a need for the review of the EcoQ–EcoQO framework to address the management implications of attempting to simultaneously meet all these targets.

A brief summary of the consideration of each EcoQ element is contained below, beginning with the four EcoQ elements, out of the ten selected for the North Sea Pilot Project, for which ICES has been requested to provide further development and advice in 2003. This is followed by summaries of the consideration of the other EcoQs, not part of the Pilot Project, for which ICES has been requested to commence development. The detailed work on each of these EcoQs/EcoQOs is contained in the sub-sections thereafter.

EcoQ element (a) Spawning stock biomass of commercial fish species

For EcoQ element (a) the spawning stock biomass of commercial fish species, the EcoQO is to be "above precautionary reference points for commercial fish species where these have been agreed to by the competent authority for fisheries management". This EcoQO was evaluated formally using the approach of signal-detection and decision theory. The context for the evaluation was focused very narrowly on the use made of B_{pa} (the biomass (B) below which the stock would be regarded as potentially depleted or overfished) and F_{pa} (the fishing mortality (F) designed to ensure with high probability that recruitment overfishing does not take place) in single years. In that narrow context, a perfect EcoQO will function as a reliable signal in the context of signal detection and decision theory. It is stressed that this context is **not** the same as a conventional fisheries management context. To analyse the performance of B_{pa} and F_{pa} as signals in a fisheries management context would have to take account of the fact that assessments are conducted and advice is provided annually. Performance of a signal in each year is affected by the performance of the same signal in the preceding year(s). It is stressed as well that an evaluation of the performance of Spawning Stock Biomass (SSB) and F as *signals* in a fisheries management context is only one part of analysing the performance of the scientific advisory and management system as a whole. Analysing the performance of the overall management system is the real test of how well conservation and sustainable use are being achieved. Such a performance analysis should be based on an analysis of management as a control system working over time with feedback mechanisms. The analyses here were conducted using all stocks in the OSPAR area for which ICES conducts analytical assessments. Three criteria can be used to determine whether a stock is within these limits and, hence, the EcoQO was met:

- Estimate of SSB was above the Precautionary Reference Point ($SSB>B_{pa}$);
- Estimate of F was below the Precautionary Reference Point ($F<F_{pa}$);
- Both of the above ($SSB>B_{pa}$ and $F<F_{pa}$).

The standard for determining the rates of hits (true positives and true negatives), misses (not advising a reduction in F when it was necessary), and false alarms (advising an unnecessary reduction in fishing) was to compare the conclusion about stock status in the assessment year with the conclusion that should have been made that year, given the *current* estimate of stock status in that year, from a converged Virtual Population Analysis (VPA). The analyses demonstrated that using F alone will result in relatively low False Alarm rates but high Miss rates. Using SSB alone results in a strong decrease of Miss rates together with a markedly higher proportion of False Alarm rates. The best results were achieved using both criteria, with a 53% Hit rate, 23% Miss rate, and 24% False Alarm rate. This suggests that the current framework is sound, and implies that management is keeping stocks very near the boundary condition for safe biological limits. Analysis of the quantitative response in changes in Total Allowable Catch (TAC) to the advice also showed in general that the advice was appropriate. Historic trajectories of the performance of management advice showed no clear trends over time in Hits when based on SSB only, F only, or SSB and F together. However, Misses were particularly common in the mid-1990s when based on SSB alone or SSB and F together. When based on F alone, Misses appeared to decrease from the mid-1990s onward. These analyses indicate that the EcoQO should be based on the proportion of stocks where $SSB>B_{pa}$, **and** where $F < F_{pa}$., considered together. The evaluation of B_{pa} and F_{pa} as EcoQOs indicates that Misses and False Alarms are about equally frequent. This suggests that advice based on SSB and F will not recommend catch reductions when in fact they are needed for about one stock in five, although the Miss rate seems to be going down in recent years. However, advice based on SSB and F relative to their reference points recommends unnecessary catch reductions about equally often. This symmetry in error rate *de facto* treats both types of errors (Misses and False Alarms) as equally undesirable. An effective way to reduce the error rate would be for management to try to maintain stocks at biomasses somewhat above B_{pa} and fishing mortality somewhat below F_{pa}. This was always the intent of ICES, as it repeatedly stresses in its advice that the precautionary reference points should be treated as boundaries on SSB and F, rather than as targets.

EcoQ element (c) Seal population trends in the North Sea

With respect to EcoQ element (c) seal population trends in the North Sea, ICES notes that the initial estimates are that approximately 30–50% of the populations of harbour seals in European waters perished during the 2002 Phocine Distemper Virus epizootic. As not all surveys after the 2002 epizootic have been completed, population estimates for harbour seals in 2002 are not available. Time series of abundances from which historic trends can be derived are only available for grey seals and harbour seals for parts of their North Sea distribution. This is a weakness of the EcoQO. For the populations investigated, in most cases the changes in population estimates of more than 10% between years were only detected one year at a time and are therefore considered to be false alarms. An exception is the decline caused by the Phocine Distemper Virus epizootic of 1988. This was detected in populations and generated appropriate management responses (e.g., research into the causes of the epizootic was conducted). The availability of data for assessment on whether the EcoQO is being met is acceptable. The number of births is a sensitive parameter responding more rapidly than total population size to changes in habitat conditions such as food availability. Pup/adult ratio is probably an indicator that will rapidly identify impaired reproduction in harbour seal populations where populations are surveyed during breeding and moulting seasons. ICES does not recommend a specific protocol for the monitoring of seal populations. Phocid seal survey methods, data collection, and analytical techniques vary between regions. It is recognized that compilation of these procedures into a single document will provide a valuable reference on seal census techniques. The management strategies for marine mammals applied by most countries in the OSPAR area are oriented towards maintaining or increasing marine mammal populations, so current management strategies are generally appropriate. A “hit” for this EcoQO triggers further research. The history of the effect of phocine distemper virus on harbour seal populations in European waters suggests that substantial reductions in seal numbers within the space of several months will trigger research in most countries. However, the comprehensiveness of these research programmes varies substantially between countries, from no research at all to detailed studies. It is certain that, despite signing the Bergen Declaration prior to the seal epizootic of 2002, no country initiated research on the basis that the EcoQO was triggered.

EcoQ element (e) By-catch of harbour porpoises

With respect to EcoQ element (e) the by-catch of harbour porpoises, in the Danish North Sea fisheries alone, the extrapolated annual by-catch was about 4,000 individuals in 1999–2001. In the recent past, this figure has been more than 7,000 per year. In addition to this, UK fisheries took in the order of 800 individuals in 1995 and 440 individuals in 1999. Total fishery by-catch cannot be evaluated because other fisheries (in particular Norwegian fisheries) are not yet monitored for by-catch. The above decline since 1994 in by-catch levels of Danish and UK fisheries was as a result of reduced fishing efforts. It is likely that this trend will continue with the major quota and effort reductions for the cod fisheries. There is only one estimate of harbour porpoise abundance in the North Sea. This was made in 1994 under the SCANS project. The SCANS survey covered most of OSPAR Region II and estimated a harbour porpoise population of 300,000 ($CV \approx 0.14$) for this area. The abundance estimate during SCANS only gave an idea of the summer distribution. There are clearly migrations and it is dangerous to base possible management measures to reduce by-catch on the harbour porpoise distribution observed during the SCANS I (and the future SCANS II) survey. Therefore, data on seasonal distribution-abundance also need to be taken into account, together with the data on abundance from the SCANS surveys. The population structure of the harbour porpoise in the North Sea is not well known, however, there is likely to be some structuring. Studies are at present inconclusive and it is likely that any "population boundaries" that exist will not be fixed in space or time. In the absence of clear population boundaries, a practical approach has to be taken with respect to the geographical basis for evaluating progress. Such areas could be aggregated SCANS sampling sub-areas. The number of sub-areas should, however, be kept at a low number, e.g., three or four, to minimize sampling variance of fisheries effort, by-catch, and population estimates. The only method to acquire reliable data for the majority of the fishing fleets is through the use of independent on-board observers. A major issue is that of scaling from on-board observations to fleet scale. In the North Sea, reasonable estimates exist for most fisheries relevant to harbour porpoise by-catch, with the notable exception of any way of estimating by-catch within the Norwegian small-boat fisheries. The historic trends in by-catch are unknown, and cannot be reconstructed. In theory, estimates might be made on the basis of fishing effort, but such information is also not available except in years from 1990 onwards. It would not be possible to assess by-catch rates as there is no information at present on trends in the abundance of harbour porpoises. An evaluation of the historic performance of this EcoQO therefore does not seem possible.

EcoQ element (f) Proportion of oiled common guillemots among those found dead or dying on beaches

For EcoQ element (f), there are considerable quantities of existing data on the proportion of oiled seabirds beached around the North Sea. In those areas where these data have been analysed, there is evidence of a decline in the proportion oiled. An analysis of this evidence indicates that it would not be possible to ensure that a target (the EcoQO is 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) has been met without several years of data. As a consequence, ICES recommends that a period of at least five years over which an average of 10% oiled common guillemots has been recorded should occur before the conclusion that the objective has been reached could be justified statistically. Guidelines for monitoring oiled birds have been developed both within OSPAR and in the Trilateral Monitoring Programme for the Wadden Sea. These guidelines are suitable for use in the whole North Sea. Resources (personnel, funds) will be required to ensure that sufficient sampling and coordination occurs at both the national and the international level. Possible management measures to reduce levels of at-sea oiling and a case study of one instance of successful management are described.

EcoQ element (d) Utilization of seal breeding sites in the North Sea

In relation to EcoQ element (d) utilization of seal breeding sites, fidelity for natal sites has been documented in harbour and grey seals. Abandoning of a breeding site could be considered as an indicator of habitat degradation and/or the start of a depletion in population. Utilization of a breeding site can therefore be considered as an indicator for ecological quality. Long-term data on the utilization of breeding sites around the North Sea are available. These data should be analysed in order to further assess the usefulness of this EcoQO, and to further develop it. The analysis, which should include a definition of appropriate spatial and temporal scales for this EcoQO, should be undertaken on the initiative of the lead country on this subject within OSPAR (the UK). The results of this analysis should be evaluated by ICES and, if appropriate, action plans for implementation after the triggering of the EcoQO should be developed.

EcoQ element (g) Mercury concentrations in seabird eggs and feathers

With respect to EcoQ element (g) mercury concentrations in eggs and feathers of North Sea seabirds, mercury contamination in the environment tends to be predominantly anthropogenic. Feathers from seabirds show different levels of mercury concentration, depending on the level of mercury in their prey items. Current levels of mercury in the feathers of certain seabird species are well known and they can be easily monitored. Feathers from birds collected today have been shown to contain up to four times more mercury than those from birds collected over 100 years ago. The level of mercury in birds' feathers thus provides a useful tool for measuring trends of mercury in the environment. Measuring the level of mercury in seabird feathers can be useful as an assessment of the results of management activities aimed at reducing mercury contamination. The proposed reference level for this EcoQO is the level measured in feathers from birds of selected species collected over 100 years ago, from places with a suspected low mercury contamination. For eggs, such a reference level is not possible to establish, given the different level of mercury in eggs from that in feathers and the lack of reference material. Mercury levels in eggs can, however, also provide useful information on the trends of mercury in the environment. The species for which the mercury concentrations in feathers should be measured are common tern, black-legged kittiwake, common guillemot, and northern gannet, from colonies in the southern and in the northern North Sea. The performance of the metric should aim at a downward direction. Setting the reference level as an objective is not realistic, given the past and present inputs of mercury into the environment. A more realistic objective for evaluating the consequences of effective management of mercury, which would be consistent with current scientific information on levels and trends of mercury in the environment, would be 1.5 times the measured reference level.

EcoQ element (h) Organochlorine concentrations in seabird eggs

In relation to EcoQ element (h) organochlorine concentrations in the eggs of North Sea seabirds, levels of organochlorines in seabird eggs show an immediate response to changes in the level of pollution of the marine environment, and can therefore be used as an EcoQ metric. Current programmes clearly indicate spatial and temporal trends in the level of organochlorines in the eggs of different species of seabirds. Given the fact that organochlorines are man-made substances, the performance of the metric should be a downward trend. However, due to the fact that these substances have long half-lives, it is not realistic to set the objective at zero. More realistic objectives for evaluating the consequences of effective management of these organochlorines, which would be consistent with current scientific information on the levels and trends in these substances for the oystercatcher and the common tern, would be <20 ng total PCBs g^{-1} egg fresh mass, <10 ng DDT and metabolites g^{-1} egg fresh mass, <2 ng HCB g^{-1} egg fresh mass, and <2 ng HCH g^{-1} egg fresh mass. Other species, which are proposed as useful species to monitor the organochlorine content of their eggs, are the common eider, northern gannet, and common guillemot.

EcoQ element (i) Plastic particles in stomachs of seabirds

With respect to EcoQ element (i) plastic particles in the stomachs of North Sea seabirds, plastic particles in the stomachs of beach-washed northern fulmars offer a reliable monitoring tool for changes in the level of plastic particle pollution at sea, and the number of these particles has been suggested as an EcoQ metric. Current studies in the Netherlands give a value of around 60% of fulmars having ten or more plastic particles in the stomach. The performance of this metric should be a downward trend. ICES can support the proposed objective of less than 2% of fulmars, out of a sample of fifty or more, having ten or more particles in the stomach. ICES advises also to gather data on the nature of the particles in the stomach, given indications that different types of litter (industrial plastic particles, user plastic particles) show different trends. The "chemical material" found in many fulmar stomachs should receive further analysis to determine its nature, likely origins, and toxic hazard.

EcoQ element (j) Local sandeel availability to black-legged kittiwakes

For EcoQ element (j) local availability in the North Sea of sandeels for black-legged kittiwakes, ICES has shown that the response of black-legged kittiwake breeding success to local sandeel availability is complex and non-linear. When sandeels are relatively abundant, changes in local sandeel availability are primarily driven by the environment. When sandeels are scarce, changes in availability may also be driven by fishing impacts on the sandeel stock. New research on the links between sandeel availability and black-legged kittiwake breeding success was presented at the 2003 meeting of WGECO. Scientists in WGECO and WGSE have not yet assimilated the new research findings with those on which past advice has been based. Before new advice on this EcoQO is formulated, ICES will continue to examine the relationship between black-legged kittiwake breeding success, sandeel availability, and sandeel fishing. The outcome will be included in the 2004 advice on this topic.

EcoQ element (k) Seabird population trends as an index of seabird community health

With respect to EcoQ element (k) seabird population trends in the North Sea as an index of seabird community health, healthy seabird communities in the North Sea are characterized by significant population changes within limits set by natural factors. Important changes could be indicating changes in the environment, possibly induced by human activities. A metric for ecological quality is change in breeding numbers of seabirds of selected species at selected key colonies. An objective that has been proposed for this metric is ≤ 20% decline over ≥ 20 years. ICES could agree, for the time being, to use such an objective, but it is clear that the EcoQO is to be further developed for individual species in order to indicate declines to a population level of a particular species, or even at colony levels. The objective should not be considered as a target to strive towards, but as a level where possibly impacts of human activities become apparent, and therefore there is a need for further research. It is expected that abundant and relatively widespread species such as the black-legged kittiwake and gannet (pelagic surface-feeding), common guillemot (pelagic pursuit-diving), common tern (coastal surface-feeding), and common eider (nearshore benthos-feeding) might be particularly useful as primary targets for assessing this EcoQO. The reference levels are variable, and largely of unknown magnitude. The expectation is that the population trends would not continue for many decades.

EcoQ element (l) Changes in the proportion of large fish and hence the average weight and average maximum length of the fish community

With respect to EcoQ element (l) changes in the proportion of large fish, a relation between the proposed metrics and fishing activity has been established. However, the probability of detecting short-term changes in the metrics is low, which makes it difficult to establish EcoQOs on which managers can act. More importantly, the available evidence suggests that the response time of the metrics to changes in fishing effort is considerable, particularly during the process of recovery. Also, the existing time series are generally too short to establish meaningful reference levels. Only one survey (the

Scottish August Demersal Survey) goes back to the early years of the 20th century, but this survey has been terminated recently and comparable information will not be available in the future. A more fundamental problem is that the values of the metric are highly dependent on the gear used, on the area where the gear is applied, and on the selection of species that is used in the analysis. Thus, if an EcoQO is set for this metric of ecological quality, it will apply to very specific conditions, and no EcoQO could therefore be expected to represent the total community under pressure. Finally, the proposed metrics have unwanted properties, because the community might be manipulated in such a way that the metric might show an apparent improvement, whereas in fact ecological quality might have declined. The following argument underlines this. While the abundance of large fish has declined because of fishing, the abundance of small fish has increased, probably as a result of lesser predation. Thus, the size composition of the community might be "restored" by more extensive fishing on small fish, which would be opposite to the conservation goals. While these problems make this element less suitable as an EcoQO, the metric has some advantage as a qualitative measure of ecological quality and as such might be used in a management context, but without the specific connotations linked to the concept of EcoQOs.

EcoQ element (o) Density of sensitive (e.g., fragile) species and EcoQ element (p) Density of opportunistic species

With respect to EcoQ element (o) density of sensitive (e.g., fragile) species and (p) density of opportunistic species, ICES has commenced the development of EcoQ metrics for the density of sensitive (e.g., fragile) species and the density of opportunistic species by classifying benthic invertebrate species recorded in recent North Sea surveys as sensitive (fragile) and opportunistic. Based on our stated definitions of sensitive (fragile) and opportunistic, a total of 180 taxa were identified as meeting the criteria for sensitive species (including biogenic structure-forming species as well as those with fragile morphological features) and 69 taxa as meeting the criteria for opportunists (including opportunistic scavengers). These lists are inevitably incomplete, since the biology and life history of all species were not known. The identification and assignation of species could be progressed with the assistance of specialists at a focused workshop. Monitoring changes in the abundance of sensitive (fragile) and opportunistic species presents many practical constraints, and the present sampling schemes in the North Sea are largely inadequate to detect species-specific trends in abundance on the spatial scales required. ICES considered five alternate approaches for developing the EcoQ elements for sensitive (fragile) and opportunistic species, from direct measurement of the absolute abundance of each sensitive or opportunistic species to an assessment of the density of a selection of indicator (sentinel) species. Direct measurements of the abundance of many species will be impractical and the power of affordable surveys to detect trends will be very poor. The most promising option may be to effectively monitor the abundance of a few indicator (sentinel) species, and this might provide a warning system to trigger further action. However, the monitoring and cost implications of this approach still need to be considered in detail. There is also a need to develop robust and objective criteria for the selection of the sentinel species. ICES remains convinced of the importance of healthy benthic communities as part of a well-managed North Sea ecosystem. ICES is keen to further develop appropriate EcoQOs for benthic systems but believes that this should be done in two ways, firstly through a focus on habitat quality and, secondly, through the development of EcoQOs targeted at specific issues, such as those already adopted as part of the pilot scheme.

Recommendations and advice

ICES advises that:

1. In relation to the development of EcoQ element (a) Spawning stock biomass of commercial fish species:

   a) The wording of the EcoQO should be modified slightly. Rather than "spawning stock biomass *also taking into account fishing mortality, ...*" [italics ours], the EcoQO should explicitly include both properties. The EcoQO should be based on the proportion of stocks where $SSB > B_{pa}$, **and** where $F < F_{pa}$, considered together.

   b) The wording should also make clear that it is the annual estimates of SSB and F that should comply with their respective reference points, and not the true SSB and F, which cannot be known in the year when the management decisions must be made.

2. With respect to EcoQ element (c) Seal population trends in the North Sea:

   a) *Current levels.* There is a lack of information on the present population levels of the seal stocks in the North Sea, especially for the harbour seal. It is hoped that these will be available for the largest part of the North Sea for consideration in 2004.

   b) *Data for assessment on whether the EcoQO is being met.* ICES recommends the use of the number of seal births as a sensitive parameter responding more rapidly than total population size to changes in habitat conditions such as food availability.

   c) *Draft guidelines.* ICES does not recommend a specific protocol for the monitoring of seal populations.

   d) *Management measures.* The management strategies for marine mammals applied by most countries in the OSPAR area are oriented towards maintaining or increasing marine mammal populations, so current management strategies are generally appropriate.

3. For EcoQ element (d) Utilization of seal breeding sites in the North Sea:

   a) To assess the usefulness of this EcoQ, and its further development, existing data on seal breeding sites should be analysed. This analysis, to be undertaken on the initiative of the lead country for this subject under OSPAR (the UK), should include a definition of appropriate temporal and spatial scales for this EcoQO. This analysis should then be reviewed by the appropriate working groups under ICES.

4. With respect to EcoQ element (f) Proportion of oiled common guillemots among those found dead or dying on beaches:

   a) ICES advises that OSPAR should request the lead country for this EcoQ (The Netherlands) to reanalyse the oiled seabird data on the basis of the geographical boundaries suggested in Section 6.5.4.1, below, and to examine trends in oiling and consistency in patterns between adjacent regions. This will also allow baselines to be set for the suggested regions.

   b) ICES advises that a national coordinator for beached bird counts is required in each country around the North Sea, and that one international coordinator should be appointed.

   c) ICES advises that a period of at least five years in which an average of 10% oiled common guillemots has been recorded should occur before the conclusion that the objective has been reached could be justified statistically. OSPAR might wish to modify the description of the EcoQO to take account of this.

   d) ICES advises that the information provided by this EcoQO would be enhanced by analysing samples of relevant pollutants taken from the plumage of seabird corpses. These samples can help indicate generic (and sometimes specific) sources of pollutants, thus enabling management actions to be appropriately targeted.

   e) ICES advises that the provision of port waste reception facilities, improved detection of illegal behaviour, and prosecution (and punishment) of offenders have been shown to be suitable management measures to reduce amounts of oil discharged to the sea.

5. With respect to EcoQ element (g) Mercury concentrations in seabird eggs and feathers:

   a) The mercury level in seabird feathers should be used as a metric for mercury levels in the marine environment. As a reference level for this EcoQO, the mercury level in the feathers of seabirds collected over 100 years ago should be used. A realistic objective, consistent with current scientific information, would be a level of 1.5 times the reference level. The species to be monitored are common tern, black-legged kittiwake, common guillemot, and northern gannet, from colonies in the southern and northern North Sea.

6. With respect to EcoQ element (h) Organochlorine concentrations in seabird eggs:

   a) The level of organochlorines in seabirds' eggs should be used as a measure for organochlorine levels in the marine environment. Objectives for the levels in oystercatcher and common tern of: <20 ng total PCBs g^{-1} egg fresh mass, <10 ng DDT and metabolites g^{-1} egg fresh mass, <2 ng HCB g^{-1} egg fresh mass, and <2 ng HCH g^{-1} egg fresh mass, would all be consistent with current scientific information on levels and trends of these substances, and effective management of their introduction into the environment. Other species which should be monitored are common eider, northern gannet, and common guillemot.

7. With respect to EcoQ element (i) Plastic particles in stomachs of seabirds:

   a) ICES can support the proposed reference level of less than 2% of beach-washed fulmars having ten or more particles in the stomach out of a sample of fifty fulmars or more. ICES recommends further that more information be gathered on the nature of the particles in the stomach.

8. For EcoQ element (j) Local sandeel availability to black-legged kittiwakes:

   a) ICES will continue to examine the relationship between black-legged kittiwake breeding success, sandeel availability, and sandeel fishing in 2004 and aim to assimilate new research findings reported in 2003 with those on which previous advice has been based. The outcome will be included in the 2004 advice on this EcoQ element.

9. With respect to EcoQ element (k) Seabird population trends as an index of seabird community health:

   a) There is a need to assess in more detail to what extent the present level of monitoring in the North Sea countries is adequate to fulfil the proposed EcoQO. This assessment should take into account the representativeness of the sample of populations in relation to the overall seabird community. In particular, it should be investigated whether the selection of species reflects the main ecological groups of seabirds in terms of their range of diets, habitat use, and life-history strategies, and whether the national programmes need to be adjusted in order to ensure that a sufficient part of the North Sea population of the selected target species is being monitored. However, as total bird counts are only made on a very irregular basis, a regular assessment of this EcoQO is forced to be limited to a selection of reference areas/colonies ("key sites"). The identification of the most suitable areas/colonies in this context will be a time-consuming issue that needs to be addressed carefully in the assessments suggested above. As a first step, ICES recommends that a detailed analysis of trends in individual colonies of kittiwakes should be carried out on the existing data (predominantly from UK seabird surveys and monitoring). This could provide for a better understanding of how colony selection may be made to render an EcoQ metric that is representative of the North Sea as a whole.

10. With respect to EcoQ element (l) Changes in the proportion of large fish:

    a) At this stage, the proposed metrics provide no clear basis for establishing an EcoQO for use in short-term fisheries management. However, they might be used more qualitatively as indicating deterioration or recovery of the ecological quality of fish communities over longer time spans.

    b) To facilitate future interpretation of spatial-temporal trends in these metrics, it is of utmost importance that satellite-based information on fishing activity as presently collated by national inspectorates is made available for scientific purposes.

11. In relation to the development of EcoQ elements (o) Density of sensitive (e.g., fragile) species and (p) Density of opportunistic species:

    a) ICES should organize a workshop to develop criteria for, and then identify, the species that should be considered under elements (o) and (p). This workshop should draw upon as wide a community of expertise as possible.

    b) The terms of reference for the workshop should include identification of the species and sources of quantitative information on their historic and current abundance (spatially resolved).

    c) In 2004, ICES should continue to develop metrics, reference points, and sampling protocols for EcoQ elements (o) and (p), with an emphasis on identifying and developing: (i) an index of opportunists or sensitivity, (ii) a metric based on the proportion of species that are opportunistic or sensitive, and (iii) the density of selected indicator (sentinel) species.

Scientific background

6.1.1 Over-arching considerations

6.1.1.1 The OSPAR Ecological Quality Objectives Framework

At its meeting in January 2003, the OSPAR Biodiversity Committee (BDC) adopted a general framework for the development of a set of Ecological Quality Objectives (EcoQOs) for the North Sea and the general activities that need to be undertaken to implement them, including ensuring that these developments are properly coordinated and communicated with other stakeholders.

The framework recognizes that there is a difference in the extent to which work on each of the issues, ecological quality elements, and ecological quality objectives has progressed, and therefore the extent to which the elements in the work programme have already been achieved. This general framework is therefore followed by descriptions of the work to be undertaken on each individual Ecological Quality Element for which OSPAR has agreed that an EcoQO for the North Sea should be developed, and in some cases applied in a pilot project for the North Sea. ICES has used this framework to develop a template for the review of the EcoQOs by ICES Working Groups in 2004; this template is described in Section 6.1.2, below.

6.1.1.2 Reference levels

There continues to be some confusion about the "reference level" as used in the EcoQ – EcoQO framework. OSPAR defines a reference level as "*the level of the EcoQ where the anthropogenic influence on the ecological system is minimal*" (Anon., 1999). Given the practical difficulties in establishing this level, the EcoQ – EcoQO framework recognizes baselines as being either the reference level or the earliest measured value of a parameter where no reference level can be determined. These then act as proxies for the predicted levels in the absence of human impacts. It specifically cautions against the use of temporal trends observed recently to hindcast to a pristine state.

The framework does NOT require that the management objective (the EcoQO) is the reference level, as that would imply no use of the marine environment, i.e., zero contamination and fish stocks all at virgin levels and population structures. Determination of the objective is a societal issue, not a scientific issue. Science needs to advise on the baseline and can provide commentary on the consequences of setting the objective at various levels, BUT it is for society to decide whether it wants a fish stock to be virginal, at B_{pa} or extirpated! Science can then advise managers on how to achieve this objective.

6.1.1.3 The Signal Detection Evaluation Approach

ICES has evaluated the EcoQOs within the general framework of signal detection and decision theory. Within this framework, the performance of an EcoQ element and EcoQO are judged relative to the reliability of the information that they convey for management actions. A good EcoQO is one where, if society's objectives are being met, the annual estimate of the EcoQ element meets the EcoQO, and if society's objectives are not being met, the annual estimate of the EcoQ element does not meet the EcoQO. In signal detection theory, these are known as "hits" (true positives and true negatives). A poor EcoQO would often give erroneous information; either society's objective was being met yet the annual estimate of the EcoQ element would not meet the EcoQO, or society's objective was not being met yet the annual estimate of the EcoQ element would meet the EcoQO nonetheless. Within signal detection theory, these types of errors are known as "False alarms" and "Misses", respectively. The signal detection theory framework provides a systematic analysis of the error rate of an EcoQO over a historic period, which informs managers of the reliability of the EcoQO as a basis for individual decisions on management actions.

For this review, the full analytical framework was used only for the EcoQO on spawning stock biomass of commercially exploited fish, where the EcoQ Objective was specified and a historic time series exists. As progress is made on identifying EcoQOs for other EcoQs, and consolidating the historic data for their evaluation, the full framework will be applied to the other EcoQs as well.

6.1.2 The ICES EcoQ – EcoQO process

Recent years have seen many ICES Study and Working Groups receive terms of reference that have involved consideration of various aspects of the proposed or adopted EcoQs. ACE has reviewed and commented on this work and used it as the basis for advice to OSPAR and other interested parties. In order to maximize the effectiveness of the ICES input to the review of EcoQs and EcoQOs in 2005, ACE provides here a standard template for specialist groups to consolidate their reviews of the EcoQ – EcoQO scheme. This template is shown in Table 6.1.2.1, and is accompanied by explanatory notes.

6.1.2.1 Notes for the use of the ACE EcoQO advisory template

The purpose of the template is to provide a standard format for the production of advice concerning EcoQOs. It is envisaged that the template will be completed by specialist Working Groups and reviewed by ACE prior to communication to the client. In many cases, much of the work to fill in the template has already been completed but has not been presented in this summary format. It is envisaged that the template will be accompanied by a substantive piece of text detailing the scientific basis for the summary presented.

1) Issue

This indicates which of the ten issues identified in the Bergen Declaration from the Fifth North Sea Conference this EcoQ refers to.

2) Element

This indicates the element of the issue to which the EcoQ metric refers.

3) ICES criteria

This provides for an assessment of the performance of the metric against the ICES criteria for a good EcoQ. In the advice template, this should be in the form of a Yes/No against each criterion. The box should then include some commentary.

Experience to date shows that few, if any, EcoQs will meet all the criteria. This should not be seen as being a definitive failure and the commentary should be used to highlight the strengths and weaknesses of the metric. For example, if an EcoQ only fails on the "understandable to non-scientists" criterion, it may still be a good management tool.

4) Ecological relevance/basis for the metric

What is the ecological importance of the measured parameter; what is the basis for using it as a measure of system health?

5) Current and historic levels

Details should be given of current levels of the EcoQ; these should be broken down at appropriate spatial scales.

Is it appropriate to set this EcoQO for the whole area; should it only apply in some areas; should different values apply in some areas?

The accompanying text should give details of all historic data sources and they should be summarized in the template. This includes presentation of time series or discontinuous temporal data sets.

6) Reference level

If information is available for the period prior to anthropogenic influences, this should be given here. If this information is not available, then the earliest available data should be presented with necessary supporting text (see item 4), above).

Table 6.1.2.1. ACE template for the review of EcoQ elements under development. Full details of the science behind the value presented here are given in the supporting text.

		COMMENTS	
1	**Issue**		
2	**Element**		
3	**ICES criteria**		Commentary (i.e., pattern of fails still makes this useful for communicating to non-specialists on health of system, or useful monitoring tool to trigger additional research)
	Relatively easy to understand by non-scientists and those who will decide on their use	Y/N	
	Sensitive to a manageable human activity	Y/N	
	Relatively tightly linked in time to that activity	Y/N	
	Easily and accurately measured, with a low error rate	Y/N	
	Responsive primarily to a human activity, with low responsiveness to other causes of change	Y/N	
	Measurable over a large proportion of the area to which the EcoQ metric is to apply	Y/N	
	Based on an existing body or time series of data to allow a realistic setting of objectives	Y/N	
4	**Ecological relevance/basis for the metric**		
5	**Current and historic levels (including geographic areas)**		
6	**Reference level**		
7	**Limit point**		
8	**Time frames**	*Detection of change*	
		Management advice	
9	**Advice on EcoQO options (scenarios)**	*Scenario 1*	
		Scenario 2	
		Scenario 3	
10	**Monitoring regimes**		
11	**Management measures to achieve EcoQO**		

If it is impossible to provide a reliable estimate of the pre-anthropogenic influences situation, this should be stated.

Where possible, an indication of the degree of natural variation in the reference level should be given.

7) Limit point

The level of the EcoQ beyond which severe or irreversible change will occur should be stated.

8) Time frames

What time scales are required to detect changes of a particular magnitude or trends of a particular size, taking into account variance in the data not attributable to human impacts? This could include a power analysis.

What is the appropriate time frame for the use of this metric in advice: annual, five yearly, decadal?

9) Advice on EcoQO options (scenarios)

It is not for science to set the EcoQ Objectives, but science has a key role in providing society with the information it needs to make the decision as to what configuration of ecosystem qualities it wishes to see. It must be borne in mind that the EcoQ – EcoQO framework sees these objectives not individually but as a suite.

A robust way of addressing this need is to provide a limited number of scenarios, for example, the consequences of the maximum level of human impact without exceeding the limit point, a minimum impact scenario, and one or two at some intermediate levels.

For example (these are simplistic accounts provided for illustrative purposes, the actual scenarios need to be developed with full scientific rigour), for commercial fish species the spawning stock biomass element may involve three scenarios:

Scenario 1 – the stock is held just above B_{pa} (so it has a high likelihood of being above B_{lim}). Consequences are irregular catches, uncertainty in market supply, overcapacity/capitalization of the fleet, high regulatory enforcement costs, perturbed ecosystem dynamics.

Scenario 2 – the stock is unexploited. Under this scenario, the stock will rebuild, but given current low levels the configuration of the rebuilt stock may differ from the virgin one. There are major economic consequences, not least the exporting of our environmental impacts to other nations where we subsequently source our fish.

Scenario 3 – the stock is exploited such that the estimated B is managed to vary around a value which is 25% above the B_{pa}. This would have some ecological impacts, predictable from Multispecies Virtual Population Analysis (MSVPA), and the impacts of that level of effort on the ecosystem and, probably, limited negative economic effects.

10) Monitoring regimes

The supporting text needs to critically evaluate the monitoring requirements needed to assess the status of the EcoQ. This text may be quite extensive in the supporting sections and needs to include consideration of both the temporal and spatial patterns of the sampling.

An indicative cost for the monitoring would be useful, i.e., number of vessel days, scientist days for sample processing, cost of laboratory analyses, etc.

Details should be given, including sites, levels of sampling (replication), and data analysis, of current monitoring programmes.

11) Management measures to achieve EcoQO

With respect to the scenarios developed, the management options that could be employed to deliver the EcoQO should be described—in summary in the table, in detail in the text.

6.1.3 Conclusions

OSPAR has set in place a framework for the development and application of EcoQOs for the ten issues identified in the Bergen Declaration. This programme represents progress, but the timetable is ambitious. In many cases, the scientific support is only rudimentarily developed (see Section 15, below).

ICES is well placed to take a lead role in the provision of scientific advice in many of the critical areas, but needs to ensure that it will be able to do so both in the time frames required and drawing upon a wider base of expertise than has been traditional. This will also include social science expertise. In order to advance this process, ACE has provided a template for use by expert groups tasked with advising on EcoQs.

6.1.4 Reference

Anon. 1999. Workshop on Ecological Quality Objectives (EcoQOs) for the North Sea. Scheveningen, The Netherlands, 1–3 September 1999. TemaNord 1999:591. 75 pp.

6.2 Development of EcoQ element (a) Spawning stock biomass of commercial fish species

Source of information

The 2003 Report of the Working Group on Ecosystem Effects of Fishing (WGECO) (ICES CM 2003/ACE:05).

Summary

The performance of the Ecological Quality Objective (EcoQO) that the spawning stock biomass of commercial fish species should be "above precautionary reference points for commercial fish species where these have been agreed to by the competent authority for fisheries management" was evaluated formally using the approach of signal detection and decision theory. The context for the evaluation was focused very narrowly on the use made of B_{pa} and F_{pa} in single years. In that narrow context, a perfect EcoQO will function as a reliable signal within the framework of signal detection and decision theory. That is, contrasting annual *estimates* of SSB and F against B_{pa} and F_{pa} should produce scientific advice to reduce fishing mortality always when the *true* SSB was below and/or the *true* F was above their respective precautionary reference points (B_{pa} and F_{pa}), but never when the *true* SSB was above B_{pa} and the *true* F was below F_{pa}. When the true SSB was above and the true F was below their respective reference points, the advice should be for *status quo* (or possibly increased) fishing. It is stressed that this context is **not** the same as a conventional fisheries management context. To analyse the performance of B_{pa} and F_{pa} as signals in a fisheries management context would have to take account of the fact that assessments are conducted, and advice is provided, annually. The performance of a signal in each year is affected by the performance of the same signal in the preceding year(s). It is stressed as well that an evaluation of the performance of SSB and F as *signals* in a fisheries management context is only one part of analysing the performance of the scientific advisory and management system as a whole. Analysing the performance of the overall management system is the real test of how well conservation and sustainable use are being achieved. Such a performance analysis should be based on an analysis of management as a control system working over time with feedback mechanisms.

The analyses were conducted using all stocks in the OSPAR area for which ICES conducts analytical assessments. Three criteria can be used to determine whether a stock is within these limits and, hence, the EcoQO was met:

- The estimate of SSB was above the Precautionary Reference Point ($SSB>B_{pa}$);
- The estimate of F was below the Precautionary Reference Point ($F<F_{pa}$);
- Both of the above ($SSB>B_{pa}$ and $F<F_{pa}$).

The standard for determining the rates of Hits (true positives and true negatives), Misses (not advising a reduction in F when it was necessary), and False Alarms (advising an unnecessary reduction in fishing) was to compare the conclusion about stock status in the assessment year with the conclusion that should have been reached that year, given the *current* estimate of stock status in that year, from a converged VPA. If the current test shows a high Hit rate and low rates of Misses and False Alarms, it is support for the view that precautionary reference points are an accurate and robust basis for fisheries management advice, generally advising managers to take actions that would move the stock in the proper direction. High Miss rates alone would suggest that precautionary reference points, as currently used, do not lead to advice that is sufficiently restrictive to ensure that stocks remain within safe biological limits. High False Alarm rates alone would indicate that precautionary reference points, as currently used, lead to overly intrusive management advice. Equal rates of Misses and False Alarms would mean that the precautionary reference points are a robust basis for sound management decision-making, although their overall rates would determine their overall accuracy.

The analyses demonstrated that using F alone will result in relatively low False Alarm rates but high Miss rates. Using SSB alone results in a strong decrease of Miss rates together with a markedly higher proportion of False Alarm rates. The best results were achieved using both criteria, with a 53% Hit rate, 23% Miss rate, and 24% False Alarm rate. This suggests that the current framework is sound, and implies that management is keeping stocks very near the boundary condition for safe biological limits. Analysis of the quantitative response in changes in TAC to the advice also showed that, in general, the advice was appropriate. If the EcoQO was not met, a strong reduction in TAC of about 18% was suggested in case of a correct advice (Hit), whereas there was an increase in TAC averaging between 10% (SSB) and 15% (F) in case of a Miss. If the EcoQO was met, a correct advice resulted in a suggested increase of the TAC between 26% (SSB) and 16% (F), whereas in case of a False Alarm the TAC was suggested to decrease between 9% and 18%. Historic trajectories of the performance of management advice showed no clear trends over time in Hits when based on SSB only, F only, or SSB and F together. However, Misses were particularly common in the mid-1990s when based on SSB alone or SSB and F together. When based on F alone, Misses appeared to decrease from the mid-1990s onward.

These analyses indicate that the EcoQO should be based on the proportion of stocks where $SSB>B_{pa}$, **and** where $F < F_{pa}$, considered together. The evaluation of B_{pa} and F_{pa} as EcoQOs indicates that Misses and False Alarms are about equally frequent. This suggests that advice based on SSB and F will not recommend catch reductions when in fact they are needed for about one stock in five, although the Miss rate seems to be going down in recent years. However, advice based on SSB and F relative to their reference points recommends unnecessary catch reductions about equally often. This symmetry in error rate *de facto* treats both types of errors (Misses and False Alarms) as equally undesirable. An effective way to reduce the error rate would be for management to try to maintain stocks at biomasses somewhat above B_{pa} and fishing mortality somewhat below F_{pa}. This was always the intent of ICES, as it repeatedly stresses in its advice that the precautionary reference points should be treated as boundaries on SSB and F, rather than as targets.

Recommendations and advice

ICES recommends that:

a) the wording of the EcoQO be modified slightly. Rather than "spawning stock biomass *also taking into account fishing mortality*, …" [italics ours], the EcoQO should explicitly include both properties. The EcoQO should be based on the proportion of stocks where $SSB > B_{pa}$, **and** where $F < F_{pa}$, considered together.
b) The wording should also make clear that it is the annual estimates of SSB and F that should comply with their respective reference points, and not the true SSB and F, which cannot be known in the year when the management decisions must be made.

ICES concludes that:

The evaluation of B_{pa} and F_{pa} as EcoQOs indicates that:

1) Misses and False Alarms are about equally frequent. Advice based on SSB and F will not recommend catch reductions when in fact they are needed for about one stock in five, although the Miss rate seems to be going down in recent years. However, advice based on SSB and F relative to their reference points recommends unnecessary catch reductions about equally often. This means that the advice is unbiased, and should be the best available advice on management.
2) Management is keeping stocks near their precautionary reference points, such that many "errors" in advice are the consequence of annual estimation inaccuracies around these boundary conditions.

3) The Miss and False Alarm rates associated with annual estimates of F and SSB relative to their precautionary reference points would be reduced if management maintained stocks at biomasses somewhat above B_{pa} and fishing mortality somewhat below F_{pa}. This was always the intent of ICES, as it repeatedly stresses in its advice that the precautionary reference points should be treated as boundaries on SSB and F, rather than as targets.

4) This evaluation of the performance of B_{pa} and F_{pa} as EcoQOs is valid ONLY in the narrow context of annual assessments and advice for decisions in a single year. The performance of B_{pa} and F_{pa} as guides to management decision-making would have to take account of the strong negative feedback built into the year-after-year sequences of assessments and advice. The performance of the entire fisheries management system would additionally have to take account of the feedback control systems in the entire assessment/ advisory/management process.

Scientific background

6.2.1 Background

Table B in the Bergen Declaration defines Ecological Quality Element (a) as the spawning stock biomass of commercial fish species. The associated Ecological Quality Objective (EcoQO) is that the spawning stock biomass of commercial fish species should be "above precautionary reference points for commercial fish species where these have been agreed to by the competent authority for fisheries management". The relevant precautionary reference points "are those for the spawning stock biomass, also taking into account fishing mortality, used in advice given by ICES in relation to fisheries management". ICES, through the work of the Advisory Committee on Fishery Management (ACFM), has established B_{pa} and F_{pa} (see box for definitions) as the respective precautionary reference points of spawning stock biomass (SSB) and fishing mortality (F) for use in formulating advice. They are set on a stock-specific basis, and take account of both stock dynamics and uncertainties in the assessment. B_{pa} is the spawning biomass at and above which there is a low probability that the true SSB is so low that productivity is impaired (i.e., has a low probability of being at B_{lim}). F_{pa} is the fishing mortality at and below which the true fishing mortality has a low probability of leading to stock collapse (i.e., has a low probability of being at F_{lim}). The ICES Precautionary Approach (PA) reference points address the fact that management is based on estimates of biomass and fishing mortality, and they have uncertainty relative to the true values for spawning biomass and fishing mortality. Comparably, the reference in this EcoQO has to be to the *estimate* of SSB that should be above the precautionary reference points, and not the true value of the SSB (which cannot be known in the assessment year).

Definition of terms

$\mathbf{F}_{lim}$ is the limit fishing mortality which should be avoided with high probability because it is associated with unknown population dynamics or stock collapse. There are very few stocks for which $\mathbf{F}_{lim}$ is accurately known. Some stocks in the ICES area have collapsed in the past when fishing mortality exceeded $\mathbf{F}_{lim}$, but generally speaking, the fishing mortality rate at which the probability of stock collapse becomes unacceptably high remains unknown. In order to have a high probability that fishing mortality will be below $\mathbf{F}_{lim}$, a precautionary reference point, $\mathbf{F}_{pa}$ lower than $\mathbf{F}_{lim}$, is defined. Used as a constraint on fishing, $\mathbf{F}_{pa}$ is designed to ensure that there is a high probability that $\mathbf{F}_{lim}$ will be avoided and that the spawning stock biomass will remain above the threshold below which the probability of good to average recruitment is decreased. In other words, $\mathbf{F}_{pa}$ is a device to ensure that recruitment overfishing does not take place. It is the upper bound on fishing mortality rate to be used by ICES in providing advice. $\mathbf{F}_{pa}$, given uncertainties, must have a high probability of being below $\mathbf{F}_{lim}$, and it must have a high probability of being sustainable based on the history of the fishery, i.e., it should be set in the range, and imply a biomass, within those previously perceived to be acceptable. Fishing mortality rates in excess of $\mathbf{F}_{pa}$ will be regarded as "overfishing".

$\mathbf{B}_{lim}$ is the limit spawning stock biomass, below which recruitment is impaired or the dynamics of the stock are unknown. Stocks may become depleted due to reduced recruitment even if fishing mortality is successfully maintained at or below $\mathbf{F}_{pa}$. Furthermore, efforts to restrain fishing below $\mathbf{F}_{pa}$ may not be successful and biomass may decline as a result. Clearly, therefore, in addition to a constraint on fishing mortality, it is desirable to have a biomass-based constraint to prevent stock decline to values where expected recruitment is low or unknown.

Whereas $\mathbf{F}_{pa}$ defines an "overfishing threshold", a definition of when the stock is regarded as being in a "depleted state" is also necessary. A threshold in this respect, $\mathbf{B}_{pa}$, needs to be set to ensure a high probability of avoiding reducing the stock to a point, $\mathbf{B}_{lim}$, below which recruitment is impaired or the dynamics of the stock are unknown. $\mathbf{B}_{pa}$ is the biomass below which the stock would be regarded as potentially depleted or overfished.

(from ICES, 2001a)

6.2.2 Approach

For EcoQOs relating to spawning stock biomass (SSB) or fishing mortality (F), the appropriate sources of information for North Sea stocks are the regular assessments by the ICES Working Groups reporting to

ACFM. Monitoring protocols have been developed as part of the assessment process, for catch reporting, catch monitoring, and the conduct of research vessel surveys. Reliable catch reporting is necessary for reliable assessments, but protocols for catch reporting are driven more by enforcement capabilities and the requirements of fishing plans than by the needs of good science.

Assessment methods are standardized as much as is appropriate by ACFM, the Resource Management Committee, and their subsidiary groups. The methods are continually under review, and are improved as better methods become available. Estimates of SSB and F from assessments have associated uncertainty, and implications of the uncertainty for their use as EcoQOs are discussed in several parts of this section.

Since this advice was requested by OSPAR, the "North Sea" has been interpreted as the greater North Sea area as defined by OSPAR Region II (Figure 5.1, Section 5). This includes all of ICES Sub-area IV (the geographic North Sea), Division IIIa (the Skagerrak and Kattegat), Divisions VIId and e (eastern and western Channel), and part of Division VIa (north and west of Scotland). Figure 6.2.2.1 gives a map of the ICES fishing areas.

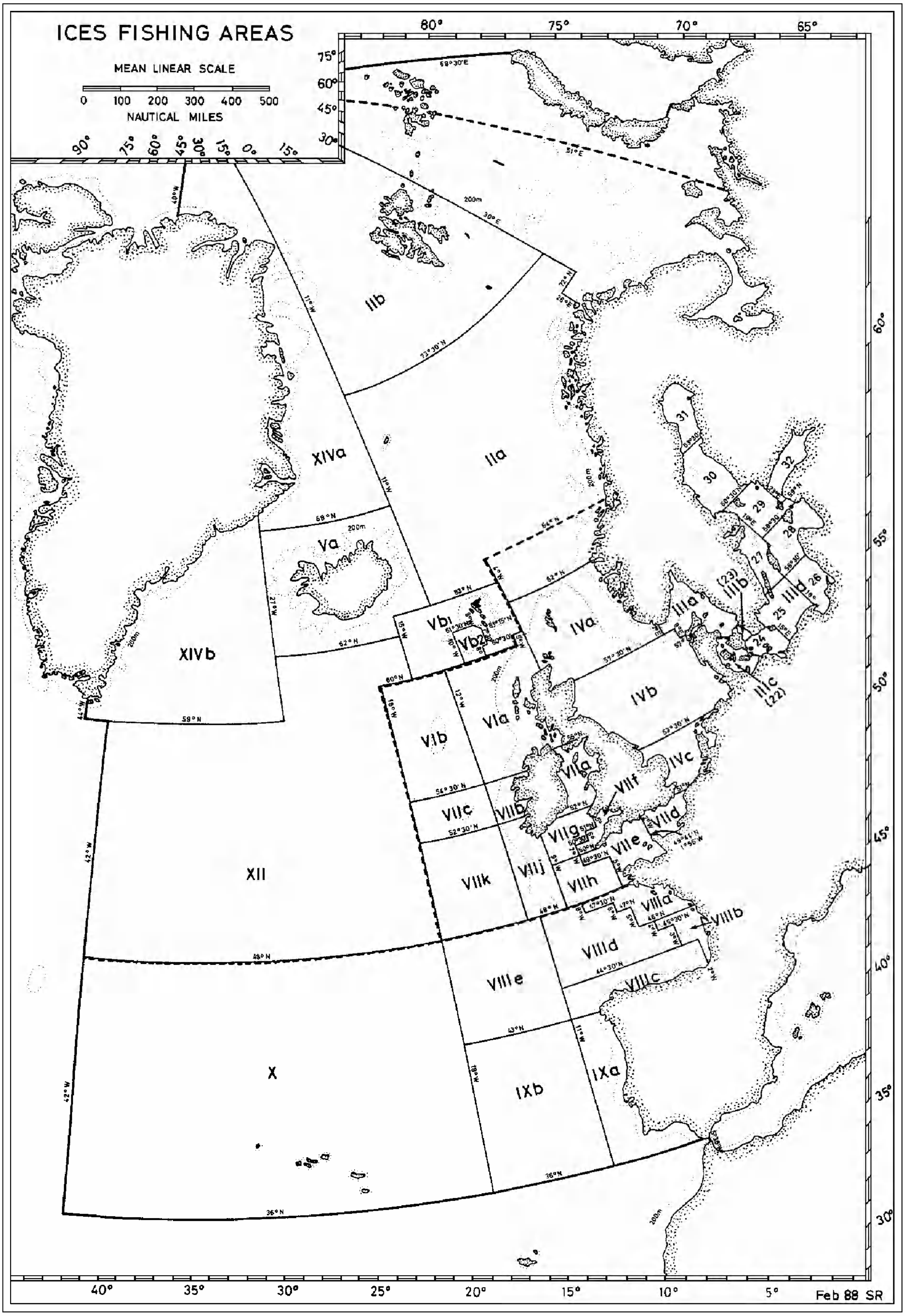
Table 6.2.2.1 lists the stocks used for the analyses described below, which comprise all stocks that fall within the OSPAR-defined North Sea area and for which ICES provides quantitative scientific analysis. The following criteria were used to determine whether stocks should be included in the analysis:

- Stocks occurring in the Kattegat were excluded if they were assessed along with stocks in the Baltic Sea;
- Stocks from the north of Scotland were excluded if the assessment unit was Sub-area VI as a whole;
- Stocks in the western Channel were excluded if they were assessed along with fish in the Celtic Sea or Bay of Biscay;
- Stocks were excluded if there were no precautionary levels of SSB or F (i.e., herring in Divisions VIa (South) and VIIb,c);
- Stocks were excluded if the state of the stock was not assessed in 2002 (i.e., North Sea horse mackerel (*Trachurus trachurus*) (Division IIIa (eastern part), Divisions IVb,c, VIId) or cod in Division VIb);
- Stocks were excluded if no TAC was set (i.e., sprat in Sub-area IV).

Table 6.2.2.1. List of stocks that were used in the analysis.

Fishery	Species	Area
Flatfish	Plaice	Division VIId (Eastern Channel)
		Division VIIe (Western Channel)
		Sub-area IV (North Sea)
	Sole	Division VIId (Eastern Channel)
		Division VIIe (Western Channel)
		Sub-area IV (North Sea)
Industrial	Norway pout	Sub-area IV (North Sea) and Division IIIa (Skagerrak/Kattegat)
	Sandeel	Sub-area IV (North Sea)
Pelagic	Herring	Divisions VIa (South) and VIIb,c
		Sub-area IV, Division VIId, and Division IIIa (autumn spawners)
	Mackerel	combined Southern, Western, and North Sea spawning components
Roundfish	Cod	Division VIa (West of Scotland)
		Sub-area IV (North Sea), Division VIId (Eastern Channel), and Division IIIa (Skagerrak)
	Haddock	Division VIa (West of Scotland)
		Division VIb (Rockall)
		Sub-area IV (North Sea) and Division IIIa (Skagerrak/Kattegat)
	Saithe	Sub-area IV (North Sea), Division IIIa (Skagerrak), and Sub-area VI (West of Scotland and Rockall)
	Whiting	Sub-area IV (North Sea) and Division VIIId (Eastern Channel)

Figure 6.2.2.1. Map of ICES Fishing Areas.



Many other fish species exist in the North Sea. A number of these are fished commercially, but they are not assessed and estimates of SSB are not available. Therefore, such stocks cannot be part of the EcoQ metric, regardless of their status. As information becomes available to allow additional commercially exploited species or stocks to become assessed, this will likely be led by ACFM, with the results being used by ACE.

6.2.3 Performance of criteria

For each of the stocks considered, the EcoQO is that the stock is within safe biological limits. Three criteria can be used to determine whether a stock is within these limits and hence the EcoQO was met:

1) the estimate of SSB was above the Precautionary Reference Point ($SSB>B_{pa}$);
2) the estimate of F was below the Precautionary Reference Point ($F<F_{pa}$);
3) both of the above ($SSB>B_{pa}$ and $F<F_{pa}$).

With respect to the Bergen Declaration, however, the Ecological Quality Objective (EcoQO) is that the stock biomass of commercial fish species should be "above precautionary reference points for commercial fish species where these have been agreed by the competent authority for fisheries management", and the relevant precautionary reference points "are those for the spawning stock biomass, also taking into account fishing mortality, used in advice given by ICES in relation to fisheries management".

If interpreted as referring to annual estimates, and not the true values of SSB or F, the EcoQO is consistent with the basis on which ICES provides advice on fisheries management. In ICES advice, precautionary reference points are treated as the bounds on "safe biological limits". ICES makes clear that management should keep stocks consistently above B_{pa} and below F_{pa}, which would require some degree of risk-averse decision-making relative to the reference points. ICES is quite specific that management should not treat them as targets, around which risk-neutral decision-making would result in the the stock varying randomly.

Evaluation of the performance of any EcoQO has to be placed in the context in which the EcoQO will be used in decision-making. In general, the management contexts for EcoQOs have not been specified, so an evaluation of B_{pa} and F_{pa} as "typical" EcoQOs should be done in a very general context. Such a general evaluation context will focus narrowly on the use made of them in single applications, taken as if a management decision is made only in one year and not revisited for some time. Many environmental management issues are dealt with in that way, so this is not an unrealistic context for general evaluation of the performance of EcoQOs. In that narrow context, a perfect EcoQO will function as a reliable signal within the framework of signal detection and decision theory. That is, contrasting annual *estimates* of SSB and F against B_{pa} and F_{pa} should produce scientific advice to reduce fishing mortality <u>always</u> when the *true* SSB was below <u>and/or</u> the *true* F was above their respective precautionary reference points (B_{pa} and F_{pa}), but <u>never</u> when the *true* SSB was above B_{pa} <u>and</u> the *true* F was below F_{pa}. When the *true* SSB was above and the true F was below their respective reference points, the advice should be for *status quo* (or possibly increased) fishing.

It is stressed that this context is **not** the same as a conventional fisheries management context. To analyse the performance of B_{pa} and F_{pa} as signals in a fisheries management context would have to take account of the fact that assessments are conducted and advice is provided annually, and management decisions in each year are affected by the consequences of the decision made the year before. As a result of the annual sequence of management decisions, the performance of a signal in each year is affected by the performance of the same signal in the preceding year(s). If the advice were imperfect in one year for any reason, and management implemented it, the result would be that in the next year there would be an even larger discrepancy between the true population value and the reference value. That is, the difference between true F and B and their respective reference points will be even larger. The larger discrepancies should be easier for the signal (estimates of F and SSB) to pick up correctly, so there would be an increased probability of catching the true signal and providing the correct advice in the next year. Hence, when the signal is evaluated in a fisheries management context, the system as a whole means that inaccuracies in advice in one year are increasingly likely to be corrected in the subsequent year. Therefore, the occurrence in a given year of a case of inaccurate advice in a signal detection theory context does not mean that there is a conservation failure in the stock as a whole. It simply means that within the context of estimates of B and F from annual assessments that are contrasted with fixed reference points, the estimates are not perfectly reliable signals. Occasionally they prompt advice that does not direct managers to reduce fishing when they should, and at other times may direct managers to reduce fishing when it was not essential to keep the stock within safe biological limits. This analysis is intended to quantify how frequent "occasionally" is.

It is stressed as well that an evaluation of the performance of SSB and F as *signals* in a fisheries management context is only one part of analysing the performance of the scientific advisory and management system as a whole. Analysing the performance of the overall management system is the real test of how well conservation and sustainable use are being achieved. Such a performance analysis should be based on an analysis of management as a control system working over time with feedback mechanisms. If there are no major bias problems in stock assessments so that stock assessments are only subject to variance uncertainty, there are inherent negative feedback mechanisms in place. These will rectify the effects of deviations between point estimates and "true" values over time, as well as many (but possibly not all) forms of decision and implementation errors. For an evaluation of the performance of the fisheries advisory and management framework as whole, the present analysis based on signal theory should be supplemented with an analysis of the performance of the control feedback mechanisms and in relation to the overall achievement of conservation targets over time.

The analyses presented here of B_{pa} and F_{pa} in a signal detection theory context are only possible because stock assessments converge going back in time. Hence, for stocks exploited at rates typical of the ICES area, SSBs and Fs from the beginning of a time series until a couple of years before the most current year of an assessment can be considered to be the "true" values of SSB and F (at least within the framework in which the values of B_{pa} and F_{pa} were also estimated). If management succeeded in keeping the stocks above B_{pa} and fishing mortality below F_{pa} for the full time series, the tests would likely be trivial. On the other hand, the test would be most powerful if stock biomass and fishing mortality were highly variable, with frequent and large excursions above and below their respective reference points. If managers are ignoring clear ICES advice, and treating the precautionary reference points as targets rather than boundaries, then stock status might vary randomly around the reference points. In such cases, the signal detection/decision theory analysis might pick up a number of true "errors" due to small and random variations in annual estimation, but the errors would be individually small. In those cases, it would be particularly informative to examine the symmetry of the different types of inaccurate advice. ICES frequently expresses concern about the possible bias in assessments (ICES, 2001a, 2002), and if this bias is an important cause of inaccurate advice, then the majority of inaccuracies should be of only one or the other of the two types listed below. If the two types of inaccuracies in advice are about equally frequent, then the advice is still, on average, the best guide available to correct management decisions. Combined with the likely amplification of the signal in the following year, if it was inaccurate in a given year, the scientific advice should not be the cause of conservation failures. This, in turn, would make it a good EcoQO, even if the signal detection error rate was non-zero.

For each stock, the evaluation of the actual annual management advice and actions, as tabulated in the "Catch Data" and assessment output tables from ICES (2002), was based on four scenarios:

1) **"True" stock does not meet EcoQO, advice is to reduce fishing ("Hit", true warning)**; the estimate of SSB and/or F *in the assessment year* led to advice to reduce catch when the estimate of SSB and/or F *in the 2002 assessment* now indicates that the Ecological Quality (EcoQ) of the stock did not meet its objective (EcoQO) (i.e., respectively SSB < B_{pa}, F > F_{pa}, or SSB < B_{pa} and F > F_{pa});

2) **"True" stock does not meet EcoQO, advice is status quo ("Miss")**; the estimate of SSB and/or F *in the assessment year* led to advice for status quo or increased TAC when the estimate of SSB and/or F *in the 2002 assessment* now indicates that the stock did not meet its EcoQO;

3) **"True" stock meets EcoQO, advice is to reduce fishing ("False Alarm")**; the estimate of SSB and/or F *in the assessment year* led to advice to reduce catch when the estimate of SSB and/or F *in the 2002 assessment* now indicates that the stock met its EcoQO;

4) **"True" stock meets EcoQO, advice is status quo ("Hit", correct)**; advice for status quo or increased TAC when the estimate of SSB and/or F *in the 2002 assessment* now indicates that the stock did meet its EcoQO.

Signal detection theory was applied to these scenarios and the proportion of Hits (1 and 4), Misses (2), and False Alarms (3) were determined per year as the proportion of the stocks for which the respective scenarios applied.

If the current test shows a high Hit rate and low rates of Misses and False Alarms, it is support for the view that precautionary reference points are an accurate and robust basis for fisheries management advice, generally advising managers to take actions that would move the stock in the proper direction. High Miss rates alone would suggest that precautionary reference points, as currently used, do not lead to advice that is sufficiently restrictive to ensure that stocks remain within safe biological limits. High False Alarm rates alone would indicate that precautionary reference points, as currently used, lead to overly intrusive management advice. Equal rates of Misses and False Alarms would mean that the precautionary reference points are a robust basis for sound management decision-making, although their overall rates would determine their overall accuracy. As for their value as EcoQOs, it is unclear what level of performance would have to be met in order for an EcoQO to be useful. However, as long as their performance characteristics are understood, their use can at least be explained.

The actual performance of B_{pa} and F_{pa} as EcoQOs, and as guides to fisheries management, is presented in Table 6.2.3.1.

Table 6.2.3.1. Proportion (%) of Hits, Misses, or False Alarms depending on the criteria used (i.e., respectively SSB > B_{pa}, F < F_{pa}, or SSB > B_{pa} and F < F_{pa}) and the type of fishery.

Criteria	**Fishery**	**Hit**	**Miss**	**False Alarm**
SSB	All	51	25	24
SSB	Flatfish	52	18	30
SSB	Industrial	16	21	63
SSB	Pelagic	57	26	17
SSB	Roundfish	56	29	15
F	All	49	44	7
F	Flatfish	52	43	5
F	Pelagic	27	63	10
F	Roundfish	52	40	8
F and SSB	All	53	23	24
F and SSB	Flatfish	50	17	33
F and SSB	Pelagic	57	26	17
F and SSB	Roundfish	55	25	20

Overall, the main difference between the criteria used is that using only F will result in relatively low False Alarm rates but high Miss rates. Using only SSB results in a strong decrease of Miss rates together with a markedly higher proportion of False Alarm rates. The best results were achieved using both criteria, with a 53% Hit rate, 23% Miss rate, and 24% False Alarm rate. This suggests that the current framework is sound, and implies that management is keeping stocks very near the boundary condition for safe biological limits.

Tables 6.2.3.2 and 6.2.3.3 give a quantitative indication of the true impact of the advice depending on the scenario: not just that advice was provided, but how management actually responded to the advice and the indicator. This also shows that, in general, the advice was appropriate. If the EcoQO was not met, a strong reduction in Total Allowable Catch (TAC) of about 18% was suggested in case of a correct advice (Hit), whereas there was an increase of TAC averaging between 10% (SSB) and 15% (F) in case of a Miss. If the EcoQO was met, a correct advice resulted in a suggested increase of the TAC between 26% (SSB) and 16% (F), whereas in case of a False Alarm the TAC was suggested to decrease between 9% and 18%. Overall, the advice using SSB appears more appropriate, with relatively small changes in case of a Miss or False Alarm but relatively higher changes of TAC in case of Hits. This again implies that management is keeping stocks near their precautionary reference points, such that many “errors” in advice are the consequence of relatively modest estimation inaccuracies around these boundary conditions.

Table 6.2.3.2. The average change of the TAC (%) that was actually implemented for various scenarios, i.e., EcoQO (SSB> B_{pa}) is met (1) or not met (0) and advice is correct (1) or not correct (0).

Scenario	**EcoQO**	**Advice**	**Flatfish**	**Industrial**	**Pelagic**	**Roundfish**	**Total**
1	0	1	−11.8		−37.7	−21.5	−18.8
2	0	0	9.6	0.0	8.7	12.2	10.3
3	1	0	−6.0	−2.4	−13.1	−17.7	−9.4
4	1	1	23.9	11.4	10.8	38.9	26.1

Table 6.2.3.3. The average change of the TAC (%) that was implemented for various scenarios, i.e., EcoQO (F<F_{pa}) is met (1) or not met (0) and advice is correct (1) or not correct (0).

Scenario	**EcoQO**	**Advice**	**Flatfish**	**Pelagic**	**Roundfish**	**Total**
1	0	1	−11.7	−34.5	−20.9	−18.1
2	0	0	11.5	10.2	20.8	15.5
3	1	0	−7.3	−4.3	−26.4	−17.5
4	1	1	17.0	5.6	17.4	16.3

6.2.4 Historic trajectories

The Ecological Quality Objective proposed by the Fifth North Sea Conference could be interpreted to mean that the trends in SSB relative to B_{pa} and/or F relative to F_{pa} should be reported for every stock, as is attempted in these analyses (Table 6.2.3.1). There is an alternative interpretation of the EcoQO proposed at the Conference, i.e., that a composite indicator recording the proportion of stocks within safe biological limits should be produced. In such a case, the reference level might be 100% of stocks above their precautionary reference points (in the pristine state, fishing is always zero and biomasses are within the range of variation of unharvested stocks). The target level would allow for assessment uncertainty, and the desired percentage of stocks complying with their precautionary reference points would correspond to the overall risk tolerance used when setting the precautionary reference points to begin with.

Both of these interpretations of the EcoQO and their associated target and reference levels are consistent with the management goals for the individual stocks as assessed by ACFM. For each of these stocks, the ACFM and ICES goal is to keep SSB above B_{pa} and F below F_{pa}. In this context, the adoption of ecosystem-based management would not result in lower conservation standards than are already in place and can therefore not have any adverse impact on the conservation and management goals for target stocks. However, it would be necessary for management to be highly risk intolerant with regard to achieving this target of 100%. Given the quantified performance of B_{pa} and F_{pa} as EcoQOs, increasing the risk intolerance of managers for Misses, in particular, (that is, being certain that all *true* stock sizes remain within safe biological limits) would necessarily require decisions that would result in more False Alarms.

The historic trajectories used were the time series of SSB and F estimates from the 2002 assessments for as far back as ICES has provided quantitative catch advice. These figures are all available on the ICES website (http://www.ices.dk/committe/acfm/comwork/report) as the third figure in each assessment.

Historic trajectories of the performance of management advice showed no clear trends in Hits when based on SSB only, F only, or SSB and F together (Figures 6.2.4.1 to 6.2.4.3). However, from these figures it appears that Misses were particularly common in the mid-1990s when based on SSB alone or SSB and F together. When based on F alone, Misses appeared to decrease from the mid-1990s onward. Because the assessments have not converged enough to be confident of what the "true" (at least stable) estimates of F and SSB in 2002 will be, this year was not included in the analysis.

The historic trajectories of the suggested EcoQ metrics for the commercial species are shown in Figure 6.2.4.4. This figure shows that at present for about 40% of the stocks SSB is above B_{pa} and for about 15% of the stocks F is below F_{pa}. All stocks that meet the criteria based on F also meet those based on SSB.

6.2.5 Basis for advice on management measures

Management measures to keep SSB and/or F within safe biological limits are widely referenced in ICES advice on fisheries management (ICES, 2002). Above all, they consist of keeping catches (and effort) at sustainable levels and minimizing wasteful fishing practices. They may be supported by technical measures, and spatial and temporal closures.

6.2.6 Reporting of the EcoQO for commercial fish stocks

Based on these results, ICES recommends that:

- the wording of the EcoQO be modified slightly. Rather than "spawning stock biomass *also taking into account fishing mortality, ...*" [italics ours], the EcoQO should explicitly include both properties. The EcoQO should be based on the proportion of stocks where SSB>B_{pa} **and** where F < F_{pa}, considered together.
- the wording should also make clear that it is the annual estimates of SSB and F that should comply with their respective reference points, and not the true SSB and F, which cannot be known in the year when the management decisions must be made.

The existing management approaches for individual stocks are all based on an assumption that the objective of management is to move SSB above B_{pa} and to keep fishing mortality sustainable. The EcoQO would simply condense this information into a form that gives an appropriate overview of the overall status of North Sea fish stocks. Care must be taken in the interpretation of this paired EcoQO for two reasons. In the past, some stocks have dropped out of the assessment system when they fell to a very low biomass (e.g., North Sea mackerel). Also, a number of stocks that are fished commercially have been depleted to a fraction of their former abundance (e.g., spurdog (*Squalus acanthias*), thornback ray (*Raja clavata*)) but are not assessed by ICES.

As the value of the metric depends on the stocks included in estimating the percentages, rigorous criteria should be used to determine which stocks should be included and this list should be clearly stated when using the EcoQ metric.

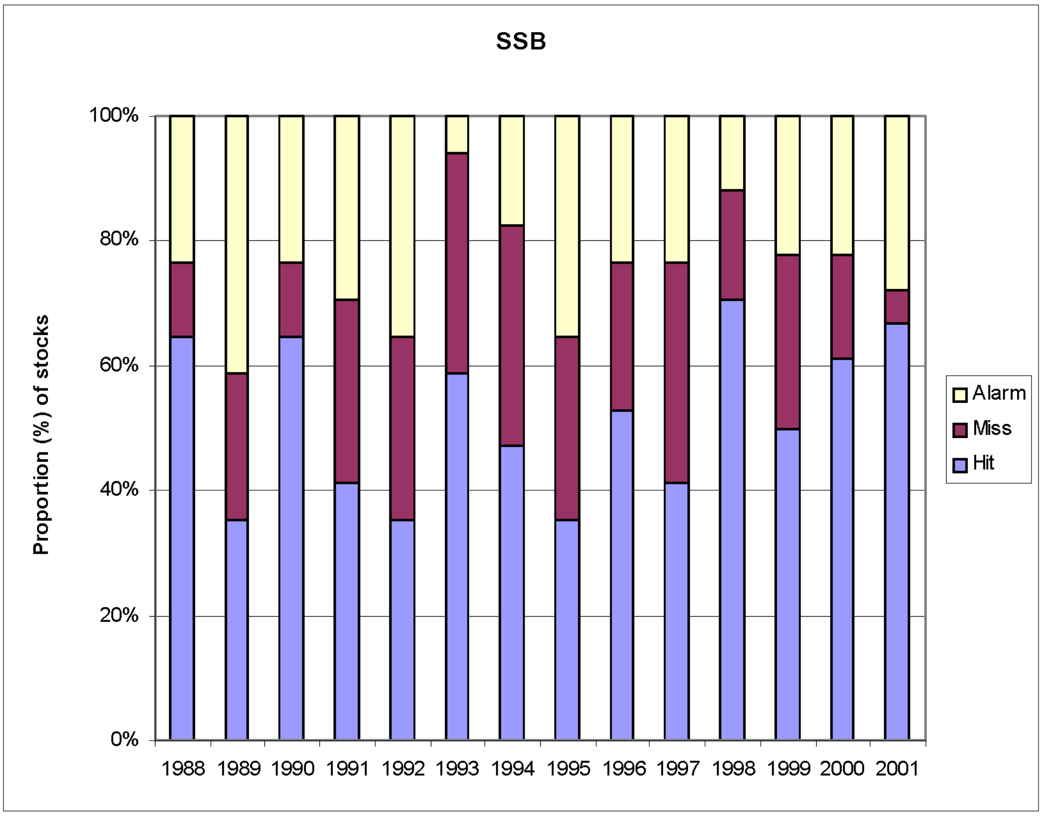


Figure 6.2.4.1. Historic trajectory of the performance of advice based on SSB.

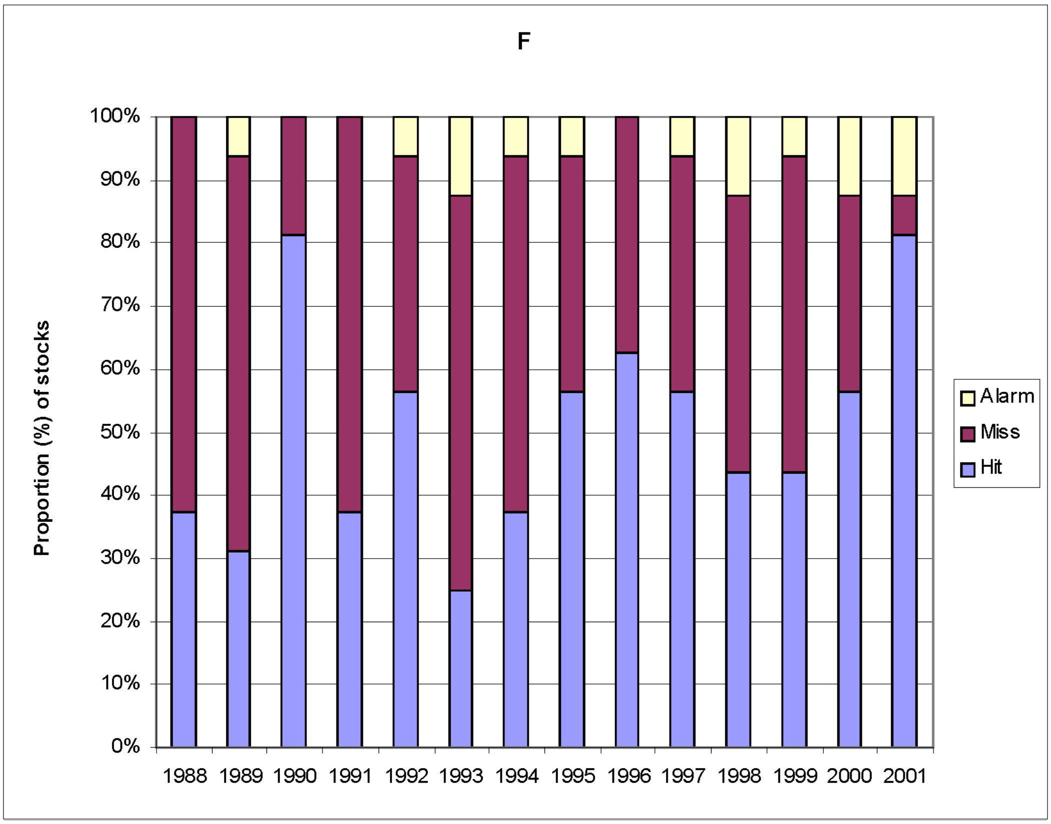


Figure 6.2.4.2. Historic trajectory of the performance of advice based on F.

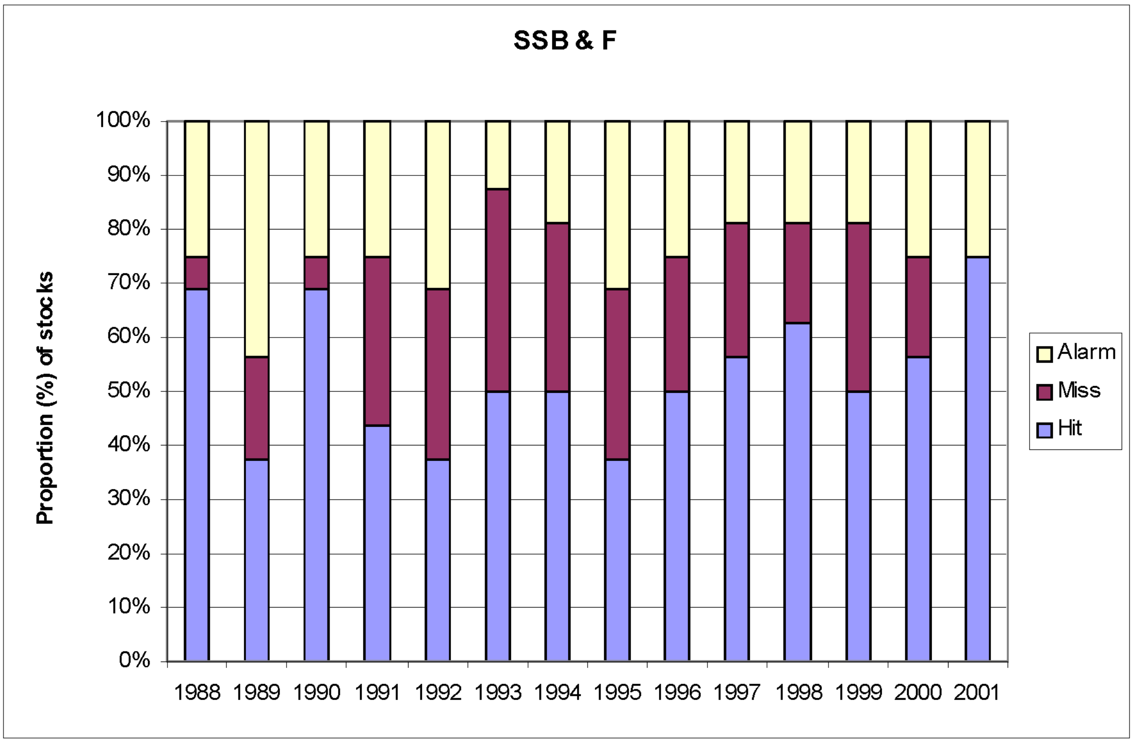


Figure 6.2.4.3. Historic trajectory of the performance of advice based on SSB and F.

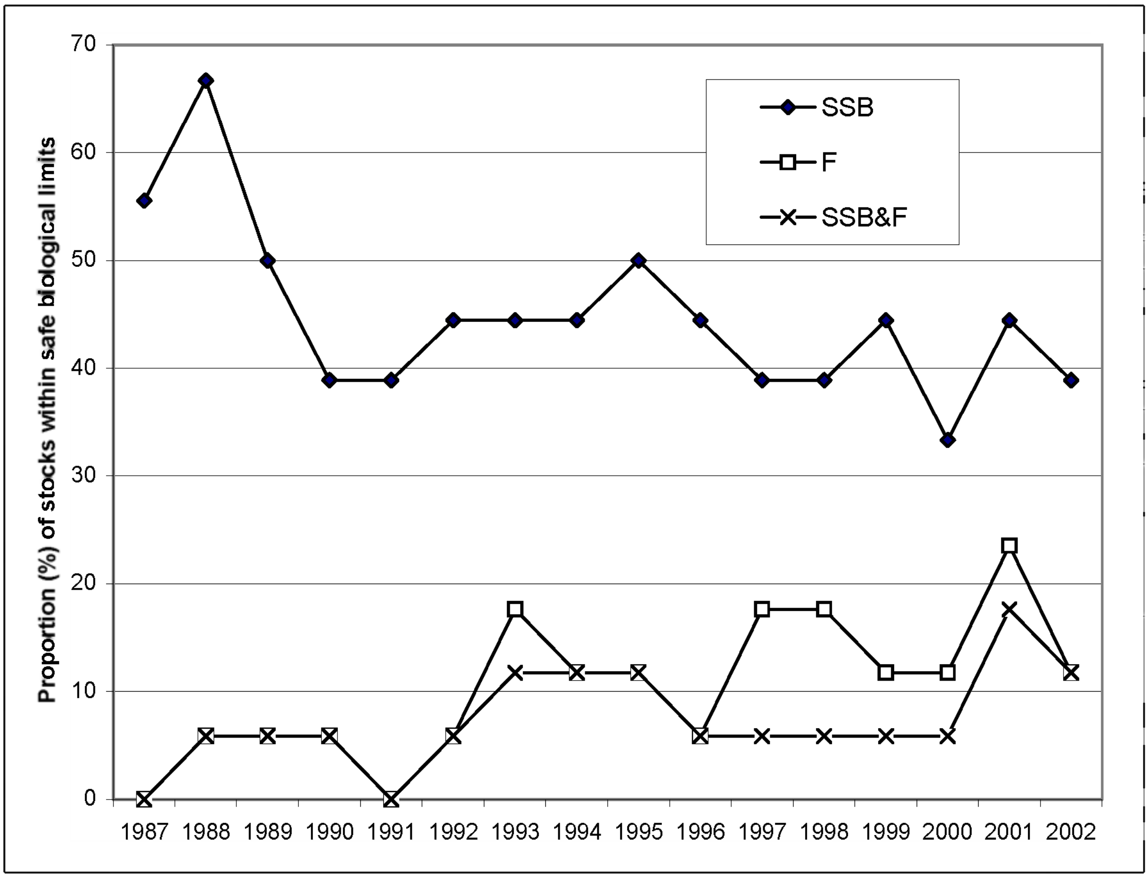


Figure 6.2.4.4. Proportion of stocks within safe biological limits based on SSB, F, and SSB and F.

The ACE evaluation of B_{pa} and F_{pa} as EcoQOs indicates that Misses and False Alarms are about equally frequent. This suggests that advice based on SSB and F will not recommend catch reductions when in fact they are needed for about one stock in five, although the Miss rate seems to be going down in recent years. However, advice based on SSB and F relative to their reference points recommends unnecessary catch reductions about equally often. This symmetry in error rate *de facto* treats both types of errors (Misses and False Alarms) as equally undesirable. In this particular EcoQO, False Alarms are more ecologically precautionary than Misses, so from the perspective of conservation, it would be desirable to reduce the Miss rate. This could be done by choosing different positions for the reference points (higher B_{pa}, lower F_{pa}), but with present knowledge this is likely to increase the False Alarm rate, and advise for unnecessary catch reductions more often. This may not please all customers of the advice. A more effective way to reduce the error rate would be for management to try to maintain stocks at biomasses somewhat above B_{pa} and fishing mortality somewhat below F_{pa}. This was always the intent of ICES, as it repeatedly stresses in its advice that the precautionary reference points should be treated as boundaries on SSB and F, rather than as targets (ICES, 2002).

When considering how the performance of these EcoQOs can be improved, we must also consider the reliability of the assessments themselves. As noted earlier, assessments are known to be imperfect, as there is uncertainty in both the analytical formulations of processes such as natural mortality and age-specific catchability of fleets, as well as in parameter estimates due to sampling error in the input data. Hence, the annual estimates of SSB and F can be expected to have some error (and assessment outputs always provide estimates of the magnitude of estimation errors).

As long as the errors are only variance, they might render estimates of SSB and F less sensitive as indicators for EcoQOs (the Hit rates are only slightly better than 50%), but they should not bias performance. However, if the errors in estimating SSB and F reflect systematic bias, performance might be impaired. This is a real risk because some assessments have been known to suffer from retrospective bias (ICES, 2002), such that successive assessments recalibrate the absolute estimates for a number of past years. The more common pattern is for SSB to be overestimated and fishing mortality to be underestimated for the current and recent past years, consistent with (but not proof of) under-reporting of fish actually killed by the fishery (under-reporting of landings and/or discarding). This bias means that the B_{pa} values from any single assessment may not be perfectly biologically calibrated with SSB estimates from assessments in earlier or later years, i.e., if B_{pa} is taken from an assessment in, say, 1999, it may not reflect perfectly the boundary of safe biological limits on SSB from an assessment in 1996 or 2002. If this problem were common and serious in ICES assessments, however, it should have led to a preponderance of Misses over False Alarms (as is the case for F when used alone for much of the period).

Expert groups under ACFM and the Resource Management Committee are working on several aspects of precautionary reference points, but new estimates of B_{pa} are not yet available. The relatively high Miss rates in the mid-1990s are consistent with the retrospective bias that was found in assessments in the late 1990s, which were strongly influenced by those catches. The relatively high Miss rates during a period when many assessments are now known to have suffered retrospective bias suggest that, with improved data, it may be possible to reduce Misses without increasing False Alarm rates unacceptably.

A Hit rate of only slightly more than 50% might be taken to suggest that B_{pa} and F_{pa} are, in fact, poor EcoQOs. This conclusion would be premature, because they clearly advised correct management action most of the time. Also, a Hit rate of around 50% is, due to uncertainties in point estimates, the expected distribution of advice based on technically correct assessments in those cases where the true population value is close to the reference value to which it is compared. An interpretation of how the distribution of Hits and Misses relates to the actual performance of the management system would require a more detailed understanding of how the distribution has related to the difference between "true" value and reference value over time and how the resulting advice works over time as a control system. Furthermore, knowledge of the actual performance of these EcoQOs in practice only exists because of a long history of use of the indicators. Moreover, this use has been in contexts where their strengths and weaknesses are examined critically on essentially an annual basis, and potential biases, such as the retrospective pattern in assessments, have been discovered and examined. It may be naïve to assume that indicators associated with other EcoQOs, with which the scientific community often has less experience and in far less critical environments, will necessarily be less vulnerable to bias or high variance.

Given that this is the first evaluation of an EcoQO for commercial fish species, it is timely to reiterate previous ICES advice in relation to this EcoQO (ICES, 2001b). Management of fishing effort at levels which deliver a high probability that SSB exceeds B_{pa} for target species is likely to ensure the effective conservation of target species in relation to the objectives of the ecosystem approach to fishery management. However, management to B_{pa} does not ensure complete ecosystem integrity (e.g., it does not address problems of local extirpation of species such as skates and rays, nor local depletions of targeted species) and this caveat must be taken into account when interpreting the EcoQO for commercial fish species.

ACE also reiterates that additional reference points for fish populations should be considered as part of the ecosystem approach to fisheries management (ICES, 1998, 2001b). Those that have been identified as necessary are reference points for:

1) Non-target fish species taken as by-catch or killed by the gear (this includes species that may be targeted in some fisheries but for which ACFM has not determined reference points (ICES, 1997));

2) Ecologically dependent fish species (species that are so tightly linked ecologically to the target species that changes in the abundance/distribution of the target, which do not approach B_{pa}, may still compromise the status of the ecologically dependent species);

3) Genetic health of fish populations (the Convention on Biological Diversity explicitly recognizes the need to conserve genetic diversity).

6.2.7 References

ICES. 1997. Report of the ICES Advisory Committee on Fishery Management, 1997. ICES Cooperative Research Report, No. 223.

ICES. 1998. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 1998/ACFM/ACME:1.

ICES. 2001a. Report of the ICES Advisory Committee on Fishery Management, 2001. ICES Cooperative Research Report, No. 246.

ICES. 2001b. Report of the ICES Advisory Committee on Ecosystems, 2001. ICES Cooperative Research Report, 249: 56–58.

ICES. 2002. Report of the ICES Advisory Committee on Fishery Management, 2002. ICES Cooperative Research Report, No. 255.

6.3 Development of EcoQ element (c) Seal population trends in the North Sea

Source of information

The 2003 Report of the Working Group on Marine Mammal Ecology (WGMME) (ICES CM 2003/ ACE:03).

Summary

For the EcoQ element (c) Seal population trends in the North Sea, the EcoQO is that no seal population in the North Sea should decline by more than 10% over less than ten years. The species covered by this EcoQO are grey seals and harbour seals.

1. Current levels of seals. Initial results suggest that approximately 30–50% of the populations of harbour seals in European waters perished during the 2002 Phocine Distemper Virus epizootic. As not all surveys after the 2002 epizootic have been completed, population estimates for harbour seals in 2002 are not currently available.

2. Historic trends. Time series of abundances from which historic trends can be derived are only available for grey seals and harbour seals for parts of their North Sea distribution. This is a weakness of the EcoQO.

For the populations investigated, in most cases the changes in population estimates of more than 10% between years were only detected one year at a time and were considered to be false alarms. An exception is the decline caused by the phocine distemper virus epizootic of 1988. This was detected in populations and generated appropriate responses (i.e., research into the causes of the epizootic was conducted).

3. Data for assessment on whether the EcoQO is being met. The number of births is a sensitive parameter responding more rapidly than total population size to changes in habitat conditions such as food availability. Pup/adult ratio is probably an indicator that will rapidly identify impaired reproduction in harbour seal populations where populations are surveyed during breeding and moulting seasons.

4. Draft guidelines. ICES does not recommend a specific protocol for the monitoring of seal populations. Phocid seal survey methods, data collection, and analytical techniques vary between regions. ICES recognizes that the compilation of these procedures into a single document will provide a valuable reference on seal census techniques.

5. Management measures. The management strategies for marine mammals applied by most countries in the OSPAR area are oriented towards maintaining or increasing marine mammal populations, so current management strategies are generally appropriate. A “Hit” for this EcoQO triggers further research. The history of the effect of phocine distemper virus on harbour seal populations in European waters suggests that substantial reductions in seal numbers within the space of several months will trigger research in most countries. However, the comprehensiveness of these research programmes varies substantially between countries, from no research at all to detailed studies. It is certain that, despite signing the Bergen Declaration prior to the seal epizootic of 2002, no country initiated research on the basis that the EcoQO was triggered.

Recommendations and advice

ICES recommends as follows:

a) *Current levels.* There is a lack of information on the present population levels of the seal stocks in the North Sea, especially for the harbour seal. It is hoped that this information will be available for the largest part of the North Sea for consideration in 2004.

b) *Data for assessment on whether the EcoQO is being met.* ICES recommends the use of the number of seal births as a sensitive parameter responding more rapidly than total population size to changes in habitat conditions such as food availability.

c) *Draft guidelines.* ICES does not recommend a specific protocol for the monitoring of seal populations.

d) *Management measures.* The management strategies for marine mammals applied by most countries in the OSPAR area are oriented towards maintaining or increasing marine mammal populations, so current management strategies are generally appropriate.

Scientific background

6.3.1 Current levels of seals

Initial results suggest that approximately 30–50% of the populations of harbour seals in European waters perished during the 2002 Phocine Distemper Virus (PDV) epizootic (Figure 6.3.1.1). As not all surveys after the 2002 epizootic have been completed, population estimates for harbour seals in 2002 are not currently available. Abundance estimates will be available for the 2004 meeting of WGMME.

The longer-term consequences of epizootics in European harbour seal populations have been explored in a recent paper (Harding *et al.*, 2002). Immunity played no substantial role in the dynamics of the 2002 epizootic. A stochastic model explored the relationship between mortality, recurrent epizootics, and the long-term growth, fluctuation, and persistence of populations. Given the period between the first and second epizootics, life history parameters of harbour seals, and known anthropogenic mortality, epizootics recurring with the same frequency are unlikely to drive European harbour seal populations to extinction. However, recurrent PDV epizootics of the same observed frequency and severity as those seen to date will reduce the long-term stochastic growth rate of harbour seal populations by approximately 50%. At the observed epizootic frequency, the risk for reduction to 10% of initial population size increased from a negligible 0.001 to a substantial risk of 0.18. Marine wildlife managers need to ensure that estimates of acceptable anthropogenic mortality take this into account.

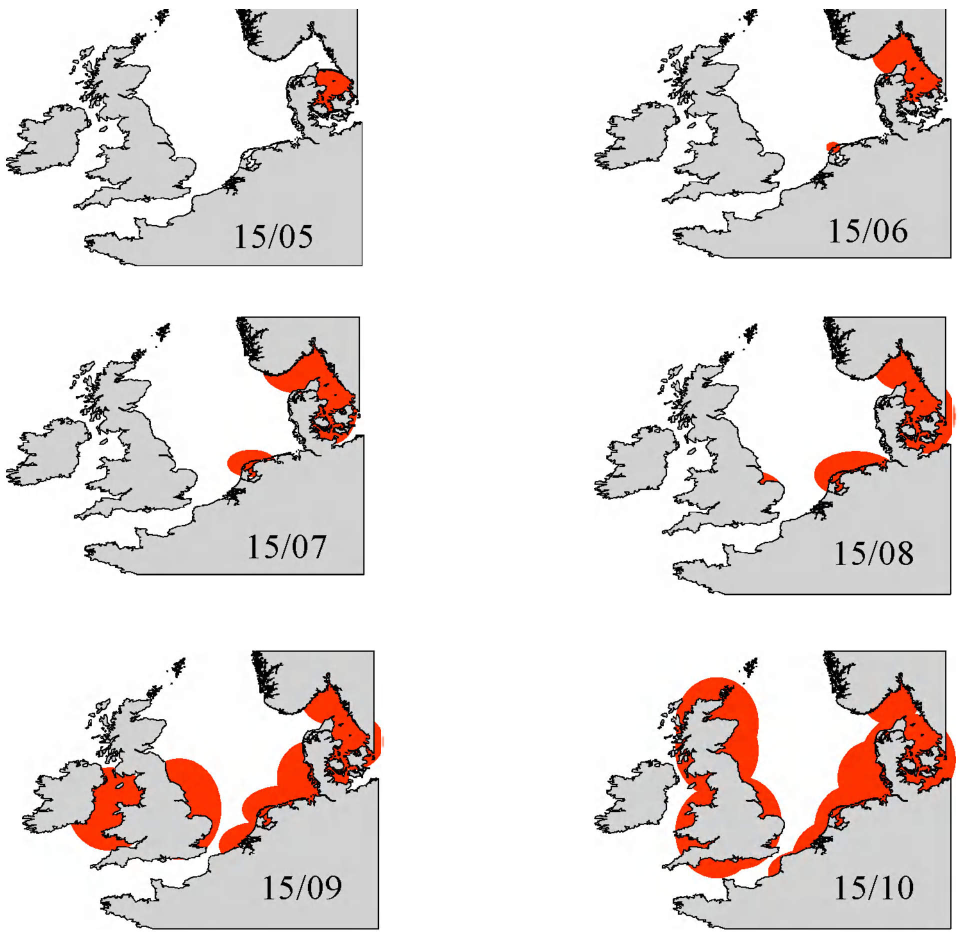


Figure 6.3.1.1. Spread of the PDV epizootic in harbour seals in European waters in 2002. Red shading indicates areas where infected seals were observed on the date given. (From Reijnders, 2003).

The most recent estimates of grey seal abundance in the North Sea that were available to ACE are presented in Table 6.3.1.1. These figures have to be revisited at the 2004 meeting of WGMME.

Table 6.3.1.1. Current estimates of the abundance of grey seals in North Sea waters (from SCOS, 2002).

Region	Year	Estimate of abundance
UK	2001	70,000
Germany	1998	100
The Netherlands	2000	500
Others		?

6.3.2 Historic trends

Data are available on the abundance of harbour seals in most of the North Sea and for grey seals in UK waters and in the Schleswig-Holstein Wadden Sea. These data are discussed below. Time series of abundances are not available for either species for all parts of the North Sea. This is a weakness of the EcoQO.

A working definition for a "False Alarm" for use with this EcoQO is that a False Alarm has occurred when the estimate of seal abundance between two consecutive years has decreased by at least 10%, but that the time series for the years immediately following suggested that the observed "decline" in the year in question was a sampling artefact.

Figure 6.3.2.1 shows the change in estimates of pup production between years based on data for grey seals in UK waters for 1960–2001 (Duck, 2002). The data indicate that on seven occasions, estimates of pup production fell by over 10% between consecutive years during this period (Figure 6.3.2.1). British grey seal populations generally increased over the period for which data are available, so a reduction in pup production between consecutive years of 10% was considered a False Alarm.

ACE did not have access to the confidence intervals associated with the data points for this time series. Therefore, ACE was unable to determine whether these False Alarms would remain if the uncertainty associated with each point estimate for each year was included in the calculation.

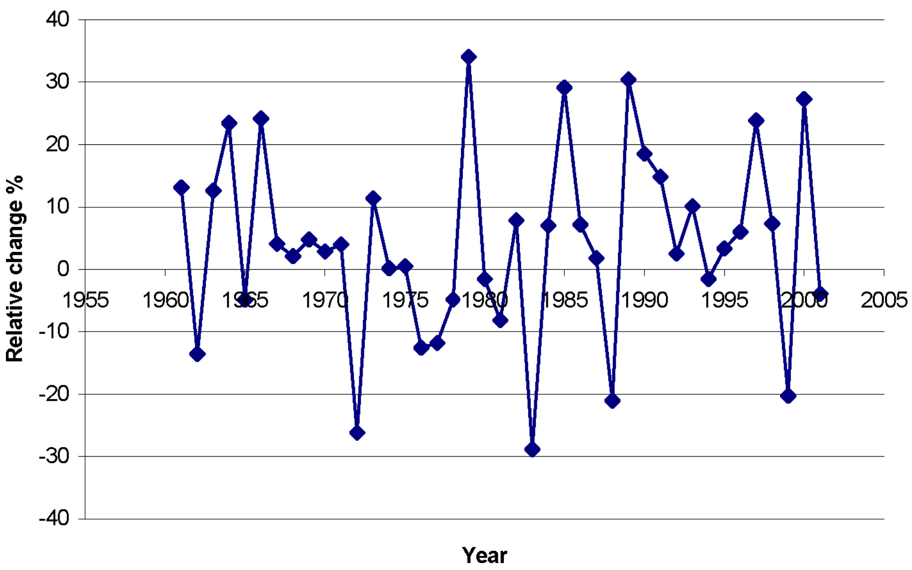


Figure 6.3.2.1. Time series of annual changes in estimates of grey seal pup production at major UK breeding sites in the North Sea, except Helmsdale, Orkney, and Shetland (extracted from Duck, 2002)

Figure 6.3.2.2 shows data for harbour seals in the Danish Straits and Skagerrak for 1988–1998 (Härkönen *et al.*, 2002). The figure shows the change in estimates of pup production between years. If these data can be considered "historic", they indicate one occasion when estimates of abundance fell by over 10% between consecutive years over this period. These harbour seal populations generally increased over the period for which data are available, so this observed reduction between consecutive years of 10% was considered a False Alarm for which data are available. ACE did not have access to the confidence intervals associated with the data points for this time series. Therefore, ACE was unable to determine whether this False Alarm would remain if the uncertainty associated with each point estimate for each year was included in the calculation.

Table 6.3.2.1 shows data for grey seals in the Schleswig-Holstein Wadden Sea waters for 1989–2000. If these data can be considered "historic", they indicate that there would be no Hits, Misses, or False Alarms for these animals.

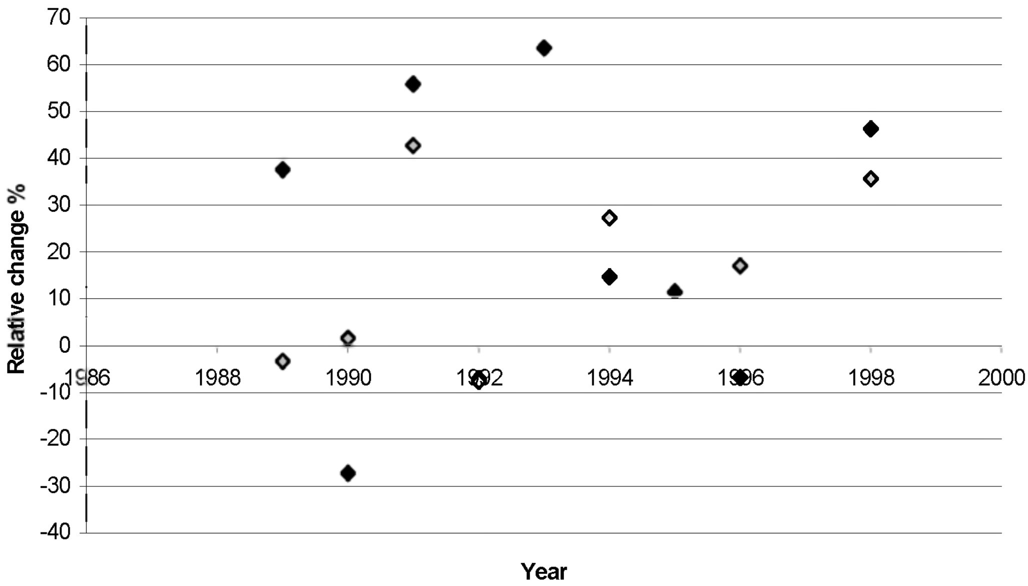


Figure 6.3.2.2. Time series of annual changes in harbour seal abundance from the Danish Straits and Skagerrak, 1988–1998. Extracted from Härkönen *et al.* (2002).

Table 6.3.2.1. Time series of the recorded numbers of grey seals in the Schleswig-Holstein Wadden Sea. Extracted from http://www.waddensea-secretariat.org.

Breeding season	Counted live pups; Number of births	Dead pups	Adults counted during the breeding season	Adults counted in spring
1988/1989	9	0	16	26
1989/1990	3	1	20	51
1990/1991	7	1	10	47
1991/1992	6	1	13	57
1992/1993	10	1	28	54
1993/1994	7	3	12	56
1994/1995	5	2	7	88
1995/1996	11	3	17	53
1996/1997	11	4	14	73
1997/1998	9	0	18	100
1998/1999	11	2	19	-
1999/2000	13	3	?	?

Table 6.3.2.2 shows the indices of abundance of harbour seals at The Wash, UK, for 1988–2001 (Duck and Thompson, 2002). If these data can be considered "historic", they indicate that a Hit would have occurred correctly in response to the phocine distemper virus epizootic of 1988. This "Hit" did indeed occur and generated an appropriate response (i.e., research into the causes of the epizootic was conducted). Table 6.3.2.3 shows the indices of abundance of harbour seals in the German Wadden Sea, 1975–2001. If these data can be considered "historic", they indicate that a Hit would have occurred correctly in response to the phocine distemper virus epizootic of 1988. This "Hit" did indeed occur and generated an appropriate response (research into the causes of the epizootic was conducted).

Evaluating the relative importance of Hits, Misses, and False Alarms requires a trade-off between Type I and Type II errors. This is an area of interaction between science and policy. Clearly it is not the responsibility of ICES to decide policy, so further interaction with policy-makers is required to develop this area of EcoQOs. It is not clear from the request what consideration should be given to trade-offs between these errors.

Changes in seal numbers in individual colonies can be caused by migration between colonies or the establishment of new colonies. It is therefore important to take this into account when considering rates of change in the populations.

Table 6.3.2.2. Counts of harbour seals in The Wash, UK. Data from Duck (pers. comm.) and Annex III of Scientific advice on matters related to the management of seal populations 2002 (SCOS, 2002).

Year	Count
1968	1468
1969	1722
1969	1473
1970	1662
1972	1632
1978	2186
1978	2176
1980	2191
1988	3087
1989	1531
1990	1532
1991	1226
1992	1724
1993	1759
1994	2277
1995	2266
1996	2151
1997	2561
1998	2367
1999	2320
2000	2528
2001	3194

Table 6.3.2.3. Time series of counts of harbour seals from the German Wadden Sea (Abt, pers. comm. to Scheidat).

Year	Neider Sachsen	Schleswig Holstein	Wadden Sea Total
1975	1,049	1,749	3,492
1976	1,163	1,682	3,526
1977	1,140	1,741	3,622
1978	1,228	1,712	3,620
1979	1,109	1,856	3,745
1980	1,298	2,025	4,410
1981	1,441	2,200	4,672
1982	1,543		5,247
1983	1,777		5,851
1984	1,936	3,300	6,249
1985	2,062		6,878
1986	2,272		7,740
1987	2,400	3,986	8,790
1988	2,508	4,124	9,800
1989	1,401	1,685	4,355
1990	1,620	1,930	5,005
1991	1,924	2,304	5,921
1992	2,255	2,792	6,988
1993	2,482	3,269	8,107
1994	3,111	3,266	8,916
1995	3,214	3,745	9,761
1996	3,529	4,537	11,013
1997	4,319	5,003	12,927
1998	4,588	5,568	14,446
1999	4,809	6,134	15,244
2000	5,233	6,700	17,008
2001	6,223	7,534	19,387
2002	6,481	7,876	20,975

6.3.3 Data for assessment on whether the EcoQO is being met

The number of births is a sensitive parameter responding more rapidly than total population size to changes in habitat conditions such as food availability. Pup/adult ratio is probably an indicator that will rapidly identify impaired reproduction in harbour seal populations where populations are surveyed during breeding and moulting seasons.

A strong assumption behind this EcoQO is that, in the absence of major mortality incidents, real population declines of greater than 5 % per year would be unusual in seal populations at or below carrying capacity levels and these declines would be detected reliably in surveys of either adults or newborn pups. It also assumes that observed declines between years are real. On a short-term scale, seal population size may not be the parameter most sensitive to environmental change. Due to the longevity and delayed maturity of seals, several years are usually needed before changes in their reproduction or immature survival rates affect their breeding numbers. Substantial increases in adult mortality would have a more immediate effect. Nevertheless, rates of change in population sizes are reasonably good indicators of important changes in seal populations, where density-dependent effects may easily reduce the usability of other population parameters such as absolute size.

6.3.4 Draft guidelines

ACE does not recommend a specific protocol for the monitoring of seal populations. Phocid seal survey methods, data collection, and analytical techniques vary between regions. ACE recognizes that the compilation of these procedures into a single document will provide a valuable reference on seal census techniques.

Monitoring protocols for marine mammals involve trade-offs between cost, resourcing, appropriate spatial-temporal survey coverage, and the appropriate survey technique, which is affected by behaviour, and current population status. Initial discussions on the development of this EcoQO (ICES, 2001) identified the value of this EcoQO specifically because most countries in the OSPAR area have appropriate monitoring programmes in place.

WGMME reviewed census techniques for land-breeding phocid seals, and statistical analyses of resulting data (including correction factors). The review identified a general similarity in harbour seal survey techniques. In contrast, two different types of methods are used for the grey seal: one method for the Baltic Sea and another type of method most commonly used elsewhere.

Harbour seals

The standard methodology for estimating harbour seal population size is by aerial surveys of haul-out sites during the pupping or moulting periods when a larger fraction of seals are hauled out (Gilbert and Wynne, 1988; Heide-Jørgensen and Härkönen, 1988; Thompson and Harwood, 1990; Stobo and Fowler, 1994; Gilbert and Guldager, 1998; Reijnders *et al.*, 1997; Frost *et al.*, 1999; Huber *et al.*, 2001; Jeffries *et al.*, 2003). A good knowledge of the behaviour, pupping and moulting of the seal, and the effect of environmental factors on them, is necessary to chose the proper survey time and for corrections for the fraction of seals not hauled-out during survey operations.

Grey seals

The most common method for estimating grey seal populations is to use counts of pups during the breeding season. This is done by a series of aerial surveys, using photography, through the course of the breeding season. Individual breeding colonies are photographed several times per year. The total number of grey seals is estimated by applying a scaling factor to the pup counts (Duck, 2002). In the Baltic region, grey seal surveys are conducted annually during the moulting period (as the pups are not easily discernable on the ice where they stay after pupping) (Jüssi and Jüssi, 2001; Helander and Karlsson, 2002).

6.3.5 Management measures

The management strategies for marine mammals applied by most countries in the OSPAR area are oriented towards maintaining or increasing marine mammal populations, so current management strategies are generally appropriate. A "Hit" for this EcoQO triggers further research. The history of the effect of phocine distemper virus on harbour seal populations in European waters suggests that substantial reductions in seal numbers within the space of several months will trigger research in most countries. However, the comprehensiveness of these research programmes varies substantially between countries, from no research at all to detailed studies. It is certain that, despite signing the Bergen Declaration prior to the seal epizootic of 2002, no country initiated research on the basis that the EcoQO was triggered.

Recent changes in the Norwegian management of grey and harbour seals in Norwegian waters appear aimed at achieving substantial reductions in the populations of these animals. This includes seals in the Norwegian sector of the North Sea. If these aims are achieved, i.e., hunters fill available seal quotas, this management strategy will trigger this EcoQO. The management measure required to reverse this is simply to return to the protocols used prior to 2003 for setting quotas.

However, the revised quotas established by the Norwegian government demonstrate a failure in the process of implementation of accepted pilot EcoQOs. There are only two EcoQOs adopted in the Bergen Declaration that deal specifically with marine mammals.

The Norwegian government, a signatory to the Bergen Declaration, then instituted a management approach, the aim of which is clearly not to achieve the objective of the EcoQO. ACE noted that this development indicates that the essentials of national responsibility under the Bergen Declaration do not appear to have been communicated to relevant line managers, in at least one country. ACE was left pondering whether countries are taking EcoQOs seriously.

6.3.6 References

Duck, C. 2002. Pup production in the British grey seal population. Annex II in Scientific advice on matters relating to British seal populations: 2002. Available through: http://www.smru.st-and.ac.uk.

Duck, C., and Thompson, D. 2002. The status of British common seal populations. Annex III in Scientific advice on matters relating to British seal populations: 2002. Available through: http://www.smru.st-and.ac.uk.

Frost, K.J., Lowry, L.F., and Ver Hoef, J.M. 1999. Monitoring the trend of harbor seals in Prince William Sound, Alaska, after the *Exxon Valdez* oil spill. Marine Mammal Science, 15: 494–506.

Gilbert, J.R., and Guldager, N. 1998. Status of harbor and gray seal populations in northern New England. NMFS/NER Cooperative Agreement 14-16-009-1557. Woods Hole, Massachusetts. Available from NOAA Fisheries, NEFSC, 166 Water St., Woods Hole, Massachusetts 02543, USA.

Gilbert, J.R., and Wynne, K.M. 1988. Status of harbor and gray seal populations in northern New England. Final report under NMFS/NER Cooperative Agreement 14-16-009-1557, Woods Hole, MA. Available from NOAA Fisheries, NEFSC, 166 Water St., Woods Hole, MA 02543, USA.

Harding, J.C., Härkönen, T., and Caswell, H. 2002. The 2002 European seal plague: epidemiology and population consequences. Ecology Letters, 5: 727–732.

Härkönen. T., Harding, K., and Heide-Jørgensen, M.-P. 2002. Rates of increase in age-structured populations: a lesson from the European harbour seals. Canadian Journal of Zoology, 80: 1498–1510.

Heide-Jørgensen, M.-P., and Härkönen, T. 1988. Rebuilding seal stocks in the Kattegat-Skagerrak. Marine Mammal Science, 4: 231–246.

Helander, B., and Karlsson, O. 2002. Inventering av gråsæl vid svenska Östersjö kusten. Naturhistoriska Riksmuseet, Stockholm, Sweden. Sælinformation 2002:1. (In Swedish).

Huber, H.R., Jeffries, S.J., Brown, R.F., DeLong, R.L., and VanBlaricom, G. 2001. Correcting aerial survey counts of harbor seals (*Phoca vitulina richardsi*) in Washington and Oregon. Marine Mammal Science, 17: 276–293.

ICES. 2001. Report of the Advisory Committee on Ecosystems, 2001. ICES Cooperative Research Report, 249: 27–31.

Jeffries, S., Huber, H., Calambokidis, J., and Laake, J. 2003. Trends and status of harbour seals in Washington State: 1978–1999. Journal of Wildlife Management, 67: 208–219.

Jüssi, I., and Jüssi, M. 2001. Action plan for grey seals in Estonia 2001–2005. Eesti ulukid/Estonian Game, 7. 88 pp.

Reijnders, P.J.H. 2003. Newsletter of the Wadden Sea Secretariat. Available at http:// www.waddensea-secretariat.org.

Reijnders, P.J.H., Ries, E.H., Tougaard, S., Norgaard, N., Heidemann, G., Schwartz, J., Vareshi, E., and Traut, I.M. 1997. Population development of harbor seals Phoca vitulina in the Wadden Sea after the 1988 virus epizootic. Journal of Sea Research, 38: 161–168.

SCOS. 2002. Special Committee on Seals. Annex I in Scientific advice on matters relating to British seal populations: 2002. SCOS 02/2. Available through: http://www.smru.st-and.ac.uk.

Stobo, W.T., and Fowler, G.M. 1994. Aerial surveys of seals in the Bay of Fundy and off southwest Nova Scotia. Canadian Technical Report of Fisheries and Aquatic Sciences, 1943. 57 pp.

Thompson, P.M., and Harwood, J. 1990. Methods for estimating the population size of common seals, *Phoca vitulina*. Journal of Applied Ecology, 27: 924–938.

6.4 Development of EcoQ element (e) By-catch of harbour porpoise in the North Sea

Source of information

The 2003 Report of the Working Group on Marine Mammal Ecology (WGMME) (ICES CM 2003/ ACE:03).

Summary

In the Danish North Sea fisheries alone, the extrapolated annual by-catch was about 4,000 individuals in 1999–2001. In the recent past, this figure has been more than 7,000 per year. In addition to this, UK fisheries took in the order of 800 individuals in 1995, and 440 individuals in 1999. Total fishery by-catch cannot be evaluated because other fisheries (in particular, Norwegian fisheries) are not yet monitored for by-catch. The above decline since 1994 in the by-catch levels of Danish and UK fisheries occurred as a result of reduced fishing efforts. It is likely that this trend will continue with the major quota and effort reductions for the cod fisheries.

There is only one estimate of harbour porpoise abundance in the North Sea. This was made in 1994 under the SCANS project. The SCANS survey covered most of OSPAR Region II and estimated a harbour porpoise population of 300,000 (CV≈ 0.14) for this area. The abundance estimate during SCANS only gave an idea of the summer distribution. There clearly are migrations and it is dangerous to base possible management measures to reduce by-catch on the harbour porpoise distribution observed during the SCANS I (and the future SCANS II) survey. Therefore, data on seasonal distribution and abundance also need to be taken into account, in addition to the data on abundance from the SCANS surveys.

The population structure of the harbour porpoise in the North Sea is not well known, however, there is likely to be some structuring. Genetic studies indicate differences between porpoises in the northwestern North Sea and those in the southern North Sea, and between them and those on the Celtic Shelf (western Channel – part of the North Sea in the current context). There is likely to be further subdivision of the population in the waters surrounding Jutland. However, studies are at present inconclusive and it is likely that any "population boundaries" that exist will not be fixed in space or time.

In the absence of clear population boundaries, a practical approach has to be taken with respect to the geographical basis for evaluating progress. Such a basis could be aggregated SCANS sampling sub-areas. The number of sub-areas should, however, be kept small, e.g., three or four, to minimize sampling variance of fisheries effort, and by-catch and population estimates.

The only method to acquire reliable data for the majority of the fishing fleets is through the use of independent on-board observers. A major issue is that of scaling from on-board observations to fleet scale. In the North Sea, reasonable estimates exist for most fisheries relevant to harbour porpoise by-catch, with the notable exception of any way of estimating by-catch within the Norwegian small boat fisheries.

The historic trends in by-catch are unknown, and cannot be reconstructed. In theory, estimates might be made on the basis of fishing effort, but such information is also not available except in the years from 1990 onwards. It would not be possible to assess by-catch rates as there is no information at present on trends in the abundance of harbour porpoise. An evaluation of the historic performance of this EcoQO therefore does not seem possible.

Scientific background

6.4.1 Baselines against which progress can be measured

6.4.1.1 Current levels of by-catch

Drift-net fisheries

The UK has several small drift-net fisheries. Observations have been made on two of these (with relatively low proportionate effort) and no by-catch has been observed. By-catches of harbour porpoises in a Norwegian drift-net fishery for salmon were examined in 1988. A financial reward was offered to fishermen to return porpoises to port for post-mortem examinations. Catch rates were among the highest ever recorded for a marine mammal in a net fishery, at around 0.65–1.47 porpoises/km.hour of fishing effort (Bjørge and Øien, 1995). This fishery was closed after the 1998 fishing season, mainly for reasons of salmon conservation.

Table 6.4.1.1.1. Estimates of harbour porpoise by-catch by fishery and season (quarter of year) for Danish bottom-set gillnet fishing in the North Sea (Vinther and Larsen, 2002).

Fishery	Season	1987	1988	1989	1990	1991	1992	1993	1994
Cod, wreck	1,2 and 4	97	99	89	104	102	117	116	123
	3	276	405	383	173	291	386	606	555
Cod, other	1 and 3	1,410	1,342	1,217	919	1,076	1,307	1,603	1,578
	2 and 4	236	323	294	401	386	443	428	456
Hake	All	119	160	212	268	405	541	697	493
Turbot	2 and 3	2,719	3,229	2,547	3,067	3,033	2,577	2,245	2,534
Plaice	All	465	380	231	260	1,018	1,172	1,014	1,627
Sole	All	0	0	0	0	0	0	0	0
All	All	5,322	5,938	4,973	5,191	6,312	6,543	6,709	7,366

Fishery	Season	1995	1996	1997	1998	1999	2000	2001	Mean
Cod, wreck	1,2 and 4	117	121	130	148	126	106	67	111
	3	568	475	587	738	511	570*	405*	462
Cod, other	1 and 3	1,546	1,472	1,514	1,943	1,705	1,420	950	1,400
	2 and 4	435	445	538	565	411	413	261	402
Hake	All	381	189	119	142	217	181	158	285
Turbot	2 and 3	2,366	1,999	1,402	1,034	737	985	1,144	2,108
Plaice	All	1,325	1,292	1,018	636	521	475	903	822
Sole	All	0	0	0	0	0	0	0	0
All	All	6,737	5,991	5,308	5,206	4,227	4,149	3,887	5,591

* By-catch in this fishery is overestimated, as the effect of the mandatory use of pingers has not been taken into account.

Set-net fisheries

Extrapolation of observed by-catch rates to fleet level in the Danish bottom-set fisheries in the North Sea gives an annual by-catch between 3,900 and 7,400 porpoises in the period 1987 to 2001 (Table 6.4.1.1.1).

Larsen *et al.* (2002) demonstrated a complete elimination of observed by-catch in the Danish North Sea wreck gillnet fishery in the third quarter of the year due to the deployment of pingers. The by-catch estimates in Table 6.4.1.1.1 were made without considering the number of animals likely to have been saved by the use of pingers. Assuming 100% effectiveness, these would have amounted to 570 animals in 2000 and 405 in 2001.

Vinther (1999) reported observations of 329 net km.days between 1995 and 1998 on Danish set-net fisheries in the Kattegat and Skagerrak. A total of five porpoises were observed as by-catch in one ICES rectangle; four of these were caught in the lumpfish fishery. This equates to fifteen animals by-caught per 1,000 net km.days.

Harbour porpoise by-catch has been estimated for UK fisheries for cod, sole, ray, and turbot in the North Sea (Table 6.4.1.1.2) for the period 1995–1999. The by-catch halved during this period as fishing effort (measured in days at sea) declined. By-catch estimates were based on observed by-catch per day at sea within metier, on the assumption that mean effort per day at sea among sampled vessels was an unbiased estimate of mean effort per day at sea for the entire metier.

Table 6.4.1.1.2. Estimates of harbour porpoise by-catch in the North Sea (CEC, 2002). These estimates are for cod, sole, ray, and turbot set-net fisheries and are derived from individual estimates for each of the fisheries in each area.

Year	North Sea	95% confidence interval
1995	818	674–1,233
1996	624	500–959
1997	627	513–957
1998	490	383–769
1999	436	351–684

Studies on the by-catch of harbour porpoises in set-net fisheries were conducted for the Swedish cod and pollack fisheries in the Kattegat in 1996–1997 (Harwood *et al.*, 1999). A total of 7,441 net km.hrs were observed over three seasons of the year in two ICES rectangles on the Skagerrak/Kattegat boundary. A total of twelve porpoises were observed as by-catch, while a further thirteen animals were reported as by-catch on unobserved vessels fishing in the same rectangles. Based on these figures, these authors extrapolated a by-catch of 105 animals per 10,000 net km.hrs in the Skagerrak/Kattegat combined. The Swedish fisheries targeting cod and pollack decreased by 59% between 1997 and 2000 due to the reduction in the stock size of cod. Swedish gillnet fisheries in the Kattegat and Skagerrak also target flounder, crabs, dogfish, pollack, sole, turbot, and herring. Overall, Swedish set-net efforts have declined greatly in recent years.

There is no programme established to monitor cetacean by-catch in Norwegian set-net fisheries, nor is there any information on fishing effort that might be used to provide an estimate of by-catch. However, there are a number of harbour porpoises taken per year in coastal gillnet fisheries (carcasses are periodically collected for biological studies). This by-catch may be substantial. The scale of harbour porpoise by-catch in the Norwegian offshore gillnet fisheries is unknown.

No by-catch of harbour porpoises has been observed in German set-net fisheries (Kock, 1997), although a project was started in 2002 to investigate possible by-catch. On the basis of evidence from stranded corpses, there were 23 known by-catches from German waters between 1987 and 1995, mostly from around the Island of Sylt. For Schleswig-Holstein there was one recorded by-catch in 2002.

No information is available on harbour porpoise by-catch in the Dutch or French fisheries. There is a low level of fishing effort by Dutch gillnetters, with no records of marine mammal by-catch from these few vessels. About four Dutch vessels are reported to be working gillnets on a regular basis, and a few others on an irregular basis. Effort data are lacking.

6.4.1.2 Current level of population abundance

There is only one estimate of harbour porpoise abundance in the North Sea. This was made in 1994 under the SCANS project (Hammond *et al.*, 2002). The SCANS survey estimated a harbour porpoise population of 300,000 (CV $\approx$ 0.14) in OSPAR Region II (approximately the SCANS survey area minus SCANS sub-area A, I', X, and K).

The population structure of the harbour porpoise in the North Sea is not well known, however, there is likely to be some structuring (Tolley *et al.*, 1999). Genetic studies indicate differences between porpoises in the northwestern North Sea and those in the southern North Sea, and between them and those on the Celtic Shelf (western Channel – part of the North Sea in the current context). There is likely to be further subdivision of the population in the waters surrounding Jutland. Studies are at present inconclusive and it is likely that any "population boundaries" that exist will not be fixed in space or time.

6.4.2 Historic performance of the EcoQO

The historic trends in by-catch are unknown, and cannot be reconstructed. In theory, estimates might be made on the basis of fishing effort, but such information is also not available except in the years from 1990 onwards. It would not be possible to assess by-catch rates as there is no information at present on trends in the abundance of harbour porpoise. An evaluation of the historic performance of this EcoQO, therefore, does not seem possible.

6.4.3 Information for future assessment of the EcoQO and draft guidelines for monitoring and evaluating the status of, and compliance with, the EcoQO

The main types of information required to assess by-catch rates of harbour porpoises in the North Sea are population abundance estimates and structure, and the scale and geographical distribution of the by-catch.

The SCANS survey in 1994 has provided the only estimate of harbour porpoise abundance in the North Sea. The line-transect methods used in that survey were the best available at that time. A second abundance survey is planned for 2004–2005 that will use the same line-transect techniques as in 1994. Techniques have, however, advanced in scaling from transect data to abundance estimate, primarily through the use of GIS and post-hoc sampling. These techniques are still being refined. It is likely that the 1994 survey data will be revisited to compare with the new survey data.

The abundance estimate during SCANS only gave an idea of the summer distribution. There clearly are migrations and it is dangerous to base possible management measures to reduce by-catch on the harbour porpoise distribution observed during the SCANS I (and the future SCANS II) survey. Therefore, data on seasonal distribution and abundance also need to be taken into account, in addition to the data on abundance from the SCANS surveys.

In the absence of clear population boundaries, a practical approach has to be taken with respect to the geographical basis for evaluating progress. Such a basis could be aggregated SCANS sampling sub-areas. The number of sub-areas should, however, be kept small, for example, three or four, to minimize sampling variance on fisheries effort, and by-catch and population estimates.

Northridge (1996) reviewed methods to assess the by-catch of cetaceans. The recommendations of this report have been followed for most by-catch estimations in the North Sea in recent years, and it is suggested that these methods continue to be followed. In summary, the only method to acquire reliable data for the majority of the fishing fleets is through the use of independent on-board observers. It is difficult, but not impossible, to estimate by-catches in fisheries conducted from smaller vessels. A major issue is that of scaling from on-board observations to fleet scale. In the North Sea, reasonable estimates exist for most fisheries relevant to harbour porpoise by-catch, with the notable exception of any way of estimating by-catch within the Norwegian small boat fisheries; even the scale of these fisheries remains completely unknown. By-catch will need to be estimated, preferably using effort information, within each of the major relevant fisheries of the North Sea and individual methods will need to be derived from the above guidance for each fishery.

6.4.4 Management measures to help meet the EcoQO

The 2002 ACE report (ICES, 2002) gives a review of and advice on management measures to reduce small cetacean by-catch. This review includes an evaluation of overall effort reduction, closed areas, use of pingers in fixed gears, gear modifications of pelagic trawls, and other mitigation measures. The advice was given as a response to a request from the European Commission; however, it is relevant for this OSPAR request as well.

6.4.5 References

Bjørge, A., and Øien, N. 1995. Distribution and abundance of harbour porpoise *Phocoena phocoena* in Norwegian waters. Reports of the International Whaling Commission, Special Issue Series, 16: 89–98.

CEC. 2002. Incidental catches of small cetaceans. Report of the meeting of the Subgroup on Fishery and the Environment (SGFEN) of the Scientific, Technical and Economic Committee for Fisheries (STECF), Brussels, 10–14 December 2001. SEC (2002) 376.

Hammond, P.S., Berggren, P., Benke, H., Borchers, D.L., Collet, A., Heide-Jørgensen, M.P., Heimlich, S., Hiby, A.R., Leopold, M.F., and Øien, N. 2002. Abundance of the harbour porpoise and other cetaceans in the North Sea and adjacent waters. Journal of Applied Ecology, 39: 361–376.

Harwood, J., Andersen, L.W., Berggren, P., Carlström, J., Kinze, C.C., McGlade, J., Metuzals, K., Larsen, F., Lockyer, C.H., Northridge, S., Rogan, E., Walton, M., and Vinther, M. 1999. Assessment and reduction of the by-catch of small cetaceans (BY-CARE). Final report to the European Commission on FAIR-CT05-0523.

ICES. 2002. Report of the ICES Advisory Committee on Ecosystems, 2002. ICES Cooperative Research Report, 254: 2–17.

Kock, K.-H. 1997. The by-catch of small cetaceans in the German fisheries in the North Sea and the Baltic in 1996 – preliminary results. Paper presented to the Scientific Committee of the International Whaling Commission, 1997, SC/49/SM21.

Larsen, F., Vinther, M., and Krog, C. 2002. Use of pingers in the Danish North Sea wreck net fishery. Paper presented to the Scientific Committee of the International Whaling Commission, Shimonoseki, May 2002, SC/54/SM32. 7 pp.

Northridge, S.P. 1996. A review of marine mammal by-catch observer schemes with recommendations for best practice. Joint Nature Conservation

Committee. (JNCC) Report No. 219, Aberdeen, UK.

Tolley, K.A., Rosel, P.E., Walton, M., Bjørge, A., and Øien, N. 1999. Genetic population structure of harbour porpoise (*Phocoena phocoena*) in the North Sea and Norwegian waters. Journal of Cetacean Research and Management, 1(3): 265–274.

Vinther, M. 1999. By-catches of harbour porpoises (*Phocoena phocoena* L.) in Danish set-net fisheries. Journal of Cetacean Research and Management, 1: 123–135.

Vinther, M., and Larsen, F. 2002. Updated estimates of harbour porpoise by-catch in the Danish bottom set gillnet fishery. Paper presented to the Scientific Committee of the International Whaling Commission, Shimonoseki, May 2002, SC/54/SM31. 13 pp.

6.5 Development of EcoQ element (f) Proportion of oiled common guillemots among those found dead or dying on beaches

Source of information

The 2003 Report of the Working Group on Seabird Ecology (WGSE) (ICES CM 2003/C:03).

Summary

There are considerable quantities of existing data on the proportion of oiled seabirds beached around the North Sea. In those areas where these data have been analysed, there is evidence of a decline in the proportions oiled. An analysis of this evidence indicated that it would not be possible to ensure that a target (the EcoQO is 10%) had been met without several years of data. As a consequence, ICES recommends that a period of at least five years over which an average of 10% oiled common guillemots has been recorded should occur before the conclusion that the objective has been reached could be justified statistically. Guidelines for monitoring oiled birds have been developed both within OSPAR and in the Trilateral Monitoring Programme for the Wadden Sea. These guidelines are suitable for use in the whole North Sea. Resources (personnel, funds) will be required to ensure that sufficient sampling and coordination occur at both the national and international levels. Possible management measures to reduce levels of at-sea oiling, and a case study of one instance of successful management are described.

Recommendations and advice

1) ICES advises OSPAR to request the lead country for this EcoQO (The Netherlands) to reanalyse the oiled seabird data on the basis of the geographical boundaries suggested in Section 6.5.4.1, below, to examine trends in oiling and consistency in patterns between adjacent regions. This will also allow baselines to be set for the suggested regions.

2) ICES advises that a national coordinator for beached bird counts is required in each country around the North Sea, and that one international coordinator should be appointed.

3) ICES advises that a period of at least five years in which an average of 10% oiled common guillemots has been recorded should occur before the conclusion that the objective has been reached could be justified statistically. OSPAR might wish to modify the description of the EcoQO to take account of this.

4) ICES advises that the information provided by this EcoQO would be enhanced by analysing samples of relevant pollutants taken from the plumage of seabird corpses. These samples can help indicate generic (and sometimes specific) sources of pollutants, thus enabling management actions to be appropriately targeted.

5) ICES advises that the provision of port waste reception facilities, improved detection of illegal behaviour, and prosecution (and punishment) of offenders have been shown to be suitable management measures to reduce amounts of oil discharged to the sea.

Scientific background

6.5.1 Introduction

For EcoQ element (f) Proportion of oiled common guillemots among those found dead or dying on beaches, the EcoQO is 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. OSPAR has requested further advice in order to ensure a scientifically sound implementation of this EcoQO.

6.5.2 Current baseline levels

Figure 6.5.2.1 illustrates the most recent (1995) analysis of current baseline levels of oiled common guillemots. These baselines are based on national or local schemes and do not correspond to the regions suggested below in Section 6.5.4. ICES does not have access to the underlying data on oiled bird rates, and so was unable to undertake any further analysis.

6.5.3 Historic trajectory of the metric

Camphuysen (2002) demonstrated a significant decline in the oiling rate of common guillemots on the North Sea coast of The Netherlands from 1976/1977 to 2001/2002 (Figure 6.5.3.1) and in the German Bight (Figure 6.5.3.2).

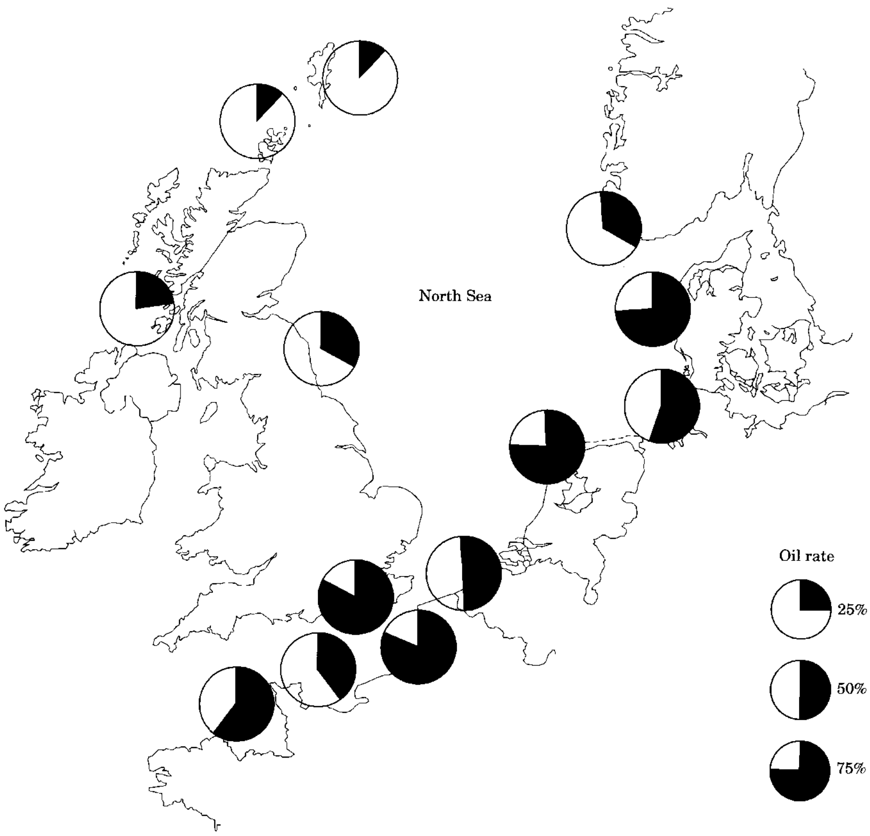


Figure 6.5.2.1. Differences in the oiling rates of common guillemots in the North Sea (Furness and Camphuysen, 1997).

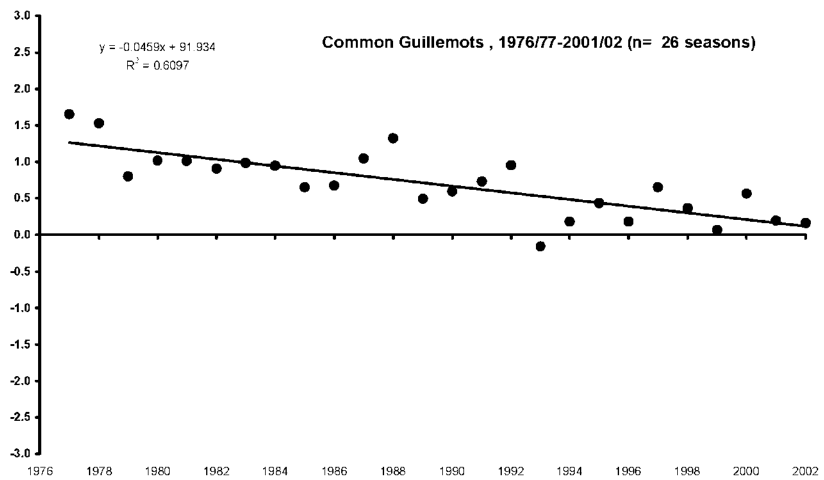


Figure 6.5.3.1. The decline in (logit-transformed) oil rates of common guillemots found dead along the North Sea coast in The Netherlands in winter from 1976/1977 (1976) to 2001/2002 (2002) (Camphuysen, 2002).

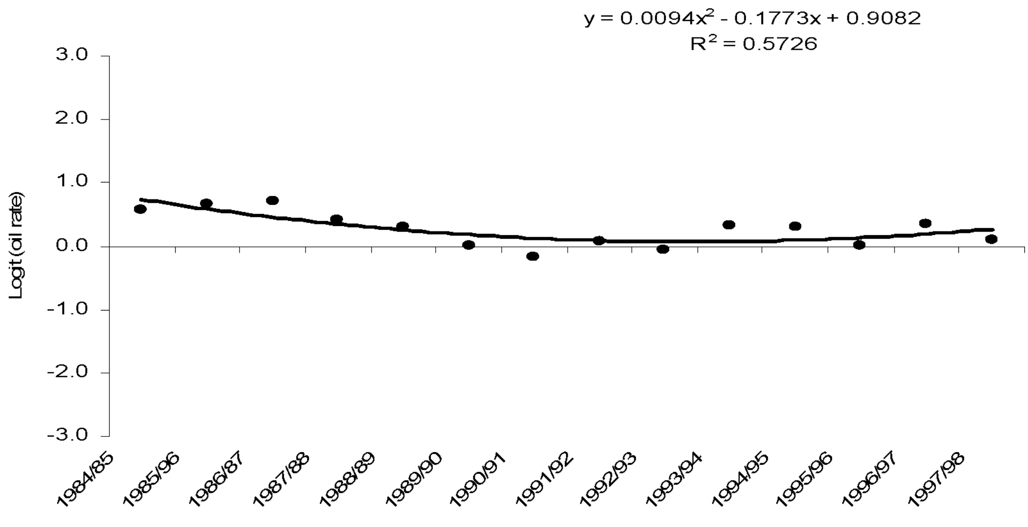


Figure 6.5.3.2. Change in the proportion of oiled common guillemots (logit-transformed oil rates) in the German Bight, winter 1984/1985 to 1997/1998 (modified data from Fleet and Reineking, 1999).

These are the most recent analyses of historic trends, and as ICES does not have access to the relevant data, it was impossible to analyse other regions. Camphuysen (1995b) showed time series of the observed index *x* for common guillemots with fitted linear trends (by least-squares estimation) for The Netherlands, Denmark, Germany, Norway, and the Shetland Islands. The declines found in Shetland, Germany, and The Netherlands were significant, whereas the shorter time series for Norway and Denmark did not reveal significant trends (Camphuysen, 1995b; Skov *et al.*, 1996).

The study of Camphuysen (2002), described in Section 6.5.4.1, indicates the likelihood of poor performance of the metric as oiling rates approach the 10% EcoQO. Only two areas of the North Sea appear to have oiling levels at or below the EcoQO: the seas around Orkney and around Shetland. Further analysis (as outlined in "Recommendations and advice", above) on the data from these areas would be helpful in assessing the risk of error in this evaluation. All other areas of the North Sea appear to have oiling rates well above the 10% level (30% or more). Exceptions might be revealed once further regional analysis is undertaken, as recommended above.

6.5.4 Draft guidelines for collecting information to determine whether the EcoQO is being met

The process of evaluating the proportion of oiled guillemots on beaches might be divided conveniently into three related phases:

1) Sampling of the North Sea and international coordination;
2) On-beach evaluation and recording;
3) Analysis and reporting.

OSPAR has developed guidelines for beached bird surveys within the Joint Assessment and Monitoring Programme (JAMP), led by The Netherlands and Germany in November 1995. These guidelines were endorsed by the OSPAR subsidiary bodies SIME and ASMO in 1996 (OSPAR, 1996). ACE has used relevant parts of these guidelines in developing the following sections. In addition, ACE has drawn heavily on a report to review this topic commissioned from the European expert on beached bird surveys by The Netherlands (Camphuysen, 2002).

6.5.4.1 Sampling of the North Sea and international coordination

Beached bird surveys are presently organized nationally or regionally on North Sea coasts. There is some international coordination, but this amounts to agreeing a common weekend for a single February survey of as many beaches as possible, and to compiling data for that weekend. There is considerable variation in the number of surveys per year, and in the proportion of coastline covered by the surveys in different areas of the North Sea. All surveys rely on volunteers walking beaches to count and record beached birds, so inevitably some bias will creep in due to variations in the availability of volunteers.

The underlying reason to establish and use EcoQOs is to aid in management decision-making. The oiled guillemot EcoQO would indicate the effectiveness of policies to reduce oil pollution in the North Sea (and Channel). It would presumably be useful for managers to know where (at a relatively fine geographical scale) oil is being discharged and approximately from what type of source. The former requires a fairly detailed geographical sampling scheme, while the latter requires a scheme that will examine the oil type on birds also. Only one of these is explicit in the EcoQO—the requirement for the EcoQO to be met in "all areas" of the North Sea.

Presently, most results of beached bird monitoring are presented on a national basis (Figure 6.5.2.1) in the eastern North Sea. Orkney and Shetland are separated due to different funding arrangements on the two islands. ACE suggests that the following regionalization scheme, adapted from Camphuysen and van Franeker (1992) and in conformity with the JAMP monitoring guidelines (OSPAR, 1996), might aid managers in identifying the source areas of oil pollution (single spills can potentially be observed directly or back-tracked using oil drift models):

1) Shetland;
2) Orkney and north coast of Scotland;
3) Moray Firth (Duncansby Head to Rattray Head);
4) Eastern Scotland (Rattray Head to Berwick on Tweed);
5) Northeast England (Berwick on Tweed to Spurn Head);
6) Eastern England (Spurn Head to North Foreland;
7) Eastern Channel (line between North Foreland and Belgian-French border to line between Cherbourg and Portland Bill);
8) Western Channel (west of line between Cherbourg and Portland Bill to Land's End to Ouessant);
9) Eastern Southern Bight (Belgian-French border to Texel);
10) Southern German Bight (Texel to Elbe);
11) Eastern German Bight (Elbe to Hanstholm);
12) Skagerrak (east of line between Hanstholm to Kristiansund, north of a line from Skagen to Gothenburg);
13) Kattegat (south to the OSPAR North Sea limit);
14) Southwestern Norway (Kristiansund to Stadt).

Analysis of existing information to determine the homogeneity (see recommendations, above) in results between sub-divisions of these proposed areas may suggest some changes in this regionalization, but it is probable that the data are too patchy for this to be successful in all parts of the North Sea. ACE suggests that a coordinator would be required for each country around the North Sea, who would aim to ensure that sufficient sampling was carried out within the national sections of each of the above areas. An international analyst/coordinator could gather together relevant results and produce an index for each of the above areas (or others, if deemed appropriate).

Sampling within the regions outlined above should be on a representative fraction of the coast directly bordering the North Sea and these fractions should be standardized over the years. Coasts bordering the open sea are of prime importance; sheltered bays and deep fjords are of secondary importance and should be treated separately (OSPAR, 1996). As a rule of thumb, at least 10% of the open sea coast length of each region should be covered.

The frequency of beached bird surveys through the year needs consideration. The higher the sampling frequency, the more precise can be the estimate of the proportion of oiled beached common guillemots and the less the ratio might be affected by a short-term variation in the amount of oiling. However, much beached bird surveying relies on volunteers, who are limited in the amount of sampling that they can do. Additionally, many of the sandy beaches that are suitable for the recording of oiled birds are "cleaned" in the summer by local authorities to improve them for tourists. At present, the only constant is the international beached bird count in late February. ACE recommends that beached bird surveys should provide monthly samples for at least the winter period (i.e., October–April). Where possible, year-round counts could be carried out in order to check seasonal trends.

Camphuysen (1995b) examined trends in oiling rates and the statistical power of appropriate trend tests, and Camphuysen (2002) examined the variability of the proportion of oiled beached birds from surveys in The Netherlands (i.e., the standard deviation of the (yearly) logit-transformed oil rate indices around the observed linear trend) (Figure 6.5.4.1.1). The results indicate that it would require six years of observation to reject the null-hypothesis with 80% chance that the objective of an oil rate of 10% has been reached when in fact 20% is still contaminated with oil. At lower levels of oil pollution, the time required would increase substantially (Camphuysen, 2002). A period of at least five years in which an average of 10% oiled common guillemots has been recorded should occur before the conclusion that the objective has been reached could be justified statistically. Even beyond this time period, it would be worth continuing surveys to ensure that there has not been a subsequent deterioration in performance.

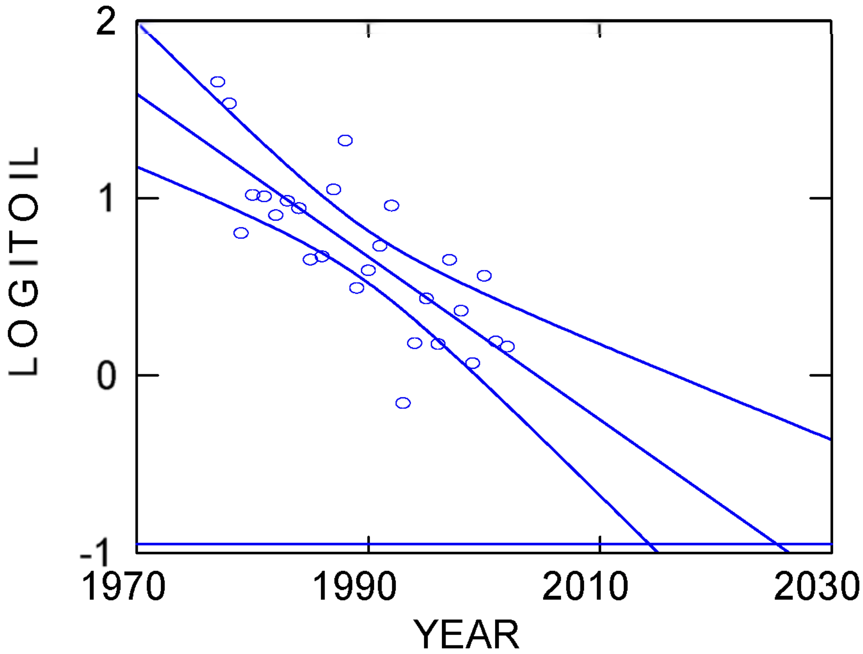


Figure 6.5.4.1.1. Significant decline in (logit-transformed) oiling rates of common guillemots found dead along the North Sea coast of The Netherlands in winter from 1976/1977 (1976) to 2001/2002 (2002), showing annual rates, the observed and predicted trend if this is continued at the same rate, and 95% confidence intervals (*n* = 26, multiple R = 0.781, multiple R^2 = 0.610, adjusted multiple R^2 = 0.593, Standard error of estimate = 0.286, residual mean square = 0.082) (NZG/NSO unpublished data, Camphuysen, 2002).

Table 6.5.4.2.1. Information recorded for each complete beached common guillemot.

Age	based on plumage characteristics (Kuschert *et al.*, 1981; Camphuysen, 1995a)
Oiling percentage	100%: fully covered in oil or chemical substance
	30%: upper or underside of rump (both = 60%) fully covered with oil
	10%: upper or underside of wing (both wings, under and upper side = 40%) covered with oil
	1%: single minor speck of oil
	other %: precise estimate of oil cover
	− ?: no oil seen, but corpse incomplete
	+ ?: oil on remains, but corpse incomplete
Recovery	presence of mark from previous count
Rings, notes	careful inspection of the presence of rings and/or other obvious causes of death (in the "notes" section of a record form)

Table 6.5.4.2.2. Information recorded for each count of beached birds.

Date	day, month, year
Place	place names visited
Stretch coding	unique code for stretch surveyed
Km surveyed	distance surveyed (nearest 100 m)
Presence of oil	yes/no (visibility)
Km of beach contaminated	length of beach with visible oil
Method of survey (vehicle)	on foot (preferred), bicycle, car, otherwise
Method of marking	clipped wings, corpses removed, other marks
Completeness of count	indication of reliability of count (problems with wind and sand, corpses may have been removed by sanitary department of local community)

6.5.4.2 On-beach evaluation and recording

Beached bird surveys should preferably be done on foot and not in vehicles (e.g., cars) (OSPAR, 1996). During each survey, the strandlines on the beach all need to be checked. In some cases, the highest and lowest strandlines may be some distance apart on the beach. Only complete corpses should be examined for the presence of oil. Beached common guillemots should be marked (e.g., clipping the primaries) or removed to avoid double counting. Marking should be permanent and easy to identify. Table 6.5.4.2.1 lists the information required from each common guillemot corpse (or live oiled bird).

For each count, the following information should be recorded (see also Table 6.5.4.2.2): date, place, km surveyed, km of coast with visible oil, characteristics of any oil noted on the beach, name(s) of observers, mark used to avoid double counts, completeness of survey and problems encountered, other significant pollution of the beach, list of beached birds.

6.5.4.3 Analysis and reporting

An overall analysis and compilation from national data would be needed to calculate oiling rates for each area (as suggested above or otherwise) of the North Sea. Trends in the overall numbers of beached birds per winter and in the overall proportion of oiled birds (oiling rate, percentage of oiled corpses of all complete corpses found) should be described. These trends might be most easily reported as a five-year running mean percentage oiled. Linear or other regression suggests that there is a model underlying any trends, while plain plotting of the percentage recorded each year would be relatively noisy due to short-term fluctuations. Figure 6.5.4.3.1 is an example of a running mean for the Dutch North Sea coast (calculated from data in Camphuysen, 2002).

One difficulty that needs to be addressed is that of the intermittent large-scale oiling incident, often due to a known shipping casualty. These cause intermittent peaks in the proportions of birds oiled and add considerably to the variance of oiling rates. In almost all large incidents, "emergency" beached bird surveys are undertaken to assess the scale of the impact of the incident, and in some cases include the "humanitarian" response to collect live oiled birds for cleaning. If the percentage oiling of live oiled birds removed from the beach is different from those (oiled or live) birds left on the beach, there would be bias introduced into the results of the regular (monthly) beached bird surveys. Conversely, guillemots in some areas are affected by intermittent large-scale mortality probably caused by starvation. Under these circumstances, large numbers of unoiled birds wash ashore, thus depressing the proportions oiled. This underlines the importance of note-taking at the time of the beached bird survey in order to aid in the interpretation of outliers within the data.

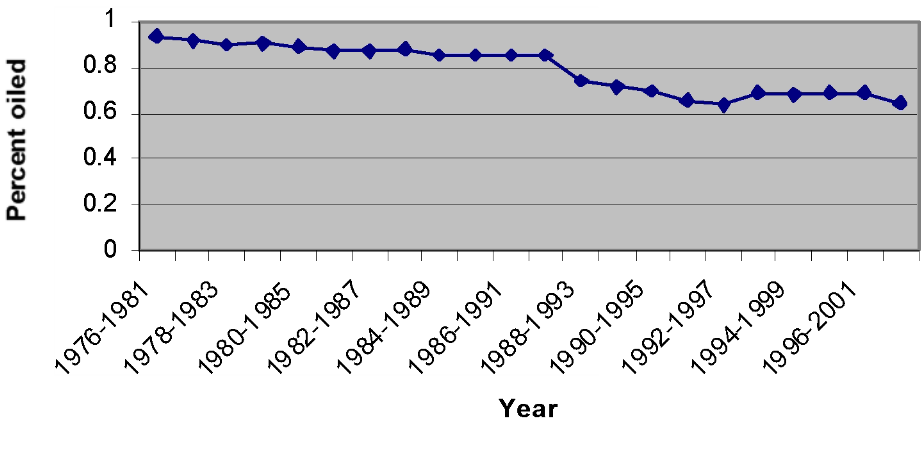


Figure 6.5.4.3.1. Five-year running mean of the percentage of oiled common guillemots found as complete corpses on Dutch North Sea coasts, 1976/1977 to 2001/2002 (from Camphuysen, 2002).

6.5.4.4 Further development of a programme to evaluate the oiled bird EcoQO

It is likely that beached bird counters taking part in recording rates of oiled guillemots for this EcoQO will also be undertaking a general survey of beached birds. Feedback on the results of these wider surveys at the same time as reporting on the EcoQO will probably be necessary in order to maintain the interest of the volunteer counters. ACE noted also that if the proposed EcoQO on plastic particles in the stomachs of northern fulmars is adopted (see Section 6.9, below), the monitoring protocols for this EcoQO would also need to be integrated into beached bird surveys.

If the results of the monitoring are to be used in decision-making in relation to management (see Section 6.5.5), then the oils contaminating beached birds need to be characterized. Substances encountered in recent years during beached bird surveys, as dumped in bulk into the sea, or as additives to, for example, lubricating oils used on vessels, include palm oil and other vegetable oils, paraffin, dodecylphenol, nonylphenol, polyisobutylene, olefines (i.e., octadecene, nonadecene, docosene), and dioctyldiphenylamine (McKelvey *et al.*, 1980; Engelen, 1987; Averbeck, 1990; Bommelé, 1991; Timm and Dahlmann, 1991; Zoun, 1991; Zoun *et al.*, 1991; Zoun and Boshuizen, 1992; Camphuysen *et al.*, 1999; Camphuysen, 2002). Seabird mortality incidents have been induced by apparently "harmless" substances such as fish oil (Newman and Pollock, 1973; Anon., 1975). The molecular features of contaminants causing disruption of the plumage of seabirds are well known and need not be repeated here (Rozemeijer *et al.*, 1992). Camphuysen (2002) proposes a sampling protocol to identify different types of oil and other substances in order to help identify sources of pollution. Since it is not possible to classify contaminants fully by eye in the field, specialized laboratory studies would be required to undertake analysis of the samples.

A further study that would support the use of this EcoQO is improvement in the understanding of drift patterns of corpses in the North Sea. Knowledge of the type of oil and likely origin of the oil would provide a powerful indicator of where further management measures might be applied.

6.5.5 Possible management measures

The majority of the oil that contaminates common guillemots is believed to come from illegal discharges from ships or from wrecked ships. Some contamination may also derive from the oil extraction industry or from riverine discharge. There are many well-known measures to reduce these causes of oil that are being applied at present. All management measures are related to the need to influence human behaviour.

On the positive side, the provision of free port waste reception facilities has proved very effective in German ports (Averbeck, 1991) and elsewhere. Free oil reception facilities were provided in Hamburg from June 1988 to the early 1990s. Oiling rates of birds found dead on German North Sea coasts generally decreased in the period 1984 to 2001; the oiling rate of common guillemots fell from 80% to 40% over this period. These trends in the oiling rates reflect the general decline in the level of oil pollution in the southern North Sea. The decreases in the German North Sea oiling rates were greater at the end of the 1980s and beginning of the 1990s than they were in later years, and were in contrast to rates on adjacent sections of German, Danish, and Dutch coasts (Camphuysen and van Franeker, 1992). Fleet and Reineking (2001) show this high-low-high-low pattern superimposed on the general decrease in Schleswig-Holstein (Figure 6.5.5.1).

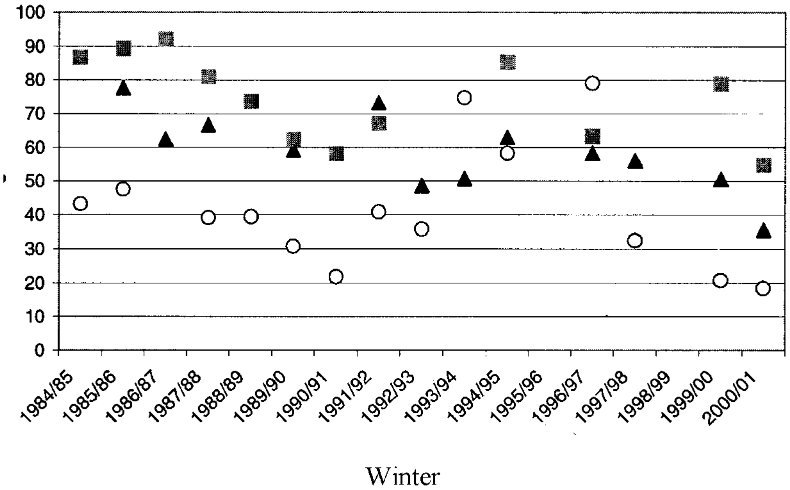


Figure 6.5.5.1. Oiling rate (percentage) of common guillemot corpses found on beaches in winter for three parts of the German North Sea coast (triangles: Niedersachsen; circles: Schleswig-Holstein; squares: Helgoland) in winters between 1984/1985 and 2000/2001.

On the negative side, probably the most critical is the need to improve detection, prosecution, and conviction rates among those discharging oil illegally. If polluters were more afraid of the consequences of their actions, it seems likely that their behaviour would change. It would be useful to compare the rates of oiling recorded on common guillemots and the density of offshore shipping to see whether there are any particular "hot spots" where oiling rates per unit of shipping were higher than elsewhere. This might enable enforcement measures to be better targeted. However, the risk of displacing illegal discharges to other places seems high.

If managers are concerned about reducing the absolute numbers of birds killed by oil, then targeting enforcement and other measures at areas/times of year when the most common guillemots are present would be wise. Several oil vulnerability atlases (e.g., Carter *et al.*, 1993; Webb *et al.*, 1995) are available to provide easy sources of information on this topic. Managers could gain even more precise information on numbers affected by coupling the digital data available in vulnerability atlases to oil drift models.

6.5.5.1 Example of an area of the North Sea where the EcoQO is being met

Effective measures to stop vessels from discharging oil near Orkney and Shetland provide an example of an area in the North Sea where this EcoQO is being met (Camphuysen and Heubeck, 2001). Beached bird surveys began in Orkney and Shetland prior to the opening of oil terminals on these islands in 1977 and 1978. Before the first tankers exported oil from these terminals, the incidence of oiled birds on beaches was relatively low. Both terminals experienced significant oil spills within months of opening, in March 1977 at Flotta (crude, 110 oiled birds found) and December 1978 at Sullom Voe (bunker fuel, 3,700 found), but the impacts of both of these incidents were relatively localized. A relatively high incidence of oiled birds was recorded in Orkney throughout 1978, with analyses indicating that crude oil and its residues comprised 32% of 43 samples taken.

In early December 1978, ten days after the first shipment of oil left the Sullom Voe Terminal in Shetland, hundreds of oiled birds began to wash ashore in Orkney and along the coasts of northeast Scotland. During the next four months, over 4,000 oiled birds were found on the coasts of Caithness, Orkney, and Shetland in a series of unattributable spills (i.e., excluding the fuel oil spill at Sullom Voe). Sampling during this period in Orkney showed an increased incidence of crude oils (65%, n=26) compared to 1978, with a similar pattern in Shetland (57% crude oils, n=42). Tankers passing the area could not be ruled out as being responsible, but the coincidental timing with the opening of the Sullom Voe terminal led to suspicions that tankers bound there were the main culprits, particularly since the terminal had opened before deballasting and oil reception facilities were completed (these only became operational in November 1979).

In the face of a public outcry and calls for the closure of the Sullom Voe Terminal until the deballasting plant was completed, measures were introduced in March 1979 to further discourage deballasting at sea by tankers bound for Sullom Voe, and to improve navigational safety. These included:

1) dedicated aerial surveillance of tankers and the seas around Shetland;
2) reporting requirements, routing, and areas of avoidance for tankers trading with Sullom Voe;
3) tankers entering Sullom Voe must carry at least 35% ballast, which was sampled to compare with beached oil;
4) preferential chartering of tankers with segregated ballast; and
5) tankers failing to comply with these rules would jeopardize their charter contract and may be refused loading facilities at the terminal.

Tankers were observed breaking these rules on a number of occasions in 1979 and 1980 and legal action was taken. However, the percentage of birds found oiled on beached bird surveys decreased in both Orkney and Shetland during 1980 and 1981. By 1982, crude oil sludges were reduced to 7% and 4% of the samples taken in Orkney and Shetland, respectively.

In summary, beached bird surveys helped demonstrate (1) low levels of oil pollution before the opening of these oil terminals, (2) the consequence of failing to provide deballasting facilities and to ensure serious enough penalties for illegal discharges at sea, and (3) the effectiveness of the measures taken to combat oil pollution.

6.5.6 References

Anon. 1975. Cape colony hit by pollution. Birds: 11.

Averbeck, C. 1990. Nonylphenol in der Meeresumwelt. Seevögel, 11: 44.

Averbeck, C. 1991. Ergebnisse des "Beached Bird Survey" der Bundesrepublik Deutschland im Februar 1990. Mitteilungen Norddeutsche Naturschutz Akademie, 2: 50–52.

Bommelé, M. 1991. Harmful effects on birds of floating lipophilic substances discharged from ships. *In* Oil pollution, beached bird surveys and policy: towards a more effective approach to an old problem, pp. 44–45. Proceedings of an international NZG/NSO workshop, 19 April 1991, Rijswijk. Ed. by C.J. Camphuysen and J.A. van Franeker. Sula, 5 (special issue).

Camphuysen, C.J. 1995a. Leeftijdsbepaling van Zeekoet *Uria aalge* en Alk *Alca torda* in de hand. Sula, 9: 1–22.

Camphuysen, C.J. 1995b. Olieslachtoffers langs de Nederlandse kust als indicatoren van de vervuiling van de zee met olie. Sula 9 (special issue). 90 pp.

Camphuysen, C.J. 2002. Oil rates in common guillemots: proportion of oiled common guillemots among those found dead or dying on beaches, guillemot-oil-EcoQO. BDC 03/2/4, Annex 1. OSPAR Commission, London.

Camphuysen, C.J., and Heubeck, M. 2001. Marine oil pollution and beached bird surveys: the development of a sensitive monitoring instrument. Environmental Pollution, 112: 443–461.

Camphuysen, C.J., and van Franeker, J.A. 1992. The value of beached bird surveys in monitoring marine oil pollution. Technichs Rapport Vogelbescherming 10, Vogelbescherming Nederland, Zeist, 191 pp.

Camphuysen, C.J., Wright, P.J., Leopold, M.F., Hüppop, O., and Reid, J.B. 1999. A review of the causes, and consequences at the population level, of mass mortalities of seabirds. Ed. by R.W. Furness and M.L. Tasker. *In* Diets of seabirds and consequences of changes in food supply. ICES Cooperative Research Report, 232: 51–56.

Carter, I.C., Williams, J.M., Webb, A., and Tasker, M.L. 1993. Seabird concentrations in the North Sea: an atlas of vulnerability to surface pollutants. Joint Nature Conservation Committee, Aberdeen. 39 pp.

Engelen, K.A.M. 1987. Zeevogels op de Waddeneilanden het slachtoffer van lijmachtige substantie. Sula, 1: 112–113.

Fleet, D.M., and Reineking, B. 1999. Zum Seevogelsterben an der deutschen und niederländischen Wattenmeerküste im Februar/ März 1999. Seevögel, 20(2): 63.

Fleet, D.M., and Reineking, B. 2001. Bestimmung, Quantifizierung und Bewertung der Öleinträge in der Nordsee zur Beurteilung der Schiffsentsorgung in deutschen Nordseehäfen. Final report to the Federal Environmental Agency, Berlin. 279 pp.

Furness, R.W., and Camphuysen, C.J. 1997. Seabirds as monitors of the marine environment. ICES Journal of Marine Science, 54: 726–737.

Kuschert, H., von Ekelöf, H.O., and Fleet, D.M. 1981. Neue Kriterien zur Altersbestimmung der Trottellumme (*Uria aalge*) und des Tordalken (*Alca torda*). Seevögel, 2: 58–61.

McKelvey, R.W., Robertson, I., and Whitehead, P.E. 1980. Effect of non-petroleum oil spills on wintering birds near Vancouver. Marine Pollution Bulletin, 11: 169–171.

Newman, G.G., and Pollock, D.E. 1973. Organic pollution of the marine environment by pelagic fish factories in the western Cape. South African Journal of Science, 69: 27–29.

OSPAR. 1996. JAMP Guidelines on standard methodology for the use of oiled beached birds as indicators of marine oil pollution. Ref no. 1995–6. OSPAR Commission, London. 49 pp.

Rozemeijer, M.J.C., Booij, K., Swennen, C, and Boon, J.P. 1992. Harmful effects of floating lipophilic substances discharged from ships on the plumage of birds. Netherlands Institute for Sea Research (NIOZ), Den Burg, The Netherlands, pp. 1–17.

Skov, H., Christensen, K.D., and Durinck, J. 1996. Trends in marine oil pollution in Denmark 1985–1995. An analysis of beached bird surveys. Danish Environmental Protection Agency, Danish Ministry of Environment: Working report no. 75. 62 pp.

Timm, D., and Dahlmann, G. 1991. Investigations into the source of non-mineral oils in the feathers of seabirds. *In* Oil pollution, beached bird surveys and policy: towards a more effective approach to an old problem, pp 15–17. Proceedings of an international NZG/NSO workshop, 19 April 1991, Rijswijk. Ed. by C.J. Camphuysen and J.A. van Franeker. Sula, 5 (special issue).

Webb, A., Stronach, A., Tasker, M.L., and Stone, C.J. 1995. Vulnerable concentrations of seabirds south and west of Britain. Joint Nature Conservation Committee, Peterborough. 47 pp.

Zoun, P.E.F. 1991. Onderzoek naar de Oorzaak van de Vogelsterfte langs de Nederlandse kust gedurende december 1988 en januari 1989. CDI-rapport no. H121519, Lelystad, 55 pp.

Zoun, P.E.F., Baars, A.J., and Boshuizen, R.S. 1991. A case of seabird mortality in the Netherlands during the winter of 1988/1989 caused by a spillage of Nonylphenol and vegetable oils. *In* Oil pollution, beached bird surveys and policy: towards a more effective approach to an old problem, pp. 47–48. Proceedings of an international NZG/NSO workshop, 19 April 1991, Rijswijk. Ed. by C.J. Camphuysen and J.A. van Franeker. Sula, 5 (special issue).

Zoun, P.E.F., and Boshuizen, R.S. 1992. Gannets victim to spillage of lubricating oil and dodecylphenol in the North Sea, winter 1990. Sula, 6: 29–30.

6.6 Initial development of EcoQ element (d) Utilization of seal breeding sites in the North Sea

Source of information

The 2003 Report of the Working Group on Marine Mammal Ecology (WGMME) (ICES CM 2003/ ACE:03).

Summary

Fidelity for natal sites has been documented in harbour seals and grey seals. Abandoning of a breeding site could be considered as an indicator of habitat degradation and/or the start of a depletion in population. Utilization of a breeding site can therefore be considered as an indicator for ecological quality. Long-term data on the utilization of breeding sites around the North Sea are available. These data should be analysed in order to further assess the usefulness of this EcoQO, and to develop it further. The analysis, which should include a definition of appropriate spatial and temporal scales for this EcoQO, should be undertaken on the initiative of the

lead country on this subject within OSPAR (the UK). The results of this analysis should be evaluated by ICES and, if appropriate, action plans for implementation after the triggering of the EcoQO should be developed.

Recommendations and advice

ICES recommends that, in order to further assess the usefulness of this EcoQO, and its further development, existing data on seal breeding sites should be analysed. This analysis, to be undertaken on the initiative of the lead country for this subject under OSPAR (the UK), should include a definition of appropriate temporal and spatial scales for this EcoQO. This analysis should then be reviewed by ICES.

Scientific background

6.6.1 Basis for the EcoQO

If habitat quality deteriorates within the geographical range of a species or a population, a change or a reduction in the species distribution may be observed before any impact may be detected in population size. With the fidelity for natal sites documented in harbour and grey seals, abandoning breeding sites could be considered as a strong indicator of habitat degradation and/or the start of a depletion.

Long time series of data on the utilization of breeding sites exist for many areas throughout the North Sea. Available data comprise counts of seals at breeding areas along the UK North Sea coast since the 1960s, in the Skagerrak and Kattegat since 1979, and in the Wadden Sea since 1970. Figures 6.6.1.1 and 6.6.1.2 show known breeding areas of, respectively, harbour seals and grey seals in the North Sea.

Harbour seals can move their breeding areas over relatively small spatial scales. Within the Orkney archipelago, harbour seals appear to have been displaced from one relatively small area, although the breeding population on the archipelago has remained relatively constant over the period of this displacement (Thompson *et al.*, 2001). In some places (e.g., Anholt), there is only one area that could possibly be used by seals for breeding.

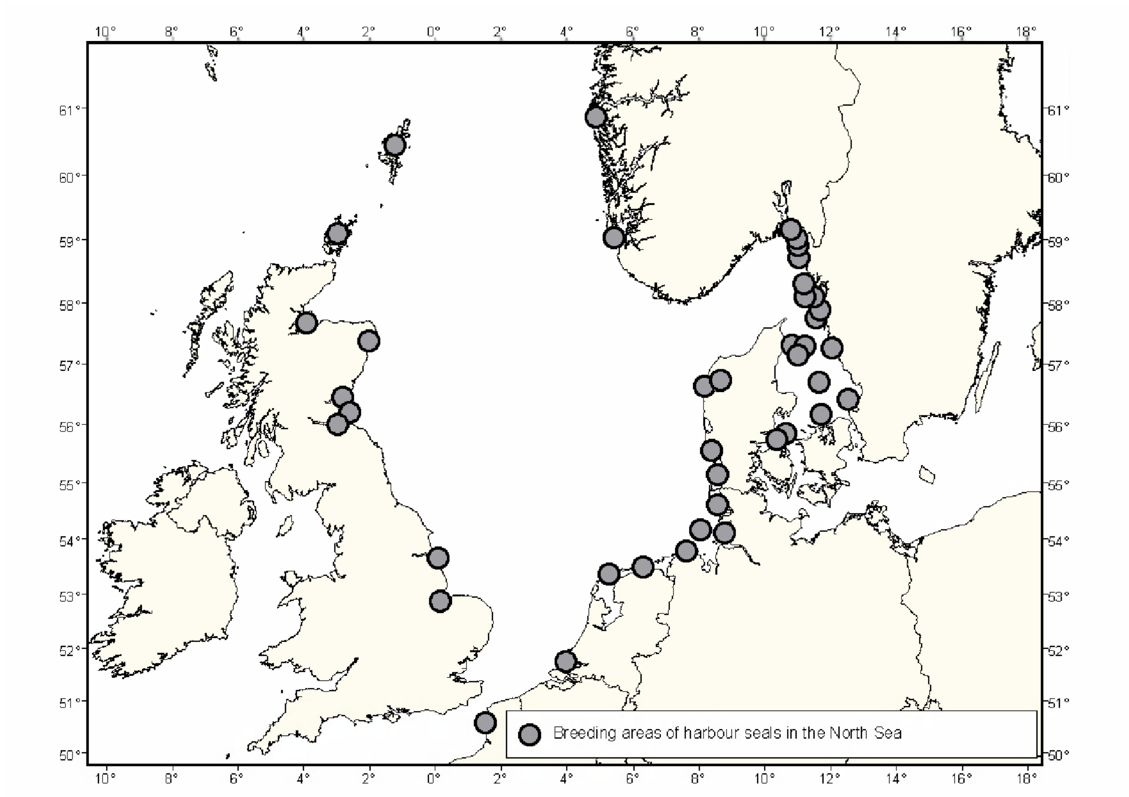


Figure 6.6.1.1. Breeding areas of harbour seals in the North Sea.

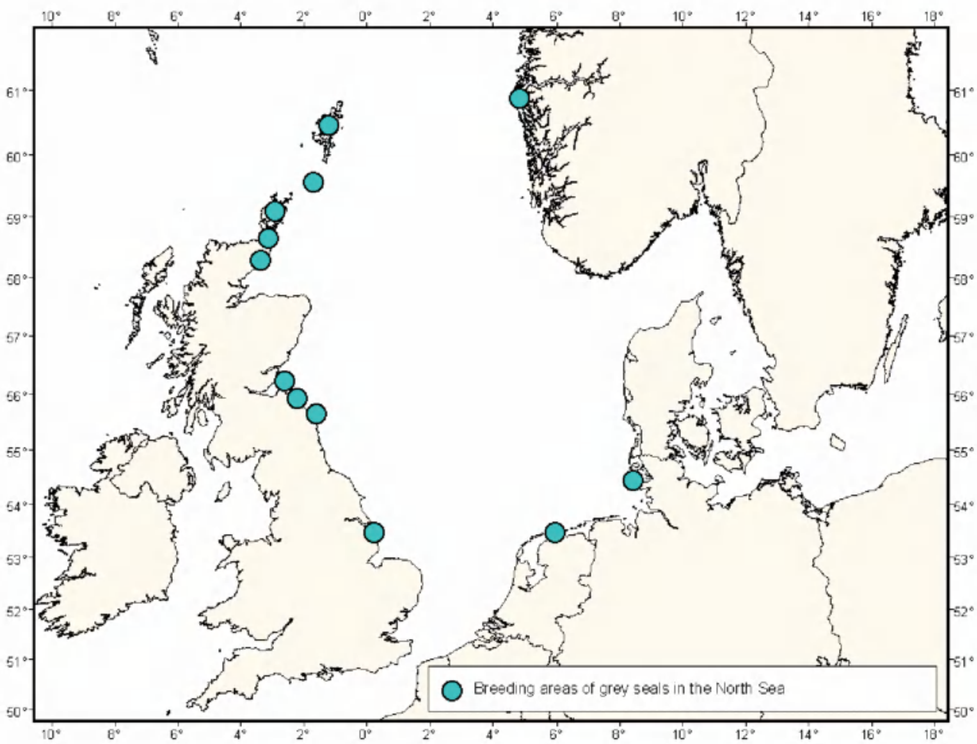


Figure 6.6.1.2. Breeding areas of grey seals in the North Sea.

6.6.2 Robustness of the EcoQO

If any breeding sites were abandoned, the EcoQO would trigger actions to determine the causes of this and, if relevant, to act. The occupation of a breeding site is at least partially responsive to human disturbance, but the linkages between the use of breeding sites and other factors, including environmental factors, are less well understood.

As this metric would only be qualitative and not quantitative, a serious impact on seal populations, causing a severe decline in numbers of breeding seals or numbers of seal pups at the colonies, without any of these colonies being abandoned, would not be detected by this EcoQ metric.

6.6.3 Use of the EcoQO in the North Sea

6.6.3.1 Proposed metric

The definition of the EcoQ metric is "utilization of seal breeding sites in the North Sea". The aim of utilizing this metric is to be able to reduce the continuous data on seal breeding sites (number of pups born) to binary data (presence/absence of pups). The abandonment of a breeding site would trigger research to be undertaken into the causes and to act on the results of that research, as appropriate.

6.6.3.2 Advice on the further development of the EcoQO

In order to be applied, this EcoQO needs to be further developed. To do this, and to evaluate the usefulness of this EcoQO, the available data should be used. The following actions would be needed for the further development of this EcoQO:

1) A definition of the appropriate spatial and temporal scales for the application of this EcoQO is required.
2) The available data on existing seal breeding sites in the North Sea should be collated, with information provided at the finest spatial scale possible. The aim of this should be to ensure that a breeding "site", as in the definition of this EcoQO, is selected at the appropriate spatial scale.
3) The information on time series data which exist on these sites, and on the recent loss of seal breeding sites in the North Sea, should be collated.
4) From 2) and 3), the relationship between recording breeding sites at different spatial and temporal scales should be modelled in order to detect the likelihood for a Hit, Miss or False Alarm which would trigger research into the reasons why the EcoQO is not being met.
5) Guidelines should be developed for establishing a programme capable of monitoring this EcoQO at the appropriate temporal and spatial scales, or ensuring that existing monitoring programmes can achieve this.

In order to accomplish the above activities, ICES advises that the lead country for this subject in OSPAR (the UK) appoint a spatial analyst who should work with the available data to develop a suite of implementation possibilities. The results of these analyses should be presented at the next meeting of the Working Group on Marine Mammal Ecology (WGMME) for evaluation, and to develop plans of actions that should be associated with the triggering of the EcoQO.

Furthermore, the relations between the distribution of seal breeding sites and other factors, including environmental factors, should be further investigated.

6.6.4 Reference

Thompson, P.M., van Parijs, S., and Kovacs, K.M. 2001. Local declines in the abundance in harbour seals: implications for the design and monitoring of protected areas. Journal of Applied Ecology, 38: 117–125.

6.7 Initial development of EcoQ element (g) Mercury concentrations in eggs and feathers of North Sea seabirds

Source of information

The 2003 Report of the Working Group on Seabird Ecology (WGSE) (ICES CM 2003/C:03).

Summary

Mercury contamination in the environment tends to be predominantly anthropogenic. Feathers from seabirds show different levels of mercury concentrations, depending on the level of mercury in their prey items. Current levels of mercury in the feathers of certain seabird species are well known and they can easily be monitored. Feathers from birds collected today have been shown to contain up to four times more mercury than those from birds collected over 100 years ago. The level of mercury in birds' feathers thus provides a useful tool for measuring trends of mercury in the environment. Measuring the level of mercury in seabird feathers can be useful as an assessment of the results of management activities aimed at reducing mercury contamination.

The proposed reference level for this EcoQO is the mercury level measured in feathers from birds of selected species collected over 100 years ago, from places with a suspected low mercury contamination. For eggs, such a reference level is not possible to establish, given the different level of mercury in eggs from that in feathers and the lack of reference material. Mercury levels in eggs can, however, also provide useful information on the trends of mercury in the environment.

The species for which the mercury concentrations in feathers should be measured are common tern, black-legged kittiwake, common guillemot, and northern gannet, from colonies in the southern and in the northern North Sea. The performance of the metric should aim at a downward direction. Setting the reference level as an objective is not realistic, given the past and present inputs of mercury into the environment. A more realistic objective for evaluating the consequences of effective management of mercury, which would be consistent with current scientific information on levels and trends of mercury in the environment, would be 1.5 times the measured reference level.

Recommendations and advice

ICES recommends that the mercury level in seabird feathers should be used as a metric for mercury levels in the environment. As a reference level for this EcoQO, the level in the feathers of seabirds collected over 100 years ago should be used. A realistic objective, consistent with current scientific information, would be a level of 1.5 times the reference level. The species to be monitored are common tern, black-legged kittiwake, common guillemot, and northern gannet, from colonies in the southern and northern North Sea.

Scientific background

6.7.1 Basis for the EcoQO

This EcoQ element was initially considered by ACE in 2001 and much of the relevant background information can be found in the 2001 ACE report (ICES, 2001). Additional information is reviewed here.

Mercury input to ecosystems tends to be predominantly anthropogenic. As a broad generalization, the bulk of mercury inputs into marine food webs arise from diffuse atmospheric inputs (Fitzgerald, 1995; Fitzgerald and Mason, 1998; Lamborg *et al.*, 2002), which will tend to generate broad and coherent patterns rather than local or short-term pulses of mercury in birds. However, local contamination resulting from river discharges can affect the levels measured in the feathers of seabirds that breed locally and have a short foraging range from the breeding area.

Mercury concentrations are high in seabird eggs and in seabird feathers (Lewis *et al.*, 1993; Monteiro and Furness, 1995; Becker *et al.*, 1998) and reflect dietary intake (Lewis and Furness, 1991, 1993; Burger, 1993, 2002; Becker *et al.*, 1993a, 1993b, 2002; Stewart *et al.*, 1997; Monteiro *et al.*, 1998; Bearhop *et al.*, 2000a, 2000b, 2000c; Monteiro and Furness, 2001a, 2001b; Champoux *et al.*, 2002; Fournier *et al.*, 2002; Nisbet *et al.*, 2002). However, this is complicated by a pattern of storage of mercury in soft tissues between moults and excretion of most of the body burden of mercury into growing feathers during the moult (Furness *et al.*, 1986; Braune and Gaskin, 1987a, 1987b; Thompson and Furness, 1989; Hario and Uuksulainen, 1993; Bearhop *et al.*, 2000c; Monteiro and Furness, 2001a, 2001b).

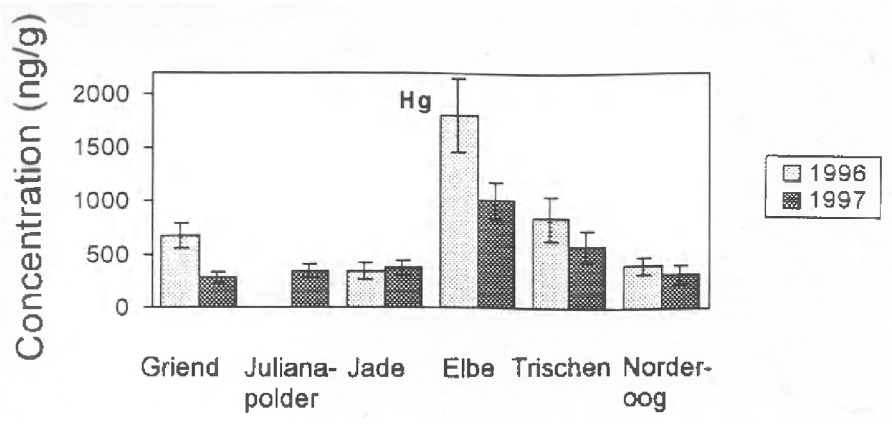


Figure 6.7.1.1. Spatial variation in mercury contamination of common tern eggs in 1996 and 1997 from breeding sites of the Wadden Sea (TMAP). Mean concentrations (ng g^{-1} fresh weight of egg content) and 95% confidence intervals are presented. N = 10 eggs each (Becker *et al.*, 1998).

Mercury levels in seabird eggs provide a very reliable measure of trends over years in local contamination, since seabirds feed close to their breeding colony during the period of egg formation. This also makes eggs very suitable for comparisons between localities (Figure 6.7.1.1) as well as over periods of years (Thyen and Becker, 2000; Sanpera *et al.*, 2000; Scheuhammer *et al.*, 2001; Braune *et al.*, 2002a, 2002b; Guitart *et al.*, 2003; Heinz and Hoffman, 2003).

By selecting particular seabird species with clearly defined diets, it is possible to monitor mercury contamination in a range of food chains. For example, some seabirds feed predominantly on epipelagic fish; other species feed on mesopelagic fish, others on intertidal molluscs, and so on (Monteiro *et al.*, 1995; Thompson *et al.*, 1998a, 1998b).

Changes in the diet composition of seabirds can alter their exposure to mercury, but examples of significant changes are rather few. One example is the change in trophic level of the northern fulmar between 1900 and 1990 (Thompson *et al.*, 1995). Another is the increase in mercury levels in scavenging seabirds resulting from the provision of demersal discards that have higher levels of mercury than the pelagic fish on which these seabirds primarily feed in the absence of discarding by the trawl fishery (Arcos *et al.*, 2002). Where such changes are not evident, the analysis of body feathers of seabird study skins in museum collections has demonstrated changes in mercury contamination over the last 150 years in a number of food chains and geographical regions. A four-fold increase in mercury levels over the last 150 years has been observed in many North Sea seabirds (Figure 6.7.1.2) (Thompson *et al.*, 1992a, 1992b, 1993a, 1993b, 1998a; Furness *et al.*, 1995; Monteiro and Furness, 1997; Monteiro *et al.*, 1999; Scharenberg and Struwe-Juhl, 2000).

6.7.2 Robustness of the proposed EcoQO

Mercury levels in birds are measured using well-established analytical methodologies that can be performed with high accuracy and reproducibility (Appelquist *et al.*, 1984; Thompson and Furness, 1989; Burger, 1993; Becker *et al.*, 1994; Bearhop *et al.*, 2000a; Christopher *et al.*, 2002). The close relationship between levels in birds and in their food is widely documented (Monteiro *et al.*, 1998; Monteiro and Furness, 2001a, 2001b). The literature on mercury in seabirds is very extensive and detailed. Unlike fish and marine mammals, seabirds do not show accumulation of mercury with age once fully grown. Levels in chicks are usually lower than in adults though in a few species levels are higher in chicks, so sampling does not need to take account of bird age except to separate chicks and older birds (Furness *et al.*, 1990; Thompson *et al.*, 1991). The use of seabird eggs to monitor mercury has already been implemented in the current Trilateral Monitoring and Assessment Programme (TMAP) in the Wadden Sea. Some relevant JAMP guidelines also exist (OSPAR, 1997).

6.7.3 The use of the EcoQO in the North Sea

6.7.3.1 Proposed metric

The proposed metric for this EcoQ is mercury concentrations in the body feathers and eggs of breeding common terns, black-legged kittiwakes, common guillemots, and northern gannets, from colonies in the southern and in the northern North Sea. These species are common, and have varying diets, breeding distributions, and foraging areas (Table 6.7.3.1.1).

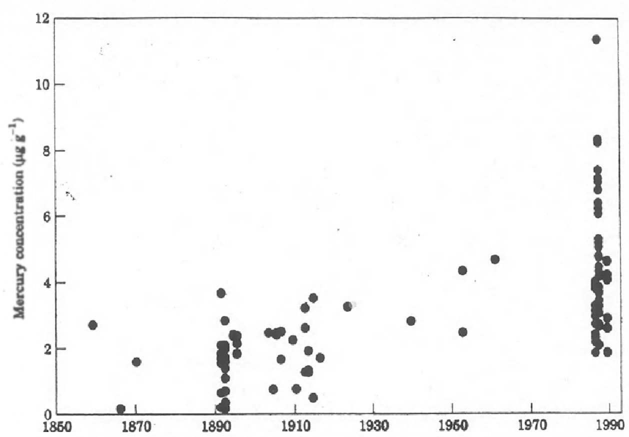


Figure 6.7.1.2. Mercury concentrations in body feathers of Atlantic puffin from southwest Britain and Ireland from 1850 to 1990 (from Thompson *et al*., 1992a).

Table 6.7.3.1.1. Seabird species suggested as monitors of marine contamination by mercury in the North Sea. Information on population size and trend, clutch size, diets, and feeding range is presented mostly from ICES (1999, 2002).

Species	Population size	Trend	Clutch size	Feeding range and niche	Diet
Black-legged kittiwake	505,000	–	2	Inshore/offshore on surface	Small pelagic fish, especially sandeel, sprat
Northern gannet	45,000 pairs	++	1	Wide-ranging, near surface	Sandeel, sprat, herring, mackerel, discards
Common tern	62,000 pairs	+/=	2–3	Coastal, surface	Small fish
Common guillemot	350,000 pairs	+	1	Inshore/offshore under water	Small pelagic fish, especially sandeel, sprat

6.7.3.2 Reference levels

Large numbers of seabirds collected from North Sea colonies during the 19th century are available in museum collections, and can provide reference levels for a period when mercury contamination from human activities was relatively small. However, previous studies of historical trends have mainly looked at Atlantic populations of seabirds and relatively few North Sea seabird populations have yet been investigated. As reference levels for feathers, the level of mercury in feathers of seabirds collected before 1900, from places which are unlikely to have been polluted with mercury, can be used. These reference levels vary considerably between seabird species. They depend on the diet and trophic status of the species, and to a small extent on the region, according to local natural sources of mercury. For many North Sea seabirds, reference levels are about one quarter of the current levels in each species.

Examples of pre-1900 levels are:

common tern:	1 mg kg^{-1};
black-legged kittiwake:	1.4 mg kg^{-1};
common guillemot:	1 mg kg^{-1};
northern gannet:	4.4 mg kg^{-1}.

For mercury levels in eggs, no reference level can be established, due to the virtual absence of egg contents in eggs gathered prior to 1900.

6.7.3.3 Current levels

For seabird body feathers, current levels have been reported in a large number of recent publications. Examples for body feathers of adult seabirds include: a) great skua: mean 7 mg kg^{-1} fresh mass feather (over 100 sampled 1995–2000), increasing by 0.4% per year from 1900–2000; b) northern gannet: 8 mg kg^{-1} fresh mass feather (over 100 sampled 1995–2000), increasing by 0.3% per year from 1900–2000; c) black-legged kittiwake: 3.3 mg kg^{-1}; and d) common guillemot: 1 mg kg^{-1}. More pelagic species (e.g., Atlantic puffin) show higher rates of increase, around 1–1.5% per year. In the southern North Sea, herring gulls showed high rates of increase of mercury contamination up to the 1960s, but showed subsequent reductions to 2000.

Current levels of mercury tend to be higher in the southern North Sea than in the northern North Sea. Examples of levels for the northern North Sea are: common tern 1.5 mg kg^{-1}; black-legged kittiwake 3 mg kg^{-1}; common guillemot 1.5 mg kg^{-1}; northern gannet

8 mg kg^{-1}, and for the southern North Sea: common tern 1.8 mg kg^{-1}; black-legged kittiwake 3.5 mg kg^{-1}; common guillemot 2 mg kg^{-1}; northern gannet 12 mg kg^{-1}.

6.7.3.4 Objectives

To reduce mercury contamination in the environment is a high priority. The analysis of seabird eggs and body feathers provides a robust way to measure trends in mercury contamination of the environment. Therefore, this EcoQ metric could be an indicator of the performance of measures aimed at reducing the input of mercury into the environment. The performance of the metric should be a downward direction in the level of mercury in seabird feathers or eggs. It would follow that a very detailed knowledge of the "reference level", or a level prior to the anthropogenic input of mercury into the environment, is not a prerequisite to start monitoring the performance of the objective. Therefore, monitoring mercury in eggs, for which no historic reference levels can be established, would also provide useful information concerning trends in mercury levels in the ecosystem.

The target level could be similar to the reference level for mercury in seabird feathers. The objective could be to reduce mercury contamination so that the levels in feathers of the seabird species in the North Sea decrease, eventually to reach an average that is no higher than the reference level for feathers of that species sampled before 1900. Reducing human inputs of mercury to the marine environment requires a reduction in mercury inputs to the atmosphere resulting from burning fossil fuels (including improved recovery of mercury), and a reduction in discharges of mercury by industry. However, given the high levels of mercury present in the environment, and still entering it as a result of human activities, a reduction to the reference level might be not realistic. A more realistic objective for evaluating the consequences of effective management of mercury contamination, which would be consistent with current scientific information on levels and trends in mercury, would be a reduction to 1.5 times the reference level.

6.7.4 Sampling requirements and selection of species and colonies

6.7.4.1 Samples required for setting reference levels

In order to provide a reference level for different species and areas, it will be necessary to carry out additional analyses of mercury levels in body feather samples from seabirds collected before 1900 from defined areas of the North Sea and now held in museum collections.

6.7.4.2 Samples required for setting current levels

Because there is a high individual variation in mercury concentrations within populations, it is necessary to calculate median or mean levels from a large number of sampled individuals (feather samples). Power analysis indicates that samples of at least 50 birds per species per area would be needed to detect a 20% difference in median or mean mercury concentrations between two samples using conventional statistical tests. Although the distributions of mercury concentrations in samples sometimes deviate from Normal distribution (showing a negative skew), the distribution is often considered to be close enough to Normal to permit the use of parametric statistics on untransformed data. In some cases, it is more appropriate to use non-parametric tests (e.g., Monteiro and Furness, 1997) and this also reduces the statistical power. However, catching large numbers of breeding adult seabirds at colonies to sample body feathers is generally rather easy, and so does not constrain sample size as the main cost of fieldwork. Analytical costs in the laboratory, however, may constrain sample size. If constrained by a small budget, then it would make sense to select a small number of seabird species to sample but to take adequate sample sizes from each of these selected species to detect a change in mercury concentrations.

Sampling should be both from the southern North Sea and from the northern North Sea for those species that breed in both areas. Probably the most suitable species to sample would be common tern, black-legged kittiwake, common guillemot, and northern gannet. These species cover a range of prey types and sizes. The selection of species could be based on the availability of data on 19th century levels of mercury in North Sea seabirds from different species and areas, or on the availability of recent time series of mercury levels.

There would be little point in producing reports on compliance with the objective annually, but two- to five-year cycles would be appropriate (ICES, 2002).

6.7.5 Historic trajectory and its historic performance

For the species and areas where historical trajectories of this metric are available, the variance in individual levels of mercury within samples is very high (Bearhop *et al.*, 2000a, 2000b; Thompson *et al.*, 1993a), but trends are well indicated by change over a period of one to several decades. This makes the median or mean mercury concentration in a large sample of birds a robust measure that is unlikely to generate False Alarms. The risk of False Alarms would be highest if the metric was obtained from only one or two seabird species and locations. If the metric is obtained from a variety of seabird species with different foraging ranges and diets, and from several different localities, any such local "hot spot" influences will be less noticeable.

6.7.6 References

Appelquist, H., Asbirk, S., and Drabæk, I. 1984. Mercury monitoring: mercury stability in bird feathers. Marine Pollution Bulletin, 15: 22–24.

Arcos, J.M., Ruiz, X., Bearhop, S., and Furness, R.W. 2002. Mercury levels in seabirds and their fish prey at the Ebro Delta (NW Mediterranean): the role of trawler discards as a source of contamination. Marine Ecology Progress Series, 232: 281–290.

Bearhop, S., Phillips, R.A., Thompson, D.R., Waldron, S., and Furness, R.W. 2000a. Variability in mercury concentrations of great skuas *Catharacta skua*: the influence of colony, diet and trophic status inferred from stable isotope signatures. Marine Ecology Progress Series, 195: 261–268.

Bearhop, S., Ruxton, G.D., and Furness, R.W. 2000c. Dynamics of mercury in blood and feathers of great skuas. Environmental Toxicology and Chemistry, 19: 1638–1643.

Bearhop, S., Waldron, S., Thompson, D., and Furness, R.W. 2000b. Bioamplification of mercury in great skua *Catharacta skua* chicks: the influence of trophic status as determined by stable isotope signatures of blood and feathers. Marine Pollution Bulletin, 40: 181–185.

Becker, P.H., Furness, R.W., and Henning, D. 1993a. Mercury dynamics in young common terns (*Sterna hirundo*) from a polluted environment. Ecotoxicology, 2: 33–40.

Becker, P.H., Furness, R.W., and Henning, D. 1993b. The value of chick feathers to assess spatial and interspecific variation in the mercury contamination of seabirds. Environmental Monitoring and Assessment, 28: 255–262.

Becker, P.H., Gonzalez-Solis, J., Behredns, B., and Croxall, J. 2002. Feather mercury levels in seabirds at South Georgia: influence of trophic position, sex and age. Marine Ecology Progress Series, 243: 261–269.

Becker, P.H., Henning, D., and Furness, R.W. 1994. Differences in mercury contamination and elimination during feather development in gull and tern broods. Archives of Environmental Contamination and Toxicology, 27: 162–167.

Becker, P.H., Thyen, S., Mickstein, S., Sommer, U., and Schmieder, K.R. 1998. Monitoring pollutants in coastal bird eggs in the Wadden Sea. Final report of the pilot study 1996–1997/Wadden Sea Ecosystem, 8: 55–101. Common Wadden Sea Secretariat, Wilhelmshaven.

Braune, B.M., and Gaskin, D.E. 1987a. A mercury budget for the Bonaparte's gull during autumn moult. Ornis Scandinavica, 18: 244–250.

Braune, B.M., and Gaskin, D.E. 1987b. Mercury levels in Bonaparte's gull (*Larus philadelphia*) during autumn moult in the Quoddy Region, New Brunswick, Canada. Archives of Environmental Contamination and Toxicology, 16: 539–549.

Braune, B.M., Donaldson, G.M., and Hobson, K.A. 2002a. Contaminant residues in seabird eggs from the Canadian Arctic. I. Temporal trends 1975–1998. Environmental Pollution, 114: 39–54.

Braune, B.M., Donaldson, G.M., and Hobson, K.A. 2002b. Contaminant residues in seabird eggs from the Canadian Arctic. II. Spatial trends and evidence from stable isotopes for intercolony differences. Environmental Pollution, 117: 133–145.

Burger, J. 1993. Metals in avian feathers: Bioindicators of environmental pollution. Reviews in Environmental Toxicology, 5: 203–311.

Burger, J. 2002. Food chain differences affect heavy metals in bird eggs in Barnegat Bay, New Jersey. Environmental Research, 90: 33–39.

Champoux, L., Rodrigue, J., DesGranges, J.L., Trudeau, S., Hontcla, A., Boily, M., and Spear, P. 2002. Assessment of contamination and biomarker responses in two species of herons on the St. Lawrence River. Environmental Monitoring and Assessment, 79: 193–215.

Christopher, S.J., Vander Pol, S.S., Pugh, R.S., Day, R.D., and Becker, P.R. 2002. Determination of mercury in the eggs of common murres (*Uria aalge*) for the seabird tissue archival and monitoring project. Journal of Analytical Atomic Spectrometry, 17: 780–785.

Fitzgerald, W.F. 1995. Is mercury increasing in the atmosphere? The need for an atmospheric mercury network. Water, Air and Soil Pollution, 80: 245–254.

Fitzgerald, W.F., and Mason, R.P. 1998. The case of atmospheric mercury contamination in remote areas. Environmental Science and Technology, 32: 1–7.

Fleming, S., Furness, R.W., and Davies, I.M. 2000. Contemporary patterns and historical rates of increase of mercury contamination in different marine food chains. ICES CM 2000/S:02.

Fournier, F., Karasov, W.H., Kenow, K.P., Meyer, M.W., and Hines, R.K. 2002. The oral bioavailability and toxicokinetics of methylmercury in common loon (*Gavia immer*) chicks. Comparative Biochemistry and Physiology A-Molecular and Integrative Physiology, 133: 703–714.

Furness, R.W., Lewis, S.A., and Mills, J.A. 1990. Mercury levels in the plumage of red-billed gulls *Larus novaehollandiae scopulinus* of known sex and age. Environmental Pollution, 63: 33–39.

Furness, R.W., Muirhead, S.J., and Woodburn, M. 1986. Using bird feathers to measure mercury in the environment: relationships between mercury content and moult. Marine Pollution Bulletin, 17: 27–30.

Furness, R.W., Thompson, D.R., and Becker, P.H. 1995. Spatial and temporal variation in mercury contamination of seabirds in the North Sea. Helgoländer Meeresuntersuchungen, 49: 605–615.

Guitart, R., Mateo, R., Sanpera, C., Hernandez-Matias, A., and Ruiz, X. 2003. Mercury and selenium levels in eggs of common terns (*Sterna hirundo*) from two breeding colonies in the Ebro Delta, Spain. Bulletin of Environmental Contamination and Toxicology, 70: 71–77.

Hario, M., and Uuksulainen, J. 1993. Mercury load according to moulting area in primaries of the nominate race of the lesser black-backed gull *Larus f. fuscus*. Ornis Fennica, 70: 32–39.

Heinz, G.H., and Hoffman, D.J. 2003. Embryotoxic thresholds of mercury: Estimates from individual mallard eggs. Archives of Environmental Contamination and Toxicology, 44: 257–264.

ICES. 1999. Report of the Working Group on Seabird Ecology. ICES CM 1999/C:5.

ICES. 2001. Report of the ICES Advisory Committee on Ecosystems, 2001. ICES Cooperative Research Report, 249: 44–47.

ICES. 2002. Report of the Working Group on Seabird Ecology. ICES CM 2002/C:4.

Kahle, S., and Becker, P.H. 1999. Bird blood as bioindicator for mercury in the environment. Chemosphere, 39: 2451–2457.

Lamborg, C.H., Fitzgerald, W.F., O'Donnell, J., and Torgersen, T. 2002. A non-steady-state compartmental model of global-scale mercury biogeochemistry with interhemispheric atmospheric gradients. Geochimica et Cosmochimica Acta, 66: 1105–1118.

Lewis, S.A., and Furness, R.W. 1991. Mercury accumulation and excretion in laboratory reared black-headed gull *Larus ridibundus* chicks. Archives of Environmental Contamination and Toxicology, 21: 316–320.

Lewis, S.A., and Furness, R.W. 1993. The role of eggs in mercury excretion by quail *Coturnix coturnix* and the implications for monitoring mercury pollution by analysis of feathers. Ecotoxicology, 2: 55–64.

Lewis, S.A., Becker, P.H., and Furness, R.W. 1993. Mercury levels in eggs, tissues, and feathers of herring gulls *Larus argentatus* from the German Wadden Sea coast. Environmental Pollution, 80: 293–299.

Monteiro, L.R., and Furness, R.W. 1995. Seabirds as monitors of mercury in the marine environment. Water, Air and Soil Pollution, 80: 851–870.

Monteiro, L.R., Furness, R.W., and del Nevo, A.J. 1995. Mercury levels in seabirds from the Azores, mid-North Atlantic Ocean. Archives of Environmental Contamination and Toxicology, 28: 304–309.

Monteiro, L.R., and Furness, R.W. 1997. Accelerated increase in mercury contamination in North Atlantic mesopelagic food chains as indicated by time series of seabird feathers. Environmental Toxicology and Chemistry, 16: 2489–2493.

Monteiro, L.R., and Furness, R.W. 2001a. Kinetics, dose-response, excretion, and toxicity of methylmercury in free-living Cory's shearwater chicks. Environmental Toxicology and Chemistry, 20: 1816–1823.

Monteiro, L.R., and Furness, R.W. 2001b. Kinetics, dose-response, and excretion of methylmercury in free-living adult Cory's shearwaters. Environmental Science and Technology, 35: 739–746.

Monteiro, L.R., Granadeiro, J.P., and Furness, R.W. 1998. Relationship between mercury levels and diet in Azores seabirds. Marine Ecology Progress Series, 166: 259–265.

Monteiro, L.R., Granadeiro, J.P., Furness, R.W., and Oliveira, P. 1999. Contemporary patterns of mercury contamination in the Portuguese Atlantic inferred from mercury concentrations in seabird tissues. Marine Environmental Research, 47: 137–156.

Nisbet, I.C.T., Montoya, J.P., Burger, J., and Hatch, J.J. 2002. Use of stable isotopes to investigate individual differences in diets and mercury exposures among common terns *Sterna hirundo* in breeding and wintering grounds. Marine Ecology Progress Series, 242: 267–274.

OSPAR. 1997. JAMP guidelines for monitoring contaminants in biota. OSPAR Commission, London.

Sanpera, C., Morera, M., Ruiz, X., and Jover, L. 2000. Variability of mercury and selenium levels in clutches of Audouin's gulls (*Larus audouinii*) breeding at the Chafarinas Islands, southwest Mediterranean. Archives of Environmental Contamination and Toxicology, 39: 119–123.

Scharenberg, W., and Struwe-Juhl, B. 2000. Total mercury in feathers of white-tailed eagle (*Haliaeetus albicilla* L.) from northern Germany over 50 years. Bulletin of Environmental Contamination and Toxicology, 64: 686–692.

Scheuhammer, A.M., Perrault, J.A., and Bond, D.E. 2001. Mercury, methylmercury, and selenium concentrations in eggs of common loons (*Gavia immer*) from Canada. Environmental Monitoring and Assessment, 72: 79–94.

Stewart, F.M., Phillips, R.A., Catry, P., and Furness, R.W. 1997. Influence of species, age and diet on mercury concentrations in Shetland seabirds. Marine Ecology Progress Series, 151: 237–244.

Thompson, D.R., Bearhop, S., Speakman, J.R., and Furness, R.W. 1998b. Feathers as a means of monitoring mercury in seabirds insights from stable isotope analysis. Environmental Pollution, 101: 193–200.

Thompson, D.R., Becker, P.H., and Furness, R.W. 1993a. Long-term changes in mercury concentrations in herring gulls *Larus argentatus* and common terns *Sterna hirundo* from the German North Sea coast. Journal of Applied Ecology, 30: 316–320.

Thompson, D.R. and Furness, R.W. 1989. Comparison of the levels of total and organic mercury in seabird feathers. Marine Pollution Bulletin, 20: 577–579.

Thompson, D.R., Furness, R.W., and Walsh, P.M. 1992a. Historical changes in mercury concentrations in the marine ecosystem of the north and north-east Atlantic ocean as indicated by seabird feathers. Journal of Applied Ecology, 29: 79–84.

Thompson, D.R., Furness, R.W., and Barrett, R.T. 1992b. Mercury concentrations in seabirds from colonies in the Northeast Atlantic. Archives of

Environmental Contamination and Toxicology, 23: 383–389.
Thompson, D.R., Furness, R.W., and Lewis, S.A. 1993b. Temporal and spatial variation in mercury concentrations in some albatrosses and petrels from the Subantarctic. Polar Biology, 13: 239–244.
Thompson, D.R., Furness, R.W., and Lewis, S.A. 1995. Diets and long-term changes in delta15N and delta13C values in northern fulmars *Fulmarus glacialis* from two northeast Atlantic colonies. Marine Ecology Progress Series, 125: 3–11.
Thompson, D.R., Furness, R.W., and Monteiro, L.R. 1998a. Seabirds as biomonitors of mercury inputs to epipelagic and mesopelagic marine food chains. Science of the Total Environment, 213: 299–305.
Thompson, D.R., Hamer, K.C., and Furness, R.W. 1991. Mercury accumulation in great skuas, *Catharacta skua* of known age and sex, and its effects upon breeding and survival. Journal of Applied Ecology, 28: 672–684.
Thyen, S., and Becker, P.H. 2000. Aktuelle Ergebnisse des Schadstoffmonitorings mit Küstenvögeln im Wattenmeer. Vogelwelt, 121: 281–291.
Thyen, S., Becker, P.H., and Behmann, H. 2000. Organochlorine and mercury contamination of little terns (*Sterna albifrons*) breeding at the western Baltic Sea, 1978–96. Environmental Pollution, 108: 225–238.

6.8 Initial development of EcoQ element (h) Organochlorine concentrations in the eggs of North Sea seabirds

Source of information

The 2003 Report of the Working Group on Seabird Ecology (WGSE) (ICES CM 2003/C:03).

Summary

Levels of organochlorines in seabird eggs show an immediate response to changes in the level of organochlorine contamination of the marine environment, and can therefore be used as an EcoQ metric. Current programmes clearly indicate spatial and temporal trends in the level of organochlorines in the eggs of different species of seabirds. Given the fact that organochlorines are man-made substances, the performance of the metric should be a downward trend. However, due to the fact that these substances are very persistent in the environment, it is not realistic to set the objective at zero. More realistic objectives for evaluating the consequences of effective management of these organochlorines, which would be consistent with current scientific information on levels and trends of these substances in oystercatcher and common tern, would be <20 ng total PCBs g^{-1} egg fresh mass, <10 ng DDT and metabolites g^{-1} egg fresh mass, <2 ng HCB g^{-1} egg fresh mass, and <2 ng HCH g^{-1} egg fresh mass. Other species, which are proposed as useful species to monitor the organochlorine content of their eggs, are common eider, northern gannet, and common guillemot.

Recommendations and advice

ICES recommends that the concentrations of organochlorines in seabirds' eggs be used as a measure for organochlorine levels in the marine environment. Objectives for the levels in oystercatcher and common tern of <20 ng total PCBs g^{-1} egg fresh mass, <10 ng DDT and metabolites g^{-1} egg fresh mass, <2 ng HCB g^{-1} egg fresh mass, and <2 ng HCH g^{-1} egg fresh mass, would all be consistent with current scientific information on levels and trends of these substances, and the effectiveness of management in reducing their introduction into the environment. Other species which should be monitored are common eider, northern gannet, and common guillemot.

Scientific background

6.8.1 Basis for the EcoQO

This EcoQ element was initially considered by ACE in 2001 and much of the relevant background information can be found in the 2001 ACE report (ICES, 2001). Additional information is reviewed here.

Marine pollution by environmental chemicals is a worldwide problem, endangering marine organisms and ecosystem health. Persistent toxic substances, such as organochlorines, are of special concern. These substances may affect all ecosystem levels. Aspects of seabird biology, such as reproduction, immune functions, and embryonic survival may be affected (Becker *et al.*, 1993; Grasman and Fox, 2001; Bosveld and van den Berg, 2002; Champoux *et al.*, 2002; Helander *et al.*, 2002). The use of seabirds as monitors of marine contamination by organochlorines such as PCBs, DDT and metabolites, HCB, HCH, and others has been advocated many times (Gilbertson *et al.*, 1987; Becker, 1989, 1991; Furness, 1993; Barrett *et al.*, 1996; Elliott *et al.*, 1992; Becker *et al.*, 1998; ICES, 2000; Munoz and Becker, 1999; Mattig *et al.*, 2000; Braune *et al.*, 2002a, 2002b; Norstrom *et al.*, 2002; Weseloh *et al.*, 2002) and has been implemented already in some current monitoring programmes in the North Sea. The objective is relevant to the North Sea, where organochlorine inputs remain high (De Jong *et al.*, 1999).

Current programmes demonstrate clearly the value of seabird eggs to indicate spatial and temporal trends in marine pollution by organochlorines (Becker *et al.*, 1998; Thyen and Becker, 2000). In the southern North Sea there has been a decreasing trend in organochlorine levels in seabird eggs since the early 1990s (Figure 6.8.1.1), but locally there are high levels (Figure 6.8.1.2) which, however, do not seem to be harmful to the birds during reproduction (Exo *et al.*, 1998).

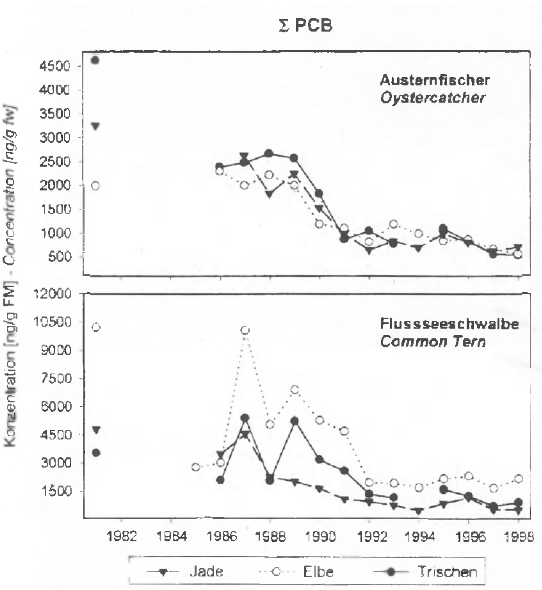


Figure 6.8.1.1. Temporal trends in PCB contamination of Eurasian oystercatcher and common tern eggs from selected breeding sites of the Wadden Sea (TMAP). FW=fresh weight of egg content (Thyen and Becker, 2000).

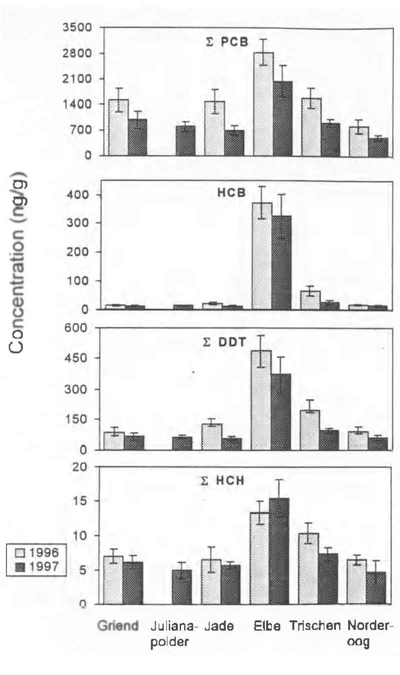


Figure 6.8.1.2. Spatial variation in organochlorine contamination of common tern eggs in 1996 and 1997 from breeding sites of the Wadden Sea (TMAP). Mean concentrations (ng g^{-1} fresh weight of egg content) and 95% confidence intervals are presented. N = 10 eggs each (from Becker *et al.*, 1998).

6.8.2 Robustness of the proposed EcoQO

Levels of organochlorines in seabirds show an immediate response to changes in contaminant loads in the marine environment. Consequently, they clearly indicate changing levels (Thyen *et al.*, 2000; Thyen; and Becker, 2000; Custer *et al.*, 2001) and changes in anthropogenic discharges and emissions of organochlorines. In this way the effectiveness of measures for the reduction of contamination can be demonstrated.

Trend data are available for various parts of the North Sea for nearly forty years. OSPAR (1997) has published guidelines for sampling and analysing seabird eggs. The key compounds are PCBs, DDT and metabolites, HCB, and HCH isomers. These can be analysed synchronously using the same analytical procedure. There is a clear parameter signal, as eggs can only be taken in the breeding season, thus reducing the effects of seasonal variation.

Monitoring can investigate temporal and spatial variation as well as local contaminant input, as seabirds forage in restricted distances from colonies during the period of egg formation. Foraging ranges vary between species, but are generally well known. Studies in the southern North Sea show clear local differences in contamination between colonies.

6.8.3 The use of the EcoQO in the North Sea

Monitoring of contaminants in seabirds is highly desirable as a cost-effective and informative procedure indicating change in marine contamination. Advantages in the use of seabirds as indicators of organochlorine contaminants have recently been reviewed (ICES, 2000) and include the following features of seabirds:

- well-known taxonomy and biology;
- tendency to accumulate high concentrations;
- ease of sampling (eggs);
- known foraging range and diets;
- resistance to toxic effects;
- low variance of contaminants levels within the population.

Consequently, seabirds offer some advantages compared to physical or other marine biota samples when organochlorine monitoring is needed.

The sampling of seabird eggs as a means of monitoring the organochlorine contamination of seabirds should be developed into integrated marine pollution monitoring programmes. Appropriate locally common and widespread species should be selected. A list of proposed species, with colonies in the southern and northern North Sea, is given in Table 6.8.3.1.

Table 6.8.3.1. Seabird species suggested as monitors of marine contamination by organochlorines in the North Sea. Information on population size and trend, clutch size, diets, and feeding range is presented mostly from ICES (1999, 2002). Common tern and Eurasian oystercatcher are already in use for monitoring in the Wadden Sea Trilateral Monitoring and Assessment Programme (TMAP).

Species	Population size	Trend	Clutch size	Feeding range	Diet
Common eider	40,000 pairs	–	3–6	Coastal	Blue mussel, other molluscs and crustaceans
Northern gannet	45,000 pairs	++	1	Wide-ranging	Sandeel, sprat, herring, mackerel, discards
Common tern	62,000 pairs	+/=	2–3	Coastal	Small fish
Common guillemot	350,000 pairs	+	1	Inshore/offshore	Small pelagic fish, especially sandeel, sprat
Eurasian oystercatcher	50,000 pairs	+	3–4	Coastal, intertidal areas	Shellfish, intertidal and terrestrial invertebrates

6.8.3.1 Proposed metric

The proposed metric is the mean concentration of each of the various organochlorine compounds (or group of compounds) in eggs of selected species of seabirds.

6.8.3.2 Reference levels

The reference level for this metric is zero ng g^{-1} egg fresh mass, as these are man-made chemicals only produced in recent decades.

6.8.3.3 Current levels

Current levels of PCBs, DDT (and metabolites), HCB, and HCH are available for eggs of the common tern and Eurasian oystercatcher from the southern North Sea (ng g^{-1} egg fresh mass, range of 6–7 sampling sites, data from 1997 (Becker *et al.*, 1998, reported in ICES, 2001)) as follows:

PCBs: common tern 702–2042 ng g^{-1};
Eurasian oystercatcher 492–1055 ng g^{-1};
DDT and metabolites: common tern 56–371 ng g^{-1};
Eurasian oystercatcher 22–103 ng g^{-1};
HCB: common tern 11–325 ng g^{-1};
Eurasian oystercatcher 4–60 ng g^{-1};
HCH: common tern 5–15 ng g^{-1};
Eurasian oystercatcher 3–10 ng g^{-1}.

There are few, if any, data for levels of organochlorines in eggs of these species from the northern North Sea, but sampling has been carried out with common guillemots and northern gannets from the UK colonies in the northern North Sea (Alcock *et al.*, 2002). Those data may be used to set current and target levels for samples of common guillemot and northern gannet. This increase to four seabird species as monitors would broaden the coverage of food chains to include offshore as well as coastal, and a range of larger pelagic fish prey (Table 6.8.3.1). Another suitable sentinel species would be the common eider, which feeds mainly on the blue mussel, but also on a variety of other molluscs and crustaceans. Eider eggs have been used to study levels of organochlorines in the coastal ecosystem of Finland (Franson *et al.*, 2000), but as far as ACE is aware, current organochlorine levels in eggs of North Sea eiders have not been documented.

6.8.3.4 Target levels and objectives

Setting very concrete target levels as concentrations is difficult and rather arbitrary, since the reference levels for organochlorines are zero, and this would also be the desirable target. However, even in the absence of any inputs, due to the very long persistence of these chemicals, the levels in the food web would remain high for many years. In any case, the performance of the metric should be a downward trend in the levels of organochlorines. Practical targets that have been suggested for the eggs of common tern and Eurasian oystercatcher, from both the southern and the northern North Sea, are:

- <20 ng total PCBs g^{-1} egg fresh mass;
- <10 ng DDT and metabolites g^{-1} egg fresh mass;
- <2 ng HCB g^{-1} egg fresh mass;
- <2 ng HCH g^{-1} egg fresh mass.

These levels would all be consistent with current scientific information on the levels and trends of these substances, and would serve to indicate the effectiveness of management in reducing their introduction into the environment.

Given that these are very persistent chemicals, such levels could not be achieved until some decades from now. Probably the prospects of meeting these levels would be higher in the northern North Sea than in the southern North Sea. There would be little point in producing reports annually, but a two- or five-year cycle might be appropriate.

6.8.3.5 Sampling requirements

Sampling eggs is normally done by taking a single freshly-laid egg from each clutch, preferably consistently the first-laid egg. There tends to be structure within seabird colonies, with higher quality and often older birds nesting in the centre of the colony and poorer quality, often first-time breeders, nesting on the edge. This structure also tends to result in differences in laying date, with higher quality and older birds laying earlier in the season. Organochlorine levels may vary between these categories, so that sampling should attempt to take a random sample of eggs, or at least a consistent sample from year to year.

6.8.3.6 Historic trajectory and its historic performance

Eggs provide a measure of organochlorines in the seabird diet in the few days before egg laying. At this time, birds are constrained to feed relatively close to the breeding site. This increases the risk of False Alarms since high concentrations can arise at one site in one year as a consequence of local contamination. Calculating a running mean over three or four years and sampling from several species and from different colonies would minimize this risk of False Alarms.

6.8.4 References

Alcock, R.E., Boumphrey, R., Malcolm, H.M., Osborn, D., and Jones, K.C. 2002. Temporal and spatial trends of PCB congeners in UK gannet eggs. Ambio, 31: 202–206.

Barrett, R.T., Skaare, J.U., and Gabrielsen, G.W. 1996. Recent changes in levels of persistent organochlorines and mercury in eggs of seabirds from the Barents Sea. Environmental Pollution, 92: 13–18.

Becker, P.H. 1989. Seabirds as monitor organisms of contaminants along the German North Sea Coast. Helgoländer Meeresuntersuchungen, 43: 395–403.

Becker, P.H. 1991. Population and contamination studies in coastal birds: the common tern *Sterna hirundo*. *In* Bird population studies: relevance to conservation and management, pp. 433–460. Ed. by C.M. Perrins, J.D. Lebreton, and G.J.M. Hirons. Oxford University Press, Oxford.

Becker, P.H., Schumann, S., and Koepff, C. 1993. Hatching failure in common terns (*Sterna hirundo*) in relation to environmental chemicals. Environmental Pollution, 79: 207–213.

Becker, P.H., Thyen, S., Mickstein, S., Sommer, U., and Schmieder, K.R. 1998. Monitoring pollutants in coastal bird eggs in the Wadden Sea. Final Report of the Pilot Study 1996–1997/Wadden Sea Ecosystem, 8: 59–101. Common Wadden Sea Secretariat, Wilhelmshaven.

Bosveld, A.T.C., and van den Berg, M. 2002. Reproductive failure and endocrine disruption by organohalogens in fish-eating birds. Toxicology, 181: 155–159.

Braune, B.M., Donaldson, G.M., and Hobson, K.A. 2002a. Contaminant residues in seabird eggs from the Canadian Arctic. I. Temporal trends 1975–1998. Environmental Pollution, 114: 39–54.

Braune, B.M., Donaldson, G.M., and Hobson, K.A. 2002b. Contaminant residues in seabird eggs from the Canadian Arctic. II. Spatial trends and evidence from stable isotopes for intercolony differences. Environmental Pollution, 117: 133–145.

Champoux, L., Rodrigue, J., DesGranges, J.L., Trudeau, S., Hontcla, A., Boily, M., and Spear, P. 2002. Assessment of contamination and biomarker responses in two species of herons on the St. Lawrence River. Environmental Monitoring and Assessment, 79: 193–215.

Custer, T.W., Custer, C.M., Hines, R.K., Stromborg, K.L., Allen, P.D., Melancon, M.J., and Henshel, D.S. 2001. Organochlorine contaminants and biomarker response in double-crested cormorants nesting in Green Bay and Lake Michigan, Wisconsin, USA. Archives of Environmental Contamination and Toxicology, 40: 89–100.

De Jong, F., Bakker, J., van Berkel, C., Dahl, K., Dankers, N., Gätje, C., Marencic, H., and Potel, P. 1999. 1999 Wadden Sea quality status report. Wadden Sea Ecosystem 9, Common Wadden Sea Secretariat, Wilhelmshaven.

Elliott, J.E., Noble, D.G., Norstrom, R.J., Whitehead, P.E., Simon, M., Pearce, P.A., and Peakall, D.B. 1992. Patterns and trends of organic contaminants in Canadian seabird eggs, 1968–90. *In* Persistent pollutants in marine ecosystems, pp. 181–194. Ed. by C.H. Walker and D.R. Livingstone. Pergamon Press, Oxford.

Exo, K.M., Becker, P.H., and Sommer, U. 1998. Environmental chemicals in eggs of inland and Wadden Sea breeding oystercatchers (*Haematopus ostralegus*). Journal für Ornithologie, 139: 401–405.

Franson, J.C., Hollmen, T., Poppenga, R.H., Hario, M., Kilpi, M., and Smith, M.R. 2000. Selected trace elements and organochlorines: Some findings in blood and eggs of nesting common eiders (*Somateria mollissima*) from Finland. Environmental Toxicology and Chemistry, 19: 1340–1347.

Furness, R.W. 1993. Birds as monitors of pollutants. *In* Birds as monitors of environmental change, pp. 86–143. Ed. by R.W. Furness and J.J.D. Greenwood. Chapman and Hall, London.

Gilbertson, M., Elliott, J.E., and Peakall, D.B. 1987. Seabirds as indicators of marine pollution. *In* The value of birds, pp. 231–248. Ed. by A.W. Diamond and F.L. Filion. ICBP Technical Publication.

Grasman, K.A., and Fox, G.A. 2001. Associations between altered immune function and organochlorine contamination in young Caspian terns (*Sterna caspia*) from Lake Huron, 1997–1999. Ecotoxicology, 10: 101–114.

Helander, B., Olsson, A., Bignert, A., Asplund, L., and Litzen, K. 2002. The role of DDE, PCB, coplanar PCB and eggshell parameters for reproduction in the white-tailed sea eagle (*Haliaeetus albicilla*) in Sweden. Ambio, 31: 386–403.

ICES. 1999. Report of the Working Group on Seabird Ecology. ICES CM 1999/C:5.

ICES. 2000. Report of the ICES Advisory Committee on the Marine Environment, 1999. ICES Cooperative Research Report, 239: 109–112, 228–253.

ICES. 2001. Report of the ICES Advisory Committee on Ecosystems, 2001. ICES Cooperative Research Report, 249: 47–50.

ICES. 2002. Report of the Working Group on Seabird Ecology. ICES CM 2002/C:4.

Mattig, F.R., Rosner, H.U., Giessing, K., and Becker, P.H. 2000. Environmental chemicals in eggs of dunlins (*Calidris alpina*) from northern Norway compared to eggs of coastal bird species breeding in the Wadden Sea. Journal für Ornithologie, 141: 361–369.

Munoz, J., and Becker, P.H. 1999. The kelp gull as bioindicator of environmental chemicals in the Magellan region. A comparison with other coastal sites in Chile. Scientia Marina, 63: 495–502.

Norstrom, R.J., Simon, M., Moisey, J., Wakeford, B., and Weseloh, D.V.C. 2002. Geographical distribution (2000) and temporal trends (1981–2000) of brominated diphenyl ethers in Great Lakes herring gull eggs. Environmental Science and Technology, 36: 4783–4789.

OSPAR. 1997. JAMP guidelines for monitoring contaminants in biota. OSPAR Commission, London.

Thyen, S., and Becker, P.H. 2000. Aktuelle Ergebnisse des Schadstoffmonitorings mit Küstenvögeln im Wattenmeer. Vogelwelt, 121: 281–291.

Thyen, S., Becker, P.H., and Behmann, H. 2000. Organochlorine and mercury contamination of little terns (*Sterna albifrons*) breeding at the western Baltic Sea, 1978–96. Environmental Pollution, 108: 225–238.

Weseloh, D.V., Hughes, K.D., Ewins, P.J., Best, D., Kubiak, T., and Shieldcastle, M.C. 2002. Herring gulls and great black-backed gulls as indicators of contaminants in bald eagles in Lake Ontario, Canada. Environmental Toxicology and Chemistry, 21: 1015–1025.

6.9 Initial development of EcoQ element (i) Plastic particles in the stomachs of North Sea seabirds

Source of information

The 2003 Report of the Working Group on Seabird Ecology (WGSE) (ICES CM 2003/C:03).

Summary

Plastic particles in the stomachs of beach-washed northern fulmars offer a reliable monitoring tool for changes in the level of plastic particle pollution at sea, and the number of these particles has been suggested as an EcoQ metric. Current studies in the Netherlands show that around 60% of fulmars have ten or more plastic particles in the stomach. The performance of this metric should be a downward trend. ICES can support the proposed objective of less than 2% of fulmars, out of a sample of fifty or more, having ten or more particles in the stomach. ICES advises also to gather data on the nature of the particles in the stomach, given indications that different types of litter (industrial plastic particles, user plastic particles) show different trends. The "chemical material" found in many fulmar stomachs should receive further analysis to determine its nature, likely origins, and toxic hazard.

Recommendations and advice

In 2001, ICES suggested that a suitable target for this EcoQO would be "a maximum of no more than 2% of individuals having ten or more plastic particles within a sample of at least 50 northern fulmars". Analyses carried out since that advice was given support this suggestion. ICES recommends further that more information be gathered on the nature of the particles in the stomach.

Scientific background

6.9.1 Basis of the EcoQO

This EcoQ element was initially considered by ACE in 2001 and much of the relevant background information can be found in the 2001 ACE report (ICES, 2001). Additional information is reviewed here.

The data presented in the report by Van Franeker and Meijboom (2002) show significant trends in plastic contamination of fulmar stomachs over the study period (1982–2000). This demonstrates that sampling fulmar stomachs can provide data on plastic pollution not available from any other current research programme, at least to monitor temporal trends, although not necessarily informing about spatial patterns.

Such monitoring would not only indicate changes in the abundance of plastic litter, but also increase public awareness of the fact that environmental problems from marine litter persist even when larger plastic items are broken down to sizes below the range of normal human perception.

6.9.2 Robustness of the proposed EcoQO

The skewed nature of numbers of plastic items per stomach makes the use of geometric means rather than arithmetic means more appropriate for statistical analyses. Since it is unclear whether the incidence (frequency of presence), the mean number of items, or the total biomass of plastic provides the most appropriate measure, it seems sensible, as in the Van Franeker and Meijboom (2002) report, to record each of these statistics.

Residence times of plastic in fulmar stomachs are not known, but are likely to be in the order of many weeks or months, and possibly even years. These long periods achieve an integration of plastic contamination over extended periods prior to the death of the birds collected on beached bird surveys. This long sampling period is a positive attribute in terms of generating representative samples of plastic pollution over what may be highly patchy spatial and temporal distributions of plastic at sea. However, some aspects of fulmar behaviour that might affect the suitability of these birds as biomonitors of plastic pollution require further attention. For example, if fulmars move into the North Sea from (probably less contaminated) areas of the Atlantic Ocean, then the number of plastic items may be less in those birds than in fulmars that have been resident within the North Sea for a longer period. Hence, a study of geographical variation in fulmar contamination around the North Sea would be helpful in quantifying spatial patterns.

If fulmars show varying amounts of plastic according to their local origin, then the contamination level might be susceptible to variation related to weather, since weather may influence regional movements of fulmars. However, the data presented by Van Franeker and Meijboom (2002) show no strong evidence for year-to-year fluctuations in fulmar contamination, so this effect may be negligible in practice.

6.9.3 The use of the EcoQO in the North Sea

Stomach contents analysis of beach-washed northern fulmars offers a reliable monitoring tool for changes in the abundance of small fragments of plastic litter at sea (Van Franeker and Meijboom, 2002).

6.9.3.1 Proposed metric

The metrics should be the number and mass of plastic particles of each defined type in the stomachs of samples of 50 to 100 beach-washed northern fulmars:

1) industrial plastic particles;
2) user plastic particles;
3) mass of "inert" chemical material.

6.9.3.2 Reference levels

Since plastic is a relatively recent human invention, the natural (reference) level of this metric would be that there would be no plastic in any stomachs.

6.9.3.3 Current levels

Around 60% of the fulmars washed ashore on the Dutch coast contained ten or more plastic particles (Van Franeker and Meijboom, 2002).

6.9.3.4 Objective

The objective should be to achieve a target of as little plastic in fulmar stomachs as possible. Given the fact that ICES considers the current level of plastic pollution as too high, the performance of the metric should be a downward trend in the number of particles in the stomach of fulmars. In this way, this monitoring would be an indication of the success of management measures aimed at reducing litter in the marine environment.

ICES would support the objective proposed after the pilot study by Van Franeker and Meijboom (2002) of less than 2% of northern fulmars having ten or more plastic particles in the stomach from a sample of 50 to 100 beach-washed northern fulmars. Given the current level of around 60% of northern fulmars with more than ten plastic particles in the stomach, considerable effort will be required to reduce marine plastic pollution to a level that achieves this target.

6.9.3.5 Sampling requirements

ICES recommends that an annual monitoring of fifty or more fulmars from different selected parts of the coastlines in the North Sea should be carried out in order to establish patterns of geographical variation. The fulmars should be collected during winter from areas of the North Sea where such sampling can be achieved as part of beached bird surveys.

Although the initial suggestion for this EcoQO (ICES, 2001) did not discriminate between "industrial plastic pellets" and "used plastic fragments", Van Franeker and Meijboom (2002) clearly show trends in opposite directions for these two types of plastic. Any EcoQO for plastic at sea should therefore take note of the differences between these two categories.

The "chemical material" found in many fulmar stomachs should receive further analysis to determine its nature, likely origins, and toxic hazard.

6.9.3.6 Historic trajectory and its historic performance

The published accounts of changes in amounts of plastic particles in seabirds show evidence of increases up to the present time. Some accounts suggest that in the last few years the amounts of industrial plastic particles have decreased in some areas, but that user plastic particles have continued to increase in frequency (Van Franeker and Meijboom, 2002; Vlietstra and Parga, 2002). The period of time that plastic remains in seabird stomachs is not well known, but certainly can be many months. Therefore, the level of contamination is unlikely to fluctuate widely over short periods. Furthermore, northern fulmars range widely over large areas (thousands of square kilometres) and are believed to ingest plastic in error for food particles. This will inevitably result in the loads carried by fulmars representing an average contamination over long periods of time and over large areas. As a result, ACE expects that the levels of contamination will be robustly measured, without misleading spatial or temporal variation. The performance of this EcoQO should not result in False Alarms, but should indicate long-term trends over periods of several years to several decades.

At present, an EC-funded project (EU Interreg IIIB J-No. 1–16–31–7-502–02 "Save the North Sea") is investigating plastic particles in northern fulmars in seven European countries (see http://marine-litter.gpa.unep.org/regional/Nederland/nl_results.htm). This should provide further data to confirm the suitability of this EcoQO.

6.9.4 References

ICES. 2001. Report of the Advisory Committee on Ecosystems, 2001. ICES Cooperative Research Report, 249: 51–52.

Van Franeker, J.A., and Meijboom, A. 2002. LITTER NSV, Marine litter monitoring by northern fulmars; a pilot study. Wageningen, Alterra, Green World Research. Alterra-rapport 401. 72 pp.

Vlietstra, L.S., and Parga, J.A. 2002. Long-term changes in the type, but not amount, of ingested plastic particles in short-tailed shearwaters in the southeastern Bering Sea. Marine Pollution Bulletin, 44 : 945–955.

6.10 Initial development of EcoQ element (j) Local availability in the North Sea of sandeels for black-legged kittiwakes

Source of information

The 2003 Report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO) (ICES CM 2003/ ACE:05).

The 2003 Report of the Working Group on Seabird Ecology (WGSE) (ICES CM 2003/C:03).

The 2001 Report of the Advisory Committee on Ecosystems (ACE) (ICES Cooperative Research Report, 249: 53–54).

The 1999 Report of the Advisory Committee on the Marine Environment (ACME) (ICES Cooperative Research Report, 239: 7–10).

Recommendations and advice

The response of black-legged kittiwake breeding success to local sandeel availability is complex and non-linear. When sandeels are relatively abundant, changes in local sandeel availability are primarily driven by the environment. When sandeels are scarce, changes in availability may also be driven by fishing impacts on the sandeel stock.

New research on the links between sandeel availability and black-legged kittiwake breeding success was presented at the 2003 meeting of WGECO. Scientists at WGECO and the Working Group on Seabird Ecology (WGSE) have not yet assimilated the new research findings with those on which past advice has been based.

Before new advice on this EcoQO is formulated, ICES will continue to examine the relationship between black-legged kittiwake breeding success, sandeel availability, and sandeel fishing. The outcome will be included in the 2004 advice on this topic.

6.11 Initial development of EcoQ element (k) Seabird population trends in the North Sea as an index of seabird community health

Source of information

The 2003 Report of the Working Group on Seabird Ecology (WGSE) (ICES CM 2003/C:03).

Summary

Healthy seabird communities in the North Sea are characterized by significant population changes due to natural factors. However, major changes could be indicating changes in the environment, possibly induced by human activities. A metric for ecological quality is change in breeding numbers of seabirds of selected species at selected key colonies. An objective for this metric that has been proposed is ≤20% decline in population size within any period of 20 or more years.

ICES could agree, for the time being, to use such an objective, but it is clear that the EcoQO needs to be further developed for individual species in order to indicate declines at a population level, or even at colony levels. The objective should not be considered as a target to strive towards, but as a level where possible impacts of human activities become apparent and, therefore, there is a need for further research.

It is expected that abundant and relatively widespread species such as the black-legged kittiwake and gannet (pelagic surface-feeding), common guillemot (pelagic pursuit-diving), common tern (coastal surface-feeding), and common eider (nearshore benthos-feeding) might be particularly useful as primary targets for assessing this EcoQO.

Recommendations and advice

ICES recommends that assessments be made concerning the extent to which the present level of monitoring in the North Sea countries is adequate to provide the data to fulfil the proposed EcoQO. This assessment should take into account the representativeness of the sample of populations in relation to the overall seabird community. It should be investigated whether the proposed selection of species reflects the main ecological groups of seabirds in terms of their range of diets, habitat use, and life-history strategies.

As a first step, ICES recommends that a detailed analysis of trends in individual colonies of kittiwakes be carried out on the existing data (predominantly from UK seabird surveys and monitoring). This could provide a better understanding of how colony selection may be made in order to render an EcoQ metric that is representative for the North Sea as a whole.

Scientific background

6.11.1 Basis for the EcoQO

At North Sea latitudes, environmental variability is expected to be relatively large and, hence, most seabird populations will be either increasing or decreasing in numbers at any one time. Consequently, healthy seabird communities in the North Sea are also characterized by significant population changes within limits set by natural factors.

Obviously, there is no need to initiate intensive research aimed at explaining all changes in seabird numbers. The magnitude of such changes may, nevertheless, serve as an adequate EcoQO for the intrinsic health of seabird communities. This is based on the simple assumption that a pronounced negative trend in the population of any seabird species could indicate that there is an undesirable effect of human activities. In other words, when a certain level of population change is reached, the public would regard this as a reduction of ecological quality. Ideally, and as a precautionary measure, reaching such a threshold should then trigger adequate studies targeted at revealing its underlying causes. If the change proves to be an undesired consequence of human activities, any useful mitigating measures should be identified and implemented. In some cases, monitoring the effect of these measures may benefit from defining additional and more specific EcoQOs for the seabird populations and/or environmental factors involved.

6.11.2 Robustness of the proposed EcoQO

On a short-term scale, seabird breeding population size may not be very sensitive to environmental change. Due to the longevity and delayed maturity of most seabirds, several years are usually needed before changes in their reproduction or immature survival rates affect their breeding numbers. Nevertheless, changes in breeding population sizes are reasonably good indicators of important changes in seabird community structure, where density dependent effects may easily reduce the usability of other population parameters. Numbers of breeding birds are relatively easy to count extensively at colonies throughout the geographical range of the populations.

However, if this EcoQ metric is assessed against the ICES criteria for good EcoQ metrics, it is clear that no manageable human activity, to which the metric is tightly linked, is identified. The metric would also be responsive to a number of different environmental factors, including factors not affected by human activities. Therefore, seabird breeding population size could be more useful as an ecological quality to monitor, and changes beyond what could be expected could serve as a warning system for change.

Also, a number of seabirds have become more common during the last 100 years, due to the availability of discards from fisheries (OSPAR, 2000). Lowering discard levels could therefore have a profound impact on these populations.

6.11.3 The use of the EcoQO in the North Sea

6.11.3.1 Proposed metric

The proposed metric is a "change in breeding numbers of seabirds of selected species at selected colonies". It is expected that abundant and relatively widespread species such as the black-legged kittiwake and gannet (pelagic surface-feeding), common guillemot (pelagic pursuit-diving), common tern (coastal surface-feeding), and common eider (nearshore benthos-feeding) might be particularly useful as primary targets for assessing this EcoQO. These species are all relatively specialized in their food choice and considered to reflect closely important changes in their very different foraging habitats (e.g., Lloyd *et al.*, 1991).

6.11.3.2 Reference levels and current levels

The reference level for this EcoQ element is a zero change in breeding bird numbers in a twenty-year period. The current numbers of breeding birds vary inter-annually in response to a diversity of natural as well as anthropogenic influences. Assessment of this EcoQO therefore requires assessment of a 20% decline for a twenty-year or more time series, and quality control techniques (such as CUSUM) provide a tool for doing this.

6.11.3.3 Objective

An objective that has been proposed is ≤20% decline in the population size within a period of ≥20 years. ICES could agree, for the time being, to use such an objective, but it is clear that the EcoQO needs to be further developed for individual species in order to indicate declines at a population level, or even at colony levels. The objective should not be considered as a target to strive towards, but as a level where possible impacts of human activities become apparent and, therefore, there is a need for further research.

6.11.3.4 Monitoring requirements

The data sampling procedures for monitoring breeding populations of different seabird species are standardized

(Walsh *et al.*, 1995) and applied in much the same way across most North Sea countries.

There is a need to assess in more detail the extent to which the present level of monitoring in North Sea countries is adequate to provide the data to fulfil the proposed EcoQO. This assessment should take into account the representativeness of the sample populations in relation to the overall seabird community.

In particular, it should be investigated whether the selection of species reflects the main ecological groups of seabirds in terms of their range of diets, habitat use, and life-history strategies (e.g., Anker-Nilssen *et al.*, 1996), and whether the national programmes need to be adjusted in order to ensure that a sufficient part of the North Sea population of the selected target species is being monitored.

6.11.3.5 Historic trajectory and its historic performance

To date, no one has tried to quantify the overall change in numbers within the entire North Sea area for any seabird species. The qualitative review of the status of North Sea seabirds (ICES, 2002) concluded that among the 23 breeding species examined, eight were assessed as increasing (northern gannet, great cormorant, common eider, Arctic skua, great skua, Mediterranean gull, common guillemot, and Atlantic puffin) and six as being more or less stable (herring gull, lesser black-backed gull, great black-backed gull, roseate tern, little tern, and black guillemot). The remaining nine species (northern fulmar, European shag, black-headed gull, mew gull, black-legged kittiwake, sandwich tern, common tern, Arctic tern, and razorbill) were judged to be decreasing by between 1% and 5% per year.

Being one of the candidates for the selection of target species, the black-legged kittiwake may here serve the purpose of illustrating how the historical records of their population trends would perform in relation to the objective for this EcoQO (Hit, Miss, or False Alarm). Based on the population numbers and trends summarized for black-legged kittiwakes by ICES (2002), the North Sea population of this species is estimated at about 302,000 pairs in 2000. It had dropped by 41.5% (from an estimated 517,000 pairs) during a fifteen-year period (1985–2000), a decrease well beyond the EcoQO target level. This rather dramatic change was, however, not reflected by the population trends for all colonies.

Almost 90% of the North Sea kittiwakes breed in the UK, where their numbers have been monitored in many colonies. Most colonies decreased by more than 20% over this period, but a few decreased by less than 20%. These few colonies thus missed the EcoQO alarm level despite the overall population having declined by more than twice this level. This illustrates the need to have a representative sample of monitoring sites or areas distributed across the main breeding areas in order to avoid Misses or False Alarms. However, the few colonies that did not show a decline by as much as 20% were mainly small colonies.

ICES recommends that a detailed analysis of trends in individual colonies of kittiwakes should be carried out on the existing data (predominantly from UK seabird surveys and monitoring). This could provide for a better understanding of how colony selection may be made to render an EcoQ metric that is representative of the North Sea as a whole.

6.11.4 References

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

ICES. 2002. Report of the Working Group on Seabird Ecology. ICES CM 2002/C:4.

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

OSPAR. 2000. Quality Status Report 2000, Region II–Greater North Sea. OSPAR Commission, London.

Tucker, G.M. and Heath, M.F. 1994. Birds in Europe. Their conservation status. BirdLife International, Cambridge.

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

6.12 Further development of EcoQ element (l) Changes in the proportion of large fish

Source of information

The 2003 Report of the Working Group on Fish Ecology (WGFE) (ICES CM 2003/G:04).

The 2003 Report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO) (ICES CM 2003/ACE:05).

Summary

The proposed metrics, namely, changes in the proportion of large fish and, hence, the average weight and average maximum length of the fish community, have been extensively investigated by the Working Group on Fish Ecology and the Working Group on Ecosystem Effects of Fishing Activities.

The proposed metrics and fishing activity are related. However, the probability of detecting short-term changes in the metrics is low, which makes it difficult to establish EcoQOs on which managers can act. More importantly, the available evidence suggests that the response time of the metrics to changes in fishing effort is considerable,

particularly so during the process of recovery. Also, the existing time series are generally too short to establish meaningful reference levels. Only one survey (the Scottish August Demersal Survey) goes back to the early years of the 20th century, but this survey has been terminated recently and comparable information will not be available in the future.

A more fundamental problem is that the values of the metric are highly dependent on the gear used, on the area where the gear is applied, and on the selection of species that is used in the analysis. Thus, if an EcoQO is set for this metric of ecological quality, it applies to very specific conditions, and no EcoQO could therefore be expected to represent the total community under pressure.

Finally, the proposed metrics have unwanted properties, because the community might be manipulated in such a way that the metric might show an apparent improvement, whereas in fact ecological quality might have declined. The following argument underlines this. While the abundance of large fish has declined because of fishing, the abundance of small fish has increased, probably as a result of lesser predation. Thus, the size composition of the community might be “restored” by more extensive fishing on small fish, which would be opposite to conservation goals. Thus, an EcoQO based on these metrics should only be used together with other EcoQOs.

While these problems make this EcoQ element less suitable as an EcoQO, the metric has some advantage as a qualitative measure of ecological quality and as such might be used in a management context, but without the specific connotations linked to the concept of EcoQOs.

Recommendations and advice

At this stage, the proposed metrics provide no clear basis for establishing an EcoQO for use in short-term fisheries management. However, they might be used more qualitatively as indicating deterioration or recovery of the ecological quality of fish communities over longer time spans.

To facilitate future interpretation of spatio-temporal trends in these metrics, it is of importance that satellite-based information on fishing activity, as presently collated by national inspectorates, be made available for scientific purposes.

Scientific background

6.12.1 Basis for the EcoQO

The basis for this EcoQ element is that overall fishing pressure exerted on the entire fish community may change its composition in undesired directions and that these effects are not accounted for when fish stocks are managed on a species-by-species basis. The element consists of two metrics:

1) average weight of fish in the community;
2) average maximum length of a fish in the community.

Although both metrics are considered to be indicators of the proportion of large fish in the community, it should be realized that they represent different aspects of the community and are complementary in that respect. The average weight of a fish in the community represents changes in the size structure of the community, whereas the average maximum length represents changes in the species composition (Piet, 2001). However, in practice these two metrics are interrelated. A change in species composition would also imply a change in average weight.

The criteria against which to evaluate metrics, as previously set forth by ACE (ICES, 2002), are that EcoQs should be:

1) relatively easy to understand by non-scientists and those who will decide on their use;
2) sensitive to a manageable human activity;
3) relatively tightly linked in time to that activity;
4) easily and accurately measured, with a low error rate;
5) responsive primarily to a human activity, with low responsiveness to other causes of change;
6) measurable over a large proportion of the area to which the EcoQ element is to apply;
7) based on an existing body or time series of data to allow a realistic setting of objectives.

6.12.2 Evaluation

The further development of this EcoQ element using the above metrics has been extensively addressed by the Working Group on Fish Ecology and the Working Group on Ecosystem Effects of Fishing Activities. The available data for the North Sea and the Celtic Sea indicate that fishing has caused the observed decline in the metrics, while this could not be shown for the Portuguese shelf, the northern coast of Norway, or the West Spitsbergen shelf.

Long-term temporal trends of the proposed metrics for three selected North Sea surveys are shown in Figure 6.12.2.1. These all clearly indicate consistent declines over a time period during which the fishing mortality of most commercial fish species has been increasing. Nevertheless, the level of the declines differs among surveys, as do the actual values. This underlines a fundamental problem of these metrics: they represent survey- and area-specific metrics that depend largely on the part of the community that is effectively sampled.

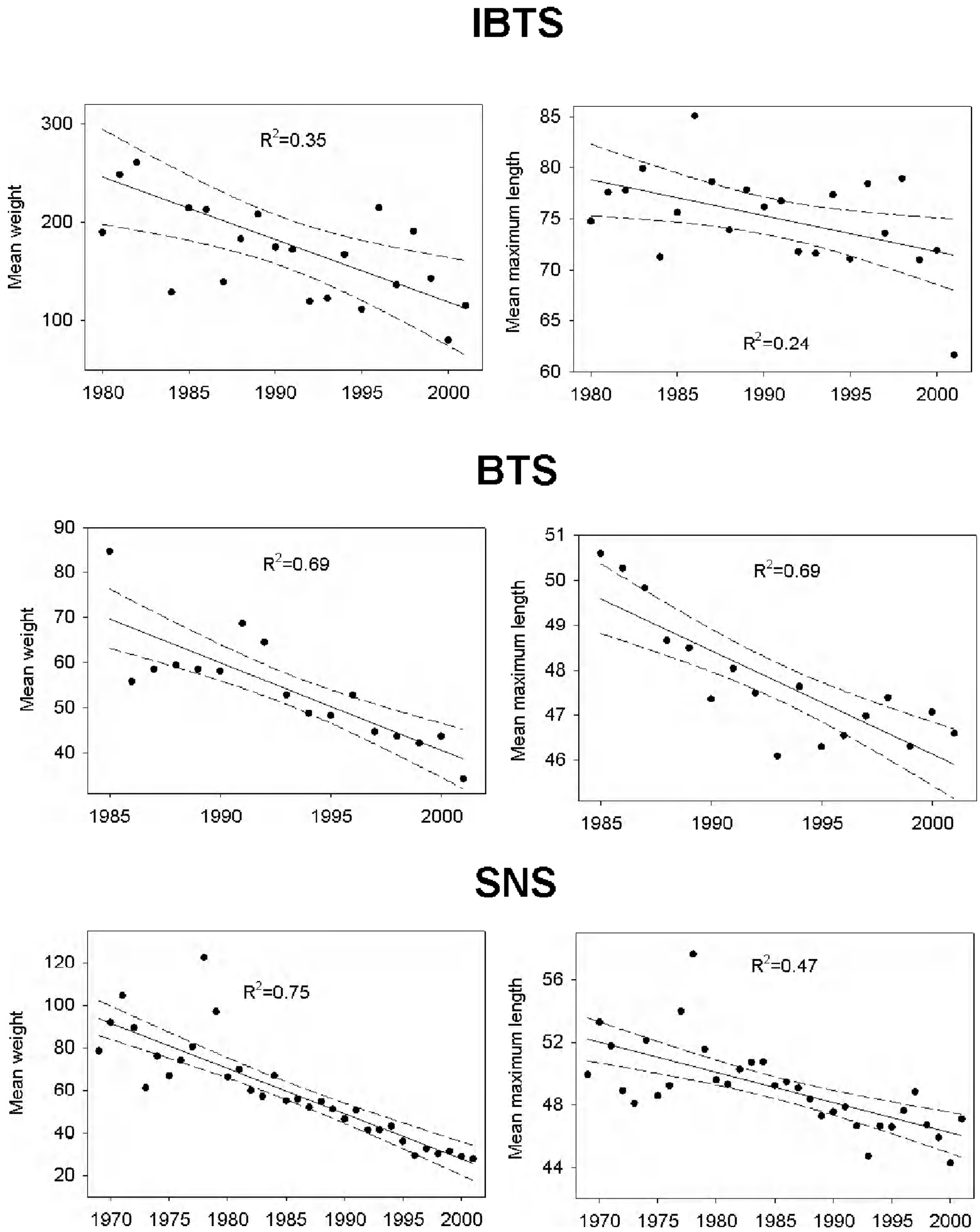


Figure 6.12.2.1. Long-term trends (with confidence limits) in the proposed EcoQ element metrics mean weight and mean maximum length of the fish community for three North Sea surveys.

Thus, selecting a particular survey and area may not ensure that overall ecological quality at the fish community level is maintained. A study of North Sea data shows that, in addition to a decrease of larger fish, small fish have increased in absolute numbers, probably related to less predation. In this situation, an EcoQO could be met by increasing the industrial fishery.

Therefore, ICES concludes that the range of possible management measures to guarantee that the objective is met makes it quite clear that the two metrics are not particularly useful as descriptors of ecological quality of the fish community if not backed up by additional descriptors, because advice to increase exploitation to meet quality objectives is probably not intended by the framework.

Overall, the two metrics appear as yet to be neither appropriate to implement as fish community metrics as part of an EcoQO, nor to serve as a basis on which to define a global North Sea reference level for management.

It is emphasized that there is a strong need for high quality fishing effort data to be able to assess the relation between these metrics and fishing activity. Fishing effort data on the spatial scales on which the metrics are based do not exist. A comprehensive data set could be established if the satellite-based position registrations collated by the respective national inspection services were made available for scientific research. This was also noted by ACE in 2002 (ICES, 2002).

For an EcoQ element metric to be useful in management, it must be possible for the managers to detect changed trends in the metric on time scales comparable to the time scales in fishery management, which usually is one year but probably may be extended somewhat for many species. The power of the International Bottom Trawl Survey (IBTS) demersal survey (which is one of the best surveys operative for the North Sea on which to base community metrics) to detect changes in several proposed metrics does not seem promising. After three years, the statistical power to detect expected trends in potential EcoQ element metrics was hardly greater than the significance level of 5%. Even after ten years, the power was still typically less than 50%. The number of years needed to detect changes with 90% probability in average weight and average maximum length given the observed trends is 16 and 14 years, respectively. From this analysis, the metrics seem to fail on criteria 3 and 4 above. No attempt has yet been made to determine the historic performance (Hit, Miss, or False Alarm) of the two EcoQ element metrics.

Although some reference level might be set using data from the Scottish August Demersal Survey, which goes back sufficiently long in time to refer to a less heavily exploited North Sea (from 1925), this time series was terminated in 1997 and no new information will become available. The metrics, therefore, seem to fail also on criterion 7. In addition, changing environmental conditions may lead to changes in these metrics that may invalidate reference levels based on previous and different environmental conditions, and the proposed metrics seem to fail also regarding criterion 5.

6.12.3 Evaluation of RIVO report on the relationship between fish community EcoQO indicators and fishing effort

This report (Piet, 2002) investigates spatial variations in these metrics in relation to historic differences in fishing effort. However, this analysis is not based on the metrics themselves, but on the temporal trends during periods within which effort was assumed to be constant. These slopes of the changes in average weight/average maximum length provide a completely different metric from the one proposed for the EcoQO. Although these slopes appear to be related to historic levels of fishing effort, the interpretation of the results is not straightforward. Also, it has little bearing on the EcoQ element.

6.12.4 Time scales of EcoQ element metrics and time scales of management decisions

As a rule, the time horizon for fisheries management is one year. Therefore, for EcoQOs to be useful for management, changes in the metrics should be detected on a comparable time scale. However, metrics in which changes can only be detected on longer time scales could still be important, even if they only provide a qualitative measure of ecological quality and may not be used within a rigid EcoQO framework. Evaluating these changes should be made part of routine assessments, even if there is no mechanism of linking them to the present-day quota-setting procedures.

6.12.5 Conclusion

Although considerable advances have been made in analysing existing survey data and further analyses are unlikely to change the present evaluation, it does not seem appropriate to implement an EcoQ element based on the two proposed metrics in a short-term management framework. On the contrary, there are problems with time series on which to establish an EcoQO, and response times and changing environmental conditions may have a large and unpredictable influence compared to the human activity. Also, the logic of the proposed metrics may fail when predation effects are taken into account: if large fish are severely reduced by fishing, the proposed metrics may as well suggest fishing down small fish as reducing fishing effort on large fish.

It also may be problematic to base a metric on both demersal and pelagic fish, since the life history and migration may differ very much between these two groups.

As noted in the 2002 ACE report and in the WGECO report, simulation studies should be conducted in order to study the behaviour of the proposed EcoQ element metrics.

6.12.6 References

ICES. 2002. Report of the ICES Advisory Committee on Ecosystems, 2002. ICES Cooperative Research Report, 254: 62, 74–75.

Piet, G.J. 2001. Development of ecological quality objectives for the North Sea fish community. Working paper for the 2001 meeting of the Working Group on Ecosystem Effects of Fishing Activities.

Piet, G.J. 2002. A study on the relationship between fish community EcoQO indicators and fishing effort (INTERNAT-NZM-DNZ). RIVO Report No. C057/02. RIVO, IJmuiden, The Netherlands.

6.13 Initial development of EcoQ elements (o) Density of sensitive (e.g., fragile) species and (p) Density of opportunistic species

Source of information

The 2003 Report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO) (ICES CM 2003/ ACE:05).

Summary

ICES commenced the development of EcoQ elements for the density of sensitive (e.g., fragile) species and the density of opportunistic species by classifying benthic invertebrate species recorded in recent North Sea surveys as sensitive (fragile) and opportunistic.

Based on definitions developed for sensitive (fragile) and opportunistic, a total of 180 taxa were identified as meeting the criteria for sensitive species (including biogenic structure-forming species as well as those with fragile morphological features) and 69 taxa as meeting the criteria for opportunists (including opportunistic scavengers). These lists were inevitably incomplete, since the biology and life history of all species was not known. The identification and assignation of species could be progressed with the assistance of specialists at a focused workshop.

Monitoring changes in the abundance of sensitive (fragile) and opportunistic species presents many practical constraints, and present sampling schemes in the North Sea are largely inadequate to detect species-specific trends in abundance on the spatial scales required.

ICES considered five alternate approaches for developing the EcoQ elements for sensitive (fragile) and opportunistic species, from direct measurement of the absolute abundance of each sensitive or opportunistic species to an assessment of the density of a selection of indicator (sentinel) species. Direct measurements of the abundance of many species will be impractical and the power of affordable surveys to detect trends will be very poor. The most promising option may be to effectively monitor the abundance of a few indicator (sentinel) species, and this might provide a warning system to trigger further action. However, the monitoring and cost implications of this approach still need to be considered in detail. There is also a need to develop robust and objective criteria for the selection of the sentinel species.

ICES remains convinced of the importance of healthy benthic communities as part of a well-managed North Sea ecosystem. ICES is keen to further develop appropriate EcoQOs for benthic systems, but believes that this should be done in two ways: firstly, through a focus on habitat quality, and secondly, through the development of EcoQOs targeted at specific issues, such as those already adopted as part of the pilot scheme.

Recommendations and advice

ICES makes the following recommendations in relation to the development of EcoQ element (o) density of sensitive (e.g., fragile) species and (p) density of opportunistic species:

1) ICES should organize a workshop to develop criteria for, and then identify, the species that should be considered under EcoQ elements (o) and (p). This workshop should draw upon as wide a community of expertise as possible.

2) The Terms of Reference for the workshop should include identification of the species and sources of quantitative information on their historic and current abundance (spatially resolved).

3) In 2004, ICES should continue to develop metrics, reference points, and sampling protocols for elements (o) and (p), with an emphasis on identifying and developing: (1) an index of opportunists or sensitivity, (2) a metric based on the proportion of species that are opportunistic or sensitive, and (3) the density of selected indicator (sentinel) species.

Scientific background

6.13.1 Definitions of terms

The development of an EcoQ element for the density of sensitive (e.g., fragile) species and the density of opportunistic species requires that the characteristics and identities of these species are known.

ACE accepted the definitions agreed by WGECO for sensitive (fragile) species, opportunistic species, and scavenger species, as given below:

Fragile Species: Sessile and slow-moving species, often characterized by rigid bodies or tubes that are particularly sensitive to physical damage, usually with a body size >2 cm and living as epifauna or sub-surface infauna.

Note: The term "fragile species" is often used in the literature, including the ICES literature, to describe species that are vulnerable to human-induced or environmental change due to their life histories. Species such as elasmobranchs would fall into this latter definition. Here, usage of the term is restricted to those species that are physically vulnerable.

Sensitive Species: A species easily depleted by a human activity and/or, if affected, is expected to recover only over a very long period, or not at all.

Note: This definition of sensitive species is based on the OSPAR Texel/Faial criteria for the identification of threatened and declining species and habitats.

Opportunistic Species: Species with early maturation, high fecundity, and a high colonization potential achieved through intrinsic long-distance dispersal and a high reproductive rate. These characteristics allow for colonizing habitats of a temporary nature often created through physical disturbance.

Scavenger Species (Invertebrates): Opportunistic feeders that respond to chemical signals and are mobile over scales of tens of metres.

Here, species that are sensitive (e.g., fragile) and opportunistic have been identified. Until the identification of relevant species is complete, it is inappropriate to try to develop metrics and reference levels. Thus, the development of metrics and reference levels will be continued next year. However, in this text the practical considerations relating to monitoring, in particular the spatial scale required and the availability of information from existing monitoring programmes, have been reviewed.

6.13.2 Identification of species

A species list of benthic invertebrates was obtained from the data collected during the 1986 North Sea Benthos Survey (NSBS) (ICES, 1997). This survey was performed to provide baseline benthos data for ICES Divisions IVa–c, and as such it is only part of the OSPAR Greater North Sea area, which includes the Channel. Van Veen grabs and box-corers were used to sample the macrobenthos, with additional samples being collected with beam trawl/Agassis trawl. The majority of stations were consequently on soft bottoms and the resulting NSBS database typifies macrobenthic infauna from soft-bottom communities. In total, 281 stations were sampled throughout the North Sea, and 1270 taxa were identified. The species list was subsequently reduced to 709 taxa by the ICES Benthos Ecology Working Group, after removing meiofauna and other poorly sampled groups and correcting for redundancies and errors (ICES, 1997). In 2003, WGECO further reduced this list to 641 by removing taxa that had only been identified to the family or genus level when confamilial or congeneric taxa were documented. However, eighty additional benthic species were added to the list by WGECO based on their known occurrence in the OSPAR area. These included 27 hydroid species (none were identified in the 1986 species list), twelve sponges, deep-sea corals, echinoderms and other sublittoral taxa, primarily megabenthos living on hard surfaces.

The final species list comprises 721 taxa but remains biased towards species living in soft bottoms. As mentioned above, the NSBS survey did not sample the Channel, which is included in OSPAR Region II. The Channel represents a different biozoographic region from the North Sea and is likely to yield a considerable number of additional species.

This composite species list was reviewed, and taxa were classified as "Fragile", "Sensitive", "Opportunistic", and "Scavengers" following the definitions above. Designations were assigned based on life history characteristics and morphology, consulting primary publications, web-based literature, and the personal knowledge of WGECO participants (i.e., *a priori*).

In reviewing the list, a fifth category was created to identify those species that form habitat for other species, either by creating novel habitat features or through substantial modification of the local environment. These were classified as "Habitat-Forming". In total, 250 taxa were classified into one or more of the five categories. Another 379 were assessed but did not meet the criteria for classification.

The resulting lists are a conservative summary, given the incompleteness of the initial list and our inability to collect or infer the necessary life history information on 92 taxa for a more thorough assessment. Thus, these compilations are seen as provisional. Nevertheless, the species identified in Tables 6.13.2.1 through 6.13.2.5 represent the major taxa in each category.

A large number of taxa were considered fragile, i.e., their morphology and size did not protect the organism from physical damage. A total of 130 genera were identified as falling into this category (Table 6.13.2.1), with many species being excluded due to their small size conferring a perceived robustness to human-induced activities imposing physical contact (e.g., Gilkinson *et al.*, 1998). Debate within WGECO resulted in the exclusion of the crabs from this category as these are routinely picked up in trawl gear with only minor damage to their legs, which can be regenerated. Obviously in some situations, their carapace would be vulnerable to crushing. Species from hard-bottom habitats were not adequately canvassed and many are expected to be fragile (deep-water corals, sea fans, etc.).

While many of the resident species in the North Sea are vulnerable to damage, only 22 genera were classified as sensitive in the provisional list prepared by WGECO (Table 6.13.2.2). The shortness of this list is primarily due to the poor representation of species from hard-bottom areas and inadequate information on lifespans and reproductive biology.

Fifty-seven genera of benthic organisms form extensive and characteristic habitats for other organisms through their tubes, gregarious habits, and/or morphology (Table 6.13.2.3). These habitats are created at various scales ranging from millimetres to kilometres. The tubes of polychaetes and amphipods can form extensive "mats" which provide shelter for smaller organisms and food for larger ones (e.g., whales feeding on ampeliscan mats). The hydroid and bryozoan genera form turf, increasing habitat complexity and providing a settling substrate for mussels and scallops. The corals form extensive reefs (e.g., *Lophelia*, Husebø *et al.*, 2002) or "forests" (e.g., *Paragorgia*), providing shelter for larger organisms, including fish. Bivalves such as oysters and mussels form smaller reefs or beds, locally increasing biodiversity. Two of the polychaete species listed in Table 6.13.2.3 are widespread in the North Sea (*Owenia* and *Myriochele*) (ICES, 1997).

Twenty-four taxa, mostly polychaetes, were classified as opportunistic (Table 6.13.2.4). These include three, *Spio*, *Spiophanes*, and *Chaetozone*, which are widely distributed throughout the North Sea (ICES, 1997). Opportunistic species are early colonizers following disturbance and have very dynamic populations with a high degree of spatial and temporal variation. Their fast rate of reproduction and high mortality allow for rapid genetic adaptation, so species such as *Capitella* are often able to quickly colonize polluted areas (Grassle and Grassle, 1974). It is recognized that the hydrozoan families Bougainvilliidae, Clavidae, Tubulariidae, and Campanulariidae are composed of largely opportunistic fauna (Gili *et al.*, 1989; Stepanjants, 1998; Migotto *et al.*, 2001). Rapid and distant dispersal is possible by hundreds of hydroid species (Gili and Hughes, 1995) too numerous to list, although this group is often overlooked in benthic surveys due to the level of expertise required for identification. Barnacles are another group that is considered opportunistic but which are not included in the composite list. Opportunistic species may be very ephemeral, colonizing an area for only a matter of days. Thus, they are very difficult to quantify or to use as indicators, unless sampling strategies are adapted to their high turnover rate and the disturbance is uniform and widespread.

Only a small number of the benthic invertebrates were classified as scavengers (Table 6.13.2.5). These are primarily echinoderms, with a few gastropod and crab species included. Some are also predators, so their presence may not be indicative of the presence of offal or dead or dying animals. Many of the amphipods and polychaetes are known to be scavengers but they are unlikely to be able to travel large distances, so a scavenging response would not be readily detected in field situations. Kaiser and Spencer (1996) examined the behavioural response of a suite of benthic scavengers to beam trawling disturbance and were unable to detect any evidence for increased densities at three time intervals after trawling, concluding with 48 hours. Thus, species identified in Table 6.13.2.5 may require a sampling strategy tailored to their density, mobility, etc., in order to detect scavenging behaviour experimentally in the field.

Based on the data for the North Sea soft sedimentary environment provided by the NSBS database and our limited additions, a total of 180 taxa were identified as meeting the criteria for sensitive species, including biogenic structure-forming species as well as those with fragile morphological features, and 69 taxa as meeting the criteria for opportunists, including the opportunistic scavengers. For illustrative purposes, WGECO used these, albeit incomplete, lists as a starting point. In developing the lists, WGECO was hampered by the fact that the Benthos Ecology Working Group (BEWG), a group with a much greater level of expertise on the individual species, had not yet met. In addressing this issue, BEWG reviewed the work of WGECO and concluded that there were omissions from the lists. Unfortunately, they did not respond directly to the request for advice in their terms of reference, but concentrated on a critique of the WGECO evaluation. A clear conclusion was that this approach was potentially fruitful but would be best developed in a specialist, focused workshop environment.

6.13.3 Consideration of the sampling requirements and the spatial scale required

The distribution and abundance of benthic organisms are highly variable in space and time. This makes it difficult to design sampling programmes to adequately monitor abundance and distribution. In turn, these practical sampling constraints make it difficult to select appropriate reference levels for ecological objectives based on an area/time with minimal human impacts, and also to detect the magnitude and pattern of any human-induced changes away from the reference level and that would require management action.

The distribution of most benthic organisms is determined by the physico-chemical environment (Pearson and Rosenberg, 1987; Duineveld *et al.*, 1991; Snelgrove and Butman, 1994). One study found that 44% of the total variation in intertidal soft-sediment communities could be explained by the grain size of sediment (Ricciardi and Bourget, 1999). It is the dynamism and heterogeneity of this environment that causes much of the variation in the composition of benthic communities. Extrinsic forcing may alter the boundaries and physical structure of a habitat over very short temporal scales. This will alter the composition and density of the benthos and potentially can lead to problems when using fixed monitoring sites.

The design of benthic monitoring schemes will need to account for the close interaction between the physical habitat and the benthos it supports. Sampling the North Sea according to areas or grids which have no biological references, such as ICES rectangles, may not provide high quality information about population and distribution trends in the species to be monitored as the underlying cause of species distribution is not addressed. However, this approach does simplify the monitoring process.

The division of the North Sea into areas according to their hydrological, physico-chemical, and biological characteristics has been conducted (Glémarec, 1973; Dyer *et al.*, 1983; Callaway *et al.*, 2002). Although these divisions may show that the species composition and abundance are different in each area, a monitoring regime in these large areas may be at too coarse a scale to detect the impact of specific human activities on the benthos.

The use of finer-scaled habitat maps of the North Sea would provide more useful information for monitoring purposes, but these maps are not yet available (see Section 7, below). However, the cost of the increased number of sampling areas and the destructive nature of most benthic sampling techniques (Solan *et al.*, 2003) need to be considered. It may prove more useful to target specific habitats that are known to support the target species (sensitive or opportunistic) and monitor the density of these species. However, the scale of this sampling strategy may prove prohibitively expensive due to the number of habitat types. The EUNIS habitat classification scheme lists a total of 880 marine habitat types, the majority of which occur in the North Sea and will support species from our list of sensitive and opportunistic species. A more practical approach would be to focus on those areas and habitats that are affected by human activity and run a less intensive monitoring regime in habitats outside these areas.

Quantitative samples of infauna are normally taken using grabs and corers, but epibenthic and mobile species are sampled using dredges and trawls that are quantified according to swept area. The spatial segregation of each sampling point should be determined by the heterogeneity of the habitat or area, but a minimum of five samples per sampling point are normally required.

Temporal variability in benthic populations is also an important consideration and will require that the samples are taken at the same time of year. Little effort has yet been expended on identifying the number of samples required in a given area to accurately represent the habitat, or the statistical power that such a sampling regime should have in order to identify statistically significant divergence from a trend. This is essential if monitoring of the EcoQ elements for sensitive (e.g., fragile) species and opportunistic species is to provide useful information for managers on time scales appropriate to management.

Two general types of statistical sampling strategies are used in benthic studies: stratified random sampling (recommended by Underwood, 1997) and fixed stations with replicated samples (recommended by ICES, 1994). Fixed stations give a high power of the mean and variance estimate but at a single geographical point, as the natural variation over the area sampled is small compared to the number of replicates, normally five (Rees *et al.*, 1999; Tunberg and Nelson, 1998). Stratified random sampling, however, often needs a larger number of replicates within the sampling area to achieve the same power since it covers a larger area with larger variation, but it can be generalized to a larger area. The strategy is then dependent on the type of change that will be monitored. If disturbances that act on a large scale, such as eutrophication or climate change, are the focus, fixed stations can be used if it is possible to demonstrate that they show the same general trend. However, if the focus is to monitor the general change of a patchy disturbance such as trawling, stratified random sampling with a mean of five pseudo-replicates at each replicate point would be more suitable.

Benthic community abundance is much more stable over long time frames (decades) than the abundance of constituent species (e.g., Hagberg *et al.*, 2003). An analysis of these data based on total abundances shows that, if stratified random samples were used over an area large enough to cover three of the five stations (20 km × 20 km) and keep the same power as for the separate sampling stations, it would need well over 20 (20–90) replicates to cover that area. The necessary numbers of replicates are likely to be even more if single species were to be considered.

6.13.4 The adequacy of existing monitoring activities to determine their status and trends

Until recently, no regular monitoring of the infauna and epifauna across the North Sea was conducted and few long historical data sets exist. Currently, the majority of benthic monitoring is conducted in relation to activities such as aggregate extraction and the construction of offshore structures (Frid *et al.*, 2000). These studies are localized and short term, and different gears and levels of effort are used, which makes comparative studies difficult when attempting to investigate and monitor long-term trends.

ICES has conducted two large-scale benthos surveys in the North Sea. Data from the 2000–2001 survey have not yet been released, but the results of the ICES North Sea Benthos Survey of 1986 have been published (Duineveld *et al.*, 1991; Heip *et al.*, 1992; Künitzer *et al.*, 1992; Basford *et al.*, 1989, 1990, 1996; ICES, 1997), with comparative historical analysis to German museum samples from 1902–1912 (Rumohr and Kujawski, 2000). The ICES surveys followed a grid pattern of ICES rectangles, and macrobenthos species were sampled both qualitatively and quantitatively, and maps of invertebrate abundance and distribution have been produced. Several institutes were involved with the collection of samples that covered the area from between 51°N and 58°N and 3°W and 9°E.

Callaway *et al.* (2002) conducted an epibenthos survey at 270 stations throughout the North Sea using samples taken during the national third quarter 2000 groundfish surveys of five European countries (Germany, The Netherlands, England, Norway, and Denmark). The sampling design was based on a grid of 139 ICES rectangles. The diversity of benthic and fish communities was divided along 50 m, 100 m, and 200 m depth contours. However, this scale is too coarse to monitor and detect trends in opportunistic and sensitive species.

The UK National Marine Monitoring Programme (NMMP) was initiated in the late 1980s to coordinate marine monitoring in the UK among a number of organizations and has recently been expanded to include partners from the Netherlands. The NMMP has 115 monitoring stations located at estuarine, intermediate, and offshore sites around the UK and in the southern North Sea to monitor water quality, sediment chemistry, bioaccumulation in fish and shellfish, and benthic communities. Their advice for position fixing at estuarine sites is that the radius of the sampling should be within 20 m, whilst at intermediate and offshore sites, the sampling area should have a radius of 50 m (NMMP, 2001). Five replicate samples are recommended at each site to establish the abundance and biomass of invertebrates. The results of the NMMP benthos survey suggested that there was no evidence of pollution impacts at many of the sites and that natural variation between east and west coast faunas was likely to be responsible for much of the measured variation in biomass and community structure. This result does not mean that there was no effect of pollution, as the power of the sampling may not have been high enough to detect effects. Rees *et al.* (1999) considered that too few stations in the North Sea were sampled during the NMMP to delineate patterns in the distribution of infaunal assemblage types compared to the ICES 1986 North Sea Benthos Survey.

6.13.5 The basis for ICES advice based on scenario considerations on the applications of possible EcoQOs

6.13.5.1 Introduction and approach

Scenario: "An outline or model of an expected or supposed sequence of events".

Previously, ACE has drawn attention to the difficulties of applying the EcoQO approach to the benthos (ICES, 2001, 2002). These concerns have related both to the practical issues relating to sampling and the establishment of an appropriate reference level. A further area of concern is the formulation of a tractable objective. In this section, consideration of a number of possible objectives based on the "density of …" elements advanced for the benthic EcoQO issue is begun. As previously noted (ICES, 2002), the general considerations apply to both "density of sensitive (e.g., fragile) species" and "density of opportunistic species", only the species' identities involved will differ.

6.13.5.2 Scenarios of possible models for the application of EcoQOs for elements (o) and (p)

As a starting point, five scenarios are employed that use different possible EcoQOs covering these elements. There was no reason to adopt different approaches for the two elements as they only differ in the identity of the species concerned. The scenarios are described in decreasing order of their information requirements.

Scenario 1: Whole list and whole North Sea.

This approach is the most direct application of the proposed EcoQ elements, with the density of each sensitive/opportunistic species being monitored with the objective of maintaining each at some target level relative to a reference/baseline level. Our analysis of the ICES benthos database shows 179 species that fulfill the criteria for being considered under these two elements. Most of these species are restricted to certain assemblages (Tables 6.13.2.1, 6.13.2.2, and 6.13.2.4) and these, in turn, are associated with particular aspects of the physical benthic environment, i.e., habitats (Künitzer *et al.*, 1992). As such, any EcoQO of North Sea density would have to be based on abundance of the species, weighted by the natural distribution of the different habitat types. Leaving aside the often repeated caution about the practicality of actively managing human activities to give a resultant abundance of benthos, this implies a massive workload.

Scenario 2: Whole list by habitat/assemblage.

Considering a reference level and target (EcoQO) for each species within each of the habitat/assemblages is intuitively a more ecologically realistic approach. It obviates the need to carry out any form of weighting to the data, but increases the number of targets to be monitored and managed.

Scenario 3: An index of opportunists or sensitivity.

Given that the sampling methods used for marine benthos yield information on all the taxa present, there is no additional cost in gaining data on all species. The high cost of Scenarios 1 and 2 arises from the costs of setting and managing for the large number of EcoQOs. An approach that reduces this cost, and may increase the communicability of such a large body of information, would be an index describing the status of the species of concern. A metric, for example, based on the proportion of individuals in the assemblage that are "opportunists" or "sensitive", would serve this purpose.

Such a metric, while increasing communicability, will be much more difficult to manage for. For example, if the proportion of sensitive species were to decrease, this could be the result of damaging activities or the result of some process benefiting other components of the community, for example, something as simple as a good recruitment event.

Scenario 4: The proportion of all species.

The proportion of the species present that are opportunistic/sensitive provides a less labour-intensive metric than the proportion of individuals that are allocated to a particular category. However, it is likely to be much less sensitive to changes in the status of the system and may have a very high False Alarm rate.

Scenario 5: The density of a selection of sentinel species.

The concept of using sentinel or indicator species is well established. For example, in the terrestrial environment, birds are now widely used in this role. They are relatively easy to census (often using volunteers) and have a high value to many stakeholder groups. Their ecological role near the top of the food chain also increases their utility as ecosystem indicators. Cold-water corals could be strong candidates for a benthic sentinel species, as could other epibenthic sessile forms such as seapens. The advantage of this approach is its likely public acceptability, ease of communication, and direct links to physical damage to benthic communities.

6.13.5.3 Overview of scenarios examined

Clearly, scenarios 1 and 2 are impractical but serve to highlight some of the difficulties of setting species-based EcoQOs in ecologically diverse systems like the marine benthos. Scenarios 3 and 4 seek to use indices to summarize the information at a community level. As such, they hide much of the detail and will be difficult to manage for. Scenario 5 looks like a promising approach in terms of developing a warning system that might trigger further action, but again will be difficult to manage for directly.

Table 6.13.2.1. Provisional list of fragile benthic species. Species were drawn from the 1986 North Sea Benthos Survey list of macrobenthos (ICES, 1997) with the addition of common species known to occur in the area but not identified in that survey (indicated by *). References marked (+) indicate that the classification has been inferred from confamilial or congeneric species.

Phylum	Species	References
Annelida	*Ampharete acutifrons* (plus three congeneric species)	http://www.jncc.gov.uk/mermaid/biotopes/459.htm
	Amphicteis gunneri	Fauchald and Jumars, 1979
	Amythasides macroglossus	Fauchald and Jumars, 1979
	Eclysippe vanelli	biodiversity.uno.edu/~worms/translation/j-samythella.html
	Melinna cristata	http://www.jncc.gov.uk/mermaid/biotopes/459.htm
	Paramphinome jeffreysi	http://www.personal.cityu.edu.hk/~bhworm/errant/amphinomidae.htm
	Pseudeurythoe hemuli	http://www.personal.cityu.edu.hk/~bhworm/errant/amphinomidae.htm
	Aphrodita aculeata	Kaiser and Spencer, 1996
	Gattyana cirrosa	Fauchald and Jumars, 1979
	Apistobranchus tullbergi	Fauchald and Jumars, 1979
	Capitella capitata	Pearson and Rosenberg, 1978; Grassle and Grassle, 1974; Frid *et al.*, 2000
	Ophryotrocha longidentata	http://www.personal.cityu.edu.hk/~bhworm/errant/dorvilleidae.htm
	Protodorvillea kefersteini	http://www.personal.cityu.edu.hk/~bhworm/errant/dorvilleidae.htm
	Eunice pennata	http://www.ceab.csic.es/~dani/Table1.pdf+Eunice%2Bpennata&hl=en&ie=UTF-8
	Marphysa belli (plus one congeneric species)	Fauchald and Jumars, 1979
	Microphthalmus listensis	Grassle and Grassle, 1974
	Hyalinoecia tubicola	Fauchald and Jumars, 1979
	Myriochele sp.	Fauchald and Jumars, 1979

Table 6.13.2.1. Continued.

Phylum	Species	References
Annelida (continued)	*Owenia fusiformis*	http://www.mms.gov/intermar/SECTION6.PDF
	Lagis koreni	http://www.marlin.ac.uk
	Pectinaria belgica (plus one congeneric species*)	Macdonald *et al.*, 1996 (+)
	Poecilochaetus serpens	Fauchald and Jumars, 1979
	Sabellaria spinulosa (plus one congeneric species*)	Macdonald *et al.*, 1996; Frid *et al.*, 2000
	Pomatoceros triqueter	http://www.marlin.ac.uk
	Serpula vermicularis	http://www.marlin.ac.uk
	Polydora caulleryi	Fauchald and Jumars, 1979
	Prionospio malmgreni (plus one congeneric species)	Pearson and Rosenberg, 1978
	Scolelepis squamata	http://www.marlin.ac.uk
	Lanice conchilega	Macdonald *et al.*, 1996
Bryozoa	*Alcyonidium diaphanum* *	Porter *et al.*, 2002
	Pentapora foliacea *	Macdonald *et al.*, 1996
	Cellaria fistulosa	http://www.marlin.ac.uk
	Cellepora pumicosa (plus one congeneric species*)	Goldson *et al.*, 2001
	Microporella ciliata	http://www.civgeo.rmit.edu.au/bryozoa/library/macgillivray/mcgprod-175.html
	Crisia eburnea	http://www.ulstermuseum.org.uk/marinelife/bryozoa/criebu.htm
	Pyripora catenularia	http://www.nhm.ac.uk/hosted_sites/iba/bryozoa_home_page/cheilostomata/electridae/default.html
	Escharella immersa	http://www.gpi.uni-kiel.de/~ae/fotos/bryozoa/escharella/escharella2.html
	Porella concinna	http://www.ulstermuseum.org.uk/marinelife/bryozoa/porcom.htm
	Bowerbankia sp.	http://www.calacademy.org/research/izg/SFBay2K/Bowerbankia.htm
Chordata	*Branchiostoma lanceolatum*	http://www.lander.edu/rsfox/branchio.html
	Corella parallelogramma	http://www.vattenkikaren.gu.se/fakta/arter/chordata/tunicata/corepara/corepae.html
	Molgula sp.	http://www.marlin.ac.uk
	Dendrodoa grossularia	http://www.marlin.ac.uk
	Pelonaia corrugata	http://www.nioz.nl/en/deps/mee/projects/poster/magda.htm
	Polycarpa fibrosa	http://www.itsligo.ie/biomar/tunicata/POLPOM.HTM
	Styela coriacea	http://www.marlin.ac.uk
Cnidaria	*Alcyonium digitatum** (plus one congeneric species*)	Kaiser *et al.*, 2000 (+)
	*Urticina felina**	Macdonald *et al.*, 1996
	*Leptopsammia pruvoti**	Macdonald *et al.*, 1996
	*Caryophyllia pruvoti** (plus two congeneric species*)	Macdonald *et al.*, 1996
	*Lophelia pertusa**	Husebø *et al.*, 2002
	*Paragorgia arborea**	Husebø *et al.*, 2002
	*Primnoa resedaeformis**	Husebø *et al.*, 2002
	*Desmophyllum cristagalli**	Hall-Spencer *et al.*, 2002
	*Enallopsammia rostrata**	Hall-Spencer *et al.*, 2002
	*Madrepora oculata**	Hall-Spencer *et al.*, 2002

Table 6.13.2.1. Continued.

Phylum	Species	References
Cnidaria (continued)	*Solenosmilia variabilis**	Hall-Spencer *et al*., 2002
	*Paramuricea placomus**	Husebø *et al.*, 2002
	*Tubularia indivisa**	Orlov, 1997; Macdonald *et al.*, 1996
	*Obelia dichotoma**	Orlov, 1997
	*Halecium muricatum**	Orlov, 1997
	*Hydrallmania falcata**	Orlov, 1997
	*Sertularia cupressina** (plus one congeneric species*)	Pulfrich, 1996; Schmidt and Warner, 1991; Orlov, 1997
	*Eunicella verrucosa**	Macdonald *et al.*, 1996
	*Funiculina quadrangularis**	Macdonald *et al.*, 1996
	*Virgulina mirabilis**	Macdonald *et al.*, 1996
	Nemertesia antennina *	Hughes, 1975; Macdonald *et al.*, 1996
Echinodermata	*Acrocnida brachiata*	Makra and Keegan, 1999
	Amphipholis squamata	http://www.ulstermuseum.org.uk/marinelife/echinode/ampsqu.htm
	Amphiura chiajei (plus two congeneric species)	http://www.marlin.ac.uk
	Echinus esculentus (plus one congeneric species)	Macdonald *et al.*, 1996 (+)
	Psammechinus miliaris	http://www.marlin.ac.uk
	Ophiura affinis (plus four congeneric species*)	Rumohr and Kujawski, 2000
	*Ophiocomina nigra**	Rumohr and Kujawski, 2000; Kaiser *et al*., 2000
	Ophiothrix fragilis	http://www.marlin.ac.uk; Kaiser *et al.*, 2000; Macdonald *et al.*, 1996
	Crossaster papposus	http://www.marlin.ac.uk
	Brissopsis lyrifera	http://www.marlin.ac.uk; Rumohr and Kujawski, 2000
	Echinocardium cordatum (plus two congeneric species)	http://www.marlin.ac.uk
	Spatangus purpureus	http://www.marlin.ac.uk; Kaiser and Spencer, 1994
	Stichastrella rosea	http://www.itsligo.ie/biomar/ECHINODE/STIROS.HTM
	*Asterias rubens**	
	*Astropecten irregularis**	
	Luidia sp.*	
	*Lepasterias muelleri**	
	*Henricia sanguninolenta**	
	*Antedon bifida**	
Mollusca	*Acteon tornatilis*	http://www.ibss.iuf.net/blacksea/species/freelife/mollusca/gastropoda/931.html
	Anomia ephippium	http://www.gol.grosseto.it/puam/comgr/acquario/anomia_ephippium.htm
	Pododesmus patelliformis	http://www.gol.grosseto.it/puam/comgr/acquario/anomia_ephippium.htm
	Aporrhais pespelecani	http://digilander.libero.it/conchiglie/Foto/Aporrhaidae/; Kaiser *et al*., 2000
	Gari fervensis	http://www.lowtideshells.com/psa-013.htm
	Tridonta borealis (plus two congeneric species)	http://www.fegi.ru/prim/sea/m_dvu5.htm
	Colus gracilis	http://www.marlin.ac.uk
	Acanthocardia echinata (plus one congeneric species)	http://www.marlin.ac.uk

Table 6.13.2.1. Continued.

Phylum	Species	References
Mollusca (continued)	*Laevicardium crassum*	http://www.lowtideshells.com/laev_crassum.htm
	Arctica islandica	Macdonald *et al.*, 1996
	Epitonium trevelyanum	http://www.marlin.ac.uk
	Glycymeris glycymeris	Kaiser *et al.*, 2000
	Hiatella arctica	http://www.pwsrcac.org/oldsite/NIS/Smithsonian%202000/AK99_9c7.PDF
	Panomya arctica	http://www.pwsrcac.org/oldsite/NIS/Smithsonian%202000/AK99_9c7.PDF
	Saxicavella jeffreysi	http://www.pwsrcac.org/oldsite/NIS/Smithsonian%202000/AK99_9c7.PDF
	Mactra stultorum	http://www.gla.ac.uk/centres/marinestation/hydraulic.htm
	Spisula elliptica (plus two congeneric species)	http://www.marlin.ac.uk
	Mya arenaria (plus one congeneric species)	http://www.marlin.ac.uk
	Modiolus barbatus (plus one congeneric species)	http://www.marlin.ac.uk
	Musculus discors (plus one congeneric species)	http://www.marlin.ac.uk
	Mytilus edulis	http://www.marlin.ac.uk
	Lunatia poliana	Rumohr and Kajawski, 2000
	*Ostrea edulis**	
	*Crassostrea gigas** (plus one congeneric species*)	
	Philine catena (plus two congeneric species)	http://www.marlin.ac.uk
	Abra alba (plus three congeneric species)	http://www.marlin.ac.uk
	Ensis sp.	Macdonald *et al.*, 1996
	Phaxas pellucidus	http://www.marlin.ac.uk
	Solen marginatus	http://www.strandwerkgroep.org/soorten.php3?soort=3&taal=en
	Tritonia hombergi	http://www.marlin.ac.uk
	Dosinia lupinus	http://www.marlin.ac.uk
Nemertea	Nemertea indet.	http://www.teaching-biomed.man.ac.uk/bs1999/bs146/biodiversity/nemer.htm
Phoronida	*Phoronis hippocrepia* (plus one congeneric species)	http://www.marlin.ac.uk
Porifera	*Halichondria oculata** (plus one congeneric species)	Macdonald *et al.*, 1996
	*Axinella dissimilis**	http://www.marlin.ac.uk
	*Ciocalypta penicillus**	http://www.marlin.ac.uk
	*Demacidon fruticosum**	http://www.marlin.ac.uk
	*Pachymatisma johnstonia**	http://www.marlin.ac.uk
	*Polymastia mamillaris** (plus one congeneric species)	http://www.marlin.ac.uk
	*Suberites carnosus** (plus two congeneric species)	http://www.marlin.ac.uk
	*Cliona celata**	http://www.itsligo.ie/biomar/porifera/clicel.htm
Pogonophora	*Siboglinum* sp.	http://www.ucmp.berkeley.edu/annelida/pogonophora.html
Priapulida	*Priapulus caudatus*	http://www.ldeo.columbia.edu/dees/ees/life/slides/phyla/priapulida.html
Sipunculida	Sipunculida indet.	http://www.arctic.uoguelph.ca/cpl/organisms/inverts/marine_inverts/sipunculids.htm

Table 6.13.2.2. Provisional list of sensitive benthic species. Species were drawn from the 1986 North Sea Benthos Survey list of macrobenthos (ICES, 1997) with the addition of common species known to occur in the area but not identified in that survey (indicated by *).

Phylum	Species	References
Cnidaria (Anthozoa)	*Caryophyllia smithii** (plus one congeneric species)	Macdonald *et al.*, 1996
	*Lophelia pertusa**	Husebø *et al.*, 2002
	*Paragorgia arborea**	Husebø *et al.*, 2002
	*Primnoa resedaeformis**	Husebø *et al.*, 2002
	*Paramuricea placomus**	Husebø *et al.*, 2002
	*Leptopsammia pruvoti**	Macdonald *et al.*, 1996
	*Eunicella verrucosa**	Macdonald *et al.*, 1996
	*Desmophyllum cristagalli**	Hall-Spencer *et al.*, 2002
	*Enallopsammia rostrata**	Hall-Spencer *et al.*, 2002
	*Madrepora oculata**	Hall-Spencer *et al.*, 2002
	*Solenosmilia variabilis**	Hall-Spencer *et al.*, 2002
Mollusca	*Armina loveni*	http://www.pictonb.freeserve.co.uk/nudibranchs/armlov.html
	Acanthocardia echinata (plus one congeneric species)	http://www.marlin.ac.uk
	Cerithiella metula	http://www-umea.slu.se/MiljoData/webrod/blotdjur.xls
	Oenopota turricula	http://www-umea.slu.se/MiljoData/webrod/blotdjur.xls
	Arctica islandica	Macdonald *et al.*, 1996
	Kelliella miliaris	http://www.um.u-tokyo.ac.jp/publish_db/Bulletin/no35/no35046.html
	Modiolus modiolus (plus one congeneric species)	Macdonald *et al.*, 1996
	Thracia convexa	http://www.gol.grosseto.it/puam/comgr/acquario/thracia_convexa.htm
	Tritonia hombergi	http://www.marlin.ac.uk
Pogonophora	*Siboglinum* sp.	http://www.ucmp.berkeley.edu/annelida/pogonophora.html
Priapulida	*Priapulus caudatus*	http://www.ldeo.columbia.edu/dees/ees/life/slides/phyla/priapulida.html

Table 6.13.2.3. Provisional list of habitat-forming benthic species. Species were drawn from the 1986 North Sea Benthos Survey list of macrobenthos (ICES, 1997) with the addition of common species known to occur in the area but not identified in that survey (indicated by *).

Phylum	Species	References
Annelida	*Ampharete falcata* (plus three congeneric species)	http://www.jncc.gov.uk/mermaid/biotopes/459.htm
	Amphicteis gunneri	Fauchald and Jumars, 1979
	Amythasides macroglossus	Fauchald and Jumars, 1979
	Eclysippe vanelli	http://www.biodiversity.uno.edu/~worms/translation/j-samythella.html (genus)
	Melinna cristata	http://www.jncc.gov.uk/mermaid/biotopes/459.htm
	Chaetopterus variopedatus	http://www2.bishopmuseum.org/HBS/invertguide/species/chaetopterus_sp.htm
	Myriochele sp.	Fauchald and Jumars, 1979
	Owenia fusiformis	http://www.marlin.ac.uk, http://www.mms.gov/intermar/SECTION6.PDF
	Sabellaria spinulosa (plus one other species)	Macdonald *et al.*, 1996
	Serpula vermicularis	http://www.marlin.ac.uk
	Lanice conchilega	Macdonald *et al.*, 1996
Arthropoda	*Ampelisca aequicornis* (plus seven congeneric species)	http://www.mms.gov/intermar/SECTION6.PDF (see *Ampelisca abdita*)
	Byblis gaimardi	http://www.marlin.ac.uk
	Haploops tubicola	http://www.marlin.ac.uk/Bio_pages/Bio_Eco_IMU.TubeAP.htm
	Aoridae indet.	http://www.crustacea.net/crustace/amphipoda/aoridae/www/gilesi.htm
Bryozoa	*Cellaria fistulosa*	http://www.marlin.ac.uk
	Crisia eburnea	http://www.ulstermuseum.org.uk/marinelife/bryozoa/criebu.htm
	Flustra foliacea	http://www.marlin.ac.uk
	Bowerbankia sp.	http://www.calacademy.org/research/izg/SFBay2K/Bowerbankia.htm
Chordata	*Molgula* sp.	http://www.marlin.ac.uk
	Dendrodoa grossularia	http://www.marlin.ac.uk
	Styela coriacea	http://www.marlin.ac.uk
Cnidaria (Hydrozoa)	*Ectopleura crocea**	Genzano, 1998
	*Tubularia indivisa**	Orlov, 1997; Macdonald *et al.*, 1996
	*Eudendrium rameum**	Orlov, 1997
	*Laomedea gelatinosa** (plus congeneric species)	Pulfrich, 1996
	*Halecium muricatum**	Orlov, 1997
	*Abietinaria abietina**	Orlov, 1997
	*Hydrallmania falcata**	Orlov, 1997
	*Diphasia fallax** (plus congeneric species)	Orlov, 1997
	*Sertularia cupressina** (plus congeneric species)	Pulfrich, 1996; Schmidt and Warner, 1991
	Nemertesia antennina *	Hughes, 1975
(Anthozoa)	*Caryophyllia smithii** (plus one other species)	http://www.marlin.ac.uk

Table 6.13.2.3. Continued.

Phylum	Species	References
(Anthozoa)	*Lophelia pertusa**	Husebø *et al.*, 2002
	*Paragorgia arborea**	Husebø *et al.*, 2002
	*Primnoa resedaeformis**	Husebø *et al.*, 2002
	*Paramuricea placomus**	Husebø *et al.*, 2002
	*Desmophyllum cristagalli**	Hall-Spencer *et al.*, 2002
	*Enallopsammia rostrata**	Hall-Spencer *et al.*, 2002
	*Madrepora oculata**	Hall-Spencer *et al.*, 2002
	*Solenosmilia variabilis**	Hall-Spencer *et al.*, 2002
Echiura	*Echiurus echiurus*	www.marlin.ac.uk/Bio_pages/Bio_Eco_CMS.AbrNucCor.htm
Mollusca	*Modiolus modiolus* (plus one congeneric species)	Macdonald *et al.*, 1996
	Musculus discors (plus one congeneric species)	http://www.marlin.ac.uk
	Mytilus edulis	http://www.marlin.ac.uk
	*Ostrea edulis**	http://www.marlin.ac.uk
	*Crassostrea angulata** (plus one congeneric species)	http://www.marlin.ac.uk
Phoronida	*Phoronis hippocrepia* (plus one congeneric species)	http://www.marlin.ac.uk
Porifera	*Halichondria oculata** (plus one congeneric species)	Macdonald *et al.*, 1996
	*Axinella dissimilis**	http://www.marlin.ac.uk
	*Ciocalypta penicillus**	http://www.marlin.ac.uk
	*Demacidon fruticosum**	http://www.marlin.ac.uk
	*Pachymatisma johnstonia**	http://www.marlin.ac.uk
	*Polymastia mamillaris** (plus one congeneric species)	http://www.marlin.ac.uk
	*Suberites carnosus** (plus two congeneric species)	http://www.marlin.ac.uk
	*Cliona celata**	http://www.itsligo.ie/biomar/porifera/clicel.htm
Pogonophora	*Siboglinum* sp.	http://www.ucmp.berkeley.edu/annelida/pogonophora.html

Table 6.13.2.4. Provisional list of opportunistic benthic species. Species were drawn from the 1986 North Sea Benthos Survey list of macrobenthos (ICES, 1997) with the addition of common species known to occur in the area but not identified in that survey (indicated by *).

Phylum	**Species**	**References**
Annelida	*Capitella capitata*	Pearson and Rosenberg, 1978
	Cirratulus cirratus	http://www.marlin.ac.uk
	Chaetozone setosa	http://www.mms.gov/intermar/SECTION6.PDF
	Microphthalmus listensis	Grassle and Grassle, 1974
	Nephtys ciliata	Pearson and Rosenberg, 1978
	Lagis koreni	http://www.marlin.ac.uk
	Pectinaria belgica	http://www.nioz.nl/en/deps/mee/projects/poster/magda.htm
	Pomatoceros triqueter	http://www.marlin.ac.uk
	Polydora ciliata (plus seven congeneric species)	Grassle and Grassle, 1974; Pearson and Rosenberg, 1978
	Scolelepis bonnieri (plus four congeneric species)	Pearson and Rosenberg, 1978
	Spio filicornis (plus two congeneric species)	Pearson and Rosenberg, 1978
	Spiophanes bombyx (plus one congeneric species)	Macdonald *et al.*, 1996
	Streblospio shrubsoli	Grassle and Grassle, 1974; Pearson and Rosenberg, 1978
	Syllides longocirrata	Grassle and Grassle, 1974
Arthropoda	*Ampelisca aequicornis* (plus seven congeneric species)	http://www.mms.gov/intermar/SECTION6.PDF (see *Ampelisca abdita*)
Cnidaria	*Bougainvillia principis**	Orlov, 1997
(Hydrozoa)	*Catablema abyssi**	Orlov, 1997
	*Eudendrium capillare**	Orlov, 1997
	*Orthopyxis integra**	Orlov, 1997
	*Opercularella nana**	Orlov, 1997
	*Tetrapoma quadridentatum**	Orlov, 1997
	*Lafoea pocillum**	Orlov, 1997
	*Filellum serpens**	Orlov, 1997
Echiura	*Echiurus echiurus*	http://www.marlin.ac.uk/Bio_pages/Bio_Eco_CMS.AbrNucCor.htm

Table 6.13.2.5. Provisional list of benthic scavenger species. Species were drawn from the 1986 North Sea Benthos Survey list of macrobenthos (ICES, 1997) with the addition of common species known to occur in the area but not identified in that survey (indicated by *).

Phylum	Species	References
Arthropoda	*Corystes cassivelaunus*	Macdonald *et al.*, 1996
	Hyas coarctatus	Rumohr and Kujawski, 2000
	Macropodia spp.*	Kaiser and Spencer, 1996
	*Pagurus bernhardus** (plus one congeneric species)	
	Liocarcinus depurator (plus two congeneric species)	Kaiser and Spencer, 1996
Echinodermata	*Amphiura chiajei* (plus one congeneric species)	http://www.marlin.ac.uk
	*Asterias rubens**	
	*Henricia sanguninolenta**	
	Ophiura affinis (plus four congeneric species*)	Rumohr and Kujawski, 2000
	*Ophiocomina nigra**	Rumohr and Kujawski, 2000
	Ophiothrix fragilis	Macdonald *et al.*, 1996
	Crossaster papposus	http://www.marlin.ac.uk
	Brissopsis lyrifera	Rumohr and Kujawski, 2000
Mollusca	*Buccinum undatum*	Rumohr and Kajawski, 2000
	Colus gracilis	http://www.marlin.ac.uk
	Neptunea antiqua	McConnaughey *et al.*, 2000

6.13.6 Conclusions

ICES remains convinced of the importance of healthy benthic communities as part of a well-managed North Sea ecosystem.

Our identification of opportunistic species is incomplete, both owing to the lack of a complete North Sea species list and our incomplete knowledge on the biology of the species. These issues could fruitfully be addressed at a workshop organized in cooperation with BEWG. This would give a more complete list of the species which should be considered in this EcoQO element.

The admittedly incomplete lists included here for elements (o) and (p) already extend to 250 taxa, with a further 92 unable to be assessed due to a lack of information on their biology. Given the scale of these lists, ICES continues to have reservations about the practical utility of the proposed EcoQ elements, both from the point of view of the effort required to monitor them effectively, and the ability to respond in a targeted way with management measures that have a high likelihood of achieving the objective in a reasonable time frame.

ICES is keen to further develop appropriate EcoQOs for benthic systems but believes that this should be done in two ways: firstly, through a focus on habitat quality, and secondly, through the development of EcoQOs targeted at specific issues, such as those already adopted as part of the pilot scheme.

6.13.7 References

Basford, D., Eleftheriou, A., and Raffaelli, D. 1990. The infauna and epifauna of the northern North Sea. Netherlands Journal of Sea Research, 25(1–2): 165–173.

Basford, D.J., Moore, D.C., and Eleftheriou, A. 1989. The epifauna of the Northern North Sea. Journal of the Marine Biological Association of the United Kingdom, 69(2): 387–407.

Basford, D.J., Moore, D.C., and Eleftheriou, A. 1996. Variations in benthos in the north-western North Sea in relation to the inflow of Atlantic Water, 1980–1984. ICES Journal of Marine Science, 53: 957–963.

Callaway, R., Alsvåg, J., de Boois, I., Cotter, J., Ford, A., Hinz, H., Jennings, S., Kröncke, I., Lancaster, J., Piet, G., Prince, P., and Ehrich, S. 2002. Diversity and community structure of epibenthic invertebrates and fish in the North Sea. ICES Journal of Marine Science, 59(6): 1199–1214.

Duineveld, G.C.A., Künitzer, A., Niermann, U., Wilde, P.A.W.J., and Gray, J.S. 1991. The macrobenthos of the North Sea. Netherlands Journal of Sea Research, 28: 53–65.

Dyer, M.F., Fry, W.G., Fry, P.D., and Cranmer, G.J. 1983. Benthic regions within the North Sea. Journal of the Marine Biological Association of the United Kingdom, 63: 683–693.

Fauchald, K., and Jumars, P.A. 1979. The diet of worms: A study of polychaete feeding guilds.

Oceanography and Marine Biology Annual Review, 17: 193–284.

Frid, C.L.J., Harwood, K.G., Hall, S.J., and Hall, J.A. 2000. Long-term changes in the benthic communities on North Sea fishing grounds. ICES Journal of Marine Science, 57: 1303–1309.

Genzano, G.N. 1998. Hydroid epizoites on hydroids *Tubularia crocea* and *Sertularella mediterranea* from the intertidal of Mar del Plata (Argentina). Russian Journal of Marine Biology, 24: 123–126.

Gili, J.-M., and Hughes, R.G. 1995. The ecology of marine benthic hydroids. Oceanography and Marine Biology Annual Review, 33: 251–426.

Gili, J.-M., Murillo, J., and Ros, J. 1989. The distribution patterns of benthic Cnidarians in the Western Mediterranean. Scientia Marina, 53: 19–35.

Gilkinson, K., Paulin, M., Hurley, S., and Schwinghamer, P. 1998. Impacts of trawl door scouring on infaunal bivalves: results of a physical trawl door model/dense sand interaction. Journal of Experimental Marine Biology and Ecology, 224: 291–312.

Glémarec, M. 1973. The benthic communities of the European North Atlantic Continental Shelf. Oceanography and Marine Biology Annual Review, 11: 263–289.

Goldson, A.J., Hughes, R.G., and Gliddon, C.J. 2001. Population genetic consequences of larval dispersal mode and hydrography: a case study with bryozoans. Marine Biology, 138: 1037–1042.

Grassle, J.F., and Grassle, J.P. 1974. Opportunistic life histories and genetic systems in marine benthic polychaetes. Journal of Marine Research, 32: 253–284.

Hagberg, J., Jonzén, N., Lundberg, P., and Ripa, J. 2003. Uncertain biotic and abiotic interactions in benthic communities. Oikos, 100(2): 353–361.

Hall-Spencer, J., Allain, V., and Fosså, J.H. 2002. Trawling damage to Northeast Atlantic ancient coral reefs. Proceedings of the Royal Society of London, 269: 507–511.

Heip, C., Basford, D., Craeymeersch, J., Dewarumez, J.-M., Dörjes, J., de Wilde, P., Duineveld, G., Eleftheriou, A., Herman, P.M.J., Kingston, P., Niermann, U., Künitzer, A., Rachor, E., Rumohr, H., Soetaert, K., and Soltwedel, T. 1992. Trends in biomass, density and diversity of North Sea macrofauna. ICES Journal of Marine Science, 49: 13–22.

Hughes, R.G. 1975. The distribution of epizoites on the hydroid *Nemertesia antennina* (L.). Journal of the Marine Biological Association of the United Kingdom, 55: 275–294.

Husebø, Å., Nøttestad, L., Fosså, J.H., Furevik, D.M., and Jørgensen, S.B. 2002. Distribution and abundance of fish in deep-sea coral habitats. Hydrobiologia, 471: 91–99.

ICES. 1994. Report of the Benthos Ecology Working Group. ICES CM 1994/L:4, pp. 92–94.

ICES. 1997. Atlas of North Sea Benthic Infauna. ICES Cooperative Research Report, 218.

ICES. 2001. Report of the ICES Advisory Committee on Ecosystems, 2001. ICES Cooperative Research Report, 249: 58–59.

ICES. 2002. Report of the ICES Advisory Committee on Ecosystems, 2002. ICES Cooperative Research Report, 254: 75–76.

Kaiser, M.J., and Spencer, B.E. 1994. Fish scavenging behaviour in recently trawled areas. Marine Ecology Progress Series, 112: 41–49.

Kaiser, M.J., and Spencer, B.E. 1996. Behavioural responses of scavengers to beam trawl disturbance. *In* Aquatic predators and their prey, pp. 116–123. Ed. by S.P.R. Greenstreet and M.L. Tasker. Blackwell Scientific Publications, Oxford.

Kaiser, M.J., Ramsay, K., Richardson, C.A., Spence, F.E., and Brand, A.R. 2000. Chronic fishing disturbance has changed shelf sea benthic community structure. Journal of Animal Ecology, 69: 494–503.

Künitzer, A., Basford, D., Craeymeersch, J.A., Dewarumez, J.M., Dorjes, J., Duineveld, G.C.A., Eleftheriou, A., Heip, C., Herman, P., Kingston, P., Niermann, U., Rachor, E., Rumohr, H., and Dewilde, P.A.J. 1992. The benthic infauna of the North-Sea—Species distribution and assemblages. ICES Journal of Marine Science, 49: 127–143.

Macdonald, D.S., Little, M., Eno, C., and Hiscock, K. 1996. Disturbance of benthic species by fishing activities: a sensitivity index. Aquatic Conservation: Marine and Freshwater Ecosystems, 6: 257–268.

Makra, A., and Keegan, B.F. 1999. Arm regeneration in *Acrocnida brachiata* (Ophiuroidea) at Little Killary, west coast of Ireland. Proceedings of the Royal Irish Academy, 99B: 95–102.

McConnaughey, R.A., Mier, K.L., and Dew, C.B. 2000. An examination of chronic trawling effects on soft bottom benthos of the eastern Bering Sea. ICES Journal of Marine Science, 57: 1377–1388.

Migotto, A.E., Marques, A.C., and Flynn, M.N. 2001. Seasonal recruitment of hydroids (Cndiaria) on experimental panels in the São Sebastião Channel, Southeastern Brazil. Bulletin of Marine Science, 68: 287–298.

NMMP. 2001. National Marine Monitoring Programme. Green Book. http://www.marlab.ac.uk/greenbook/Main%20text.pdf.

Orlov, D. 1997. Epizoic associations among the White Sea hydroids. Scientia Marina, 61: 17–26.

Pearson, T.H., and Rosenberg, R. 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanography and Marine Biology Annual Review, 16: 229–311.

Pearson, T.H., and Rosenberg, R. 1987. Feast and famine: structuring factors in marine benthic communities. *In* Organization of communities: past and present, pp. 373–395. Ed. by J.H.R. Gee and P.S. Giller. British Ecological Society and Blackwell Scientific Publications, Oxford.

Porter, J.S., Ellis, J.R., Hayward, P.J., Rogers, S.I., and Callaway, R. 2002. Geographic variation in the abundance and morphology of the bryozoan *Alcyonidium diaphanum* (Ctenostomata: Alcyonidiidae) in UK coastal waters. Journal of the Marine Biological Association of the United Kingdom, 82: 529–535.

Pulfrich, A. 1996. Attachment and settlement of post-larval mussels (*Mytilus edulis* L.) in the Schleswig-Holstein Wadden Sea. Journal of Sea Research, 36: 239–250.

Rees, H.L., Pendle, M.A., Waldock, R., Limpenny, D.S., and Boyd, S.E. 1999. A comparison of benthic biodiversity in the North Sea, English Channel and Celtic Seas. ICES Journal of Marine Science, 56: 228–246.

Ricciardi, A., and Bourget, E. 1999. Global patterns of macroinvertebrate biomass in marine intertidal communities. Marine Ecology Progress Series, 185: 21–35.

Rumohr, H., and Kujawski, T. 2000. The impact of trawl fishery on the epifauna of the southern North Sea. ICES Journal of Marine Science, 57: 1389–1394.

Schmidt, G.H., and Warner, G.F. 1991. The settlement and growth of *Sertularia cupressina* (Hydrozoa, Sertulariidae) in Langstone Harbour, Hampshire, UK. Hydrobiologia, 216/217: 215–219.

Snelgrove, P.V.R., and Butman, C.A. 1994. Animal-sediment relationships revisited: cause vs. effect. Oceanography and Marine Biology Annual Review, 32: 111–177.

Solan, M., Germano, J.D., Rhoads, D.C., Smith, C., Michaud, E., Parry, D., Wenzhofer, F., Kennedy, B., Henriques, C., Battle, E., Carey, D., Iocco, L., Valente, R., Watson, J., and Rosenberg, R. 2003. Towards a greater understanding of pattern, scale and process in marine benthic systems: a picture is worth a thousand worms. Journal of Experimental Marine Biology and Ecology, 285/286: 313–338.

Stepanjants, S.D. 1998. Obelia (Cnidaria, Medusozoa, Hydrozoa): phenomenon, aspects of investigations, perspectives for utilization. Oceanography and Marine Biology Annual Review, 36: 179–215.

Tunberg, B., and Nelson, W.G. 1998. Do climatic oscillations influence cyclical patterns of soft bottom microbenthic communities on the Swedish west coast? Marine Ecology Progress Series, 170: 85–94.

Underwood, A.J. 1997. Experiments in ecology: their logical design and interpretation using analysis of variance. Cambridge University Press.

7 PROGRESS IN MARINE HABITAT CLASSIFICATION AND MAPPING IN THE ICES AREA, INCLUDING THE BALTIC SEA

Request

Item 5 of the 2003 requests from the Helsinki Commission, which states as follows:

Following the request by the Second Meeting of the Nature Conservation and Coastal Zone Management Group (HELCOM HABITAT 2/2001) with reference to the final Minutes of the Meeting, the Habitat Group requested ICES to include the Baltic Sea in its work on a marine classification and mapping system generally accepted and covering the whole Baltic Sea area. The EUNIS classification system should be taken into consideration as well as other ongoing projects in the region such as the CHARM project on "Characterization of the Baltic Sea Ecosystem Dynamics and function of coastal types", which is connected with the EU Water Framework Directive.

Source of information

The 2003 Report of the Working Group on Marine Habitat Mapping (WGMHM) (ICES CM 2003/E:08).

Summary

Considerable progress is being made on the development of habitat classification schemes and the mapping of marine habitats in the ICES area. To date, efforts have focused mainly on benthic habitats. The EUNIS classification scheme has proved useful in some applications, but lacking in others. A number of initiatives are under way to investigate the appropriate classification for the water column and the coupling of the benthic and pelagic classifications. Investigators are also turning their attention to methods for dealing with temporal variation. The plethora of activity demonstrates the wide range of interest in habitat classification and mapping and the potential applications. This interest is being reinforced by several successful practical applications of habitat classification and mapping.

Continued progress on the development of classification schemes and habitat mapping will best be achieved through pilot projects such as the proposed OSPAR project for the North Sea.

The development of a habitat classification scheme for the Baltic Sea is complicated by the somewhat unique environmental conditions in the Baltic which strongly influence the biology. In the Baltic, benthic communities are more strongly correlated with salinity than with substrate and exposure. This is a highly dynamic system that makes the application of classification schemes like EUNIS difficult. A dedicated project for the Baltic will be needed in order to make any progress on habitat classification and mapping for the area.

Recommendations and advice

ICES endorses the OSPAR mapping initiative as a very clear mechanism for implementing a mapping programme at an international scale using a logical step-wise approach that is scientifically justified. ICES is prepared to actively contribute to the OSPAR mapping initiatives through:

- provision of expertise and advice on mapping techniques and the development of guidelines, and
- practical development of a North Sea map.

ICES recommends the continued development of classification systems including continued development of the EUNIS system, the development of schemes for the pelagic habitat, the coupling of pelagic and benthic systems, and the consideration of transitional (continuous) techniques as used by landscape ecologists.

ICES notes that the Baltic Sea is a dynamic environment that presents additional challenges over other areas being considered in developing a satisfactory classification scheme. Further development of a classification scheme for the Baltic would best be achieved through a dedicated project.

ICES endorses data management initiatives to archive metadata held by a number of agencies. The importance of good management and archiving of data, particularly the management of metadata, is stressed as it is critical to the development of habitat maps and associated information.

Scientific background

7.1 Introduction

There are numerous initiatives on marine habitat classification and mapping in many ICES Member Countries. The focus and purpose of these initiatives are varied, but many of the difficulties being encountered and the techniques that are being evaluated are common to all. The ICES Working Group on Marine Habitat Mapping (WGMHM) has provided a forum for the discussion of these issues among the ICES Member Countries and has undertaken work to assist in the resolution of difficulties and the development of compatible approaches to classification and mapping. Methods and technology for marine habitat classification and mapping have evolved substantially over the past few years (e.g., ICES, 2001, 2002) and are continuing to evolve. This section discusses recent developments in the ICES area in general and specifically in the Baltic Sea, and provides recommendations on future directions.

7.2 Progress in marine habitat classification

ACE (ICES, 2001) has previously reviewed several habitat classification systems and selected EUNIS (European Nature Information System) as appearing to be most appropriate for wide application. It was also recognized that considerable further testing and development of EUNIS was required. Initiatives to apply the EUNIS classification have met with varied success. An attempt to apply EUNIS to seabed maps of the Basque coast met with limited success due to a lack of data and difficulties in interpreting qualitative categories in EUNIS (e.g., moderate versus highly exposed). Similarly, Dutch and Spanish initiatives concluded that the EUNIS system required significant development before it could usefully be applied to pelagic habitats at a regional level. The classification of habitats in epipelagic waters of the Bay of Biscay has been approached using spatio-temporal studies of phytoplankton, zooplankton, small pelagic fish, and their predators. Preliminary results, based exclusively on phytoplankton studies, indicate a potential classification based on mean chlorophyll *a* values derived from SEAWIFS data. Both seasonal and annual variability characterizes these data, so any classification system must incorporate some temporal notation. Riverine delivery of freshwater to the epipelagic has a significant influence on chlorophyll *a* distribution and is itself temporally variable. Such influences would best be dealt with by applying threshold values to delineate habitat categories (e.g., salinity >35). EUNIS currently makes use of gradients in defining pelagic habitats, but these need further categorization in terms of their periodicity (ephemeral, seasonal, permanent) in order to define distinct (exclusive) classification criteria. Finally, maps of pelagic habitats will require considerable amounts of data (in all four dimensions) and reliable numerical models may provide a practical alternative to real data.

The marine benthic classification for Britain and Ireland (originally published as the BioMar classification in 1997) has been completely revised. This classification was the culmination of the Marine Nature Conservation Review programme of the UK Joint Nature Conservation Committee (JNCC) tasked with describing the variety (rather than the extent) of coastal (intertidal and nearshore subtidal) marine habitats. The original BioMar classification had been adopted to populate levels 4 and 5 of the EUNIS system. It has been revised over the past three years in the light of greater understanding of what is required to classify northern European seabed habitats and the availability of new data. The classification is now drawn from multivariate analysis of some 30,000 benthic samples from the coasts of Britain and Ireland, incorporating both species and physical data, and can populate the EUNIS classification as far as level 6. The revised classification places a greater emphasis on energy status for rocky habitats (e.g., high, moderate, or low energy environments) and follows the EUNIS divisions at level 3 for sediment habitats. The revised classification is accessible on the Internet at www.jncc.gov.uk/MarineHabitatClassification. Being primarily a coastal classification, its limitations are recognized and there is a need for complementary data to address the classification of habitats specific to offshore waters, particularly from surveys that target both benthic infauna and epifauna. The analysis of large data sets has led to a detailed and more robust classification using a bottom-up approach. This provides greater confidence in the higher level units, which are more physically defined, and allows better predictive mapping where information on only the physical character is available. Consistency in the classification of the lower level units (levels 4 and 5) is important for mapping purposes because these units can be aggregated in various ways to suit different purposes.

The EUNIS classification system pays little attention to temporal changes in habitats and that such consideration is important in any mapping context. Recent developments have done much to address the initial inadequacies of the classification. Increased use and understanding of the classification have enabled it to develop from a descriptive tool to one that increasingly accounts for ecosystem function. Further development is still required but should not unduly divert ICES away from efforts to encourage, promote, and support active mapping initiatives. The best opportunity for further evolution of the classification would appear to be from experience gained when applying it.

The strict use of the EUNIS hierarchy or any other classification system will not always be appropriate. However, mapping in modern GIS and proper data management permit easy aggregation of finer units into broader or more specialized units (e.g., into broader EUNIS types, for Habitats Directive types or for fish habitat).

7.3 Progress in marine habitat mapping in the ICES area

7.3.1 Marine habitat mapping projects in the ICES area

The annual reports on habitat mapping from the ICES Member Countries include a tabulation of more than 150 projects. The size, purpose, and objectives of these projects vary considerably. There are some examples of the successful application of habitat mapping. For example, multi-beam surveys, accompanied with ground-truth data, have been used to map benthic habitats on the Canadian sector of Georges Bank. The scallop fishing industry, which partly funded this work, has benefited through a reduction in the effort required to land their quota. The environment has benefited through a marked reduction in habitat disruption.

In the UK, a number of habitat mapping techniques are being used to monitor the scale and spatial extent of any impact and also to assess the longevity of physical and biological disturbance from aggregate extraction at relinquished sites. There appears to be a link between an impoverished faunal community and the last period of high intensity dredging carried out at a site. There also appears to be a higher level of variability in the particulate nature of substrata within the area of high dredging intensity. Techniques such as single-line bathymetry, digital video and stills photography, and acoustic ground-discrimination systems (AGDS) have helped to further define the nature of the impact both spatially and temporally. Preliminary findings suggest that assumptions of relatively speedy physical and biological recovery at gravely sites may need to be revised, since in some situations physical and biological effects can still be detected some nine years after the cessation of dredging.

There are a number of important points that can be gleaned from an analysis of these habitat mapping projects:

1) There is a wide variety of scientific and managerial reasons for benthic habitat mapping which lead to strong support of habitat mapping by a wide spectrum of stakeholders.
2) The appropriate scale of habitat mapping is variable.
3) An integrated approach to habitat mapping works, no matter what scale of mapping is needed.
4) Remotely sensed geophysical information, followed by ground-truthing techniques, is a robust and successful survey philosophy.
5) The collection of visual ground-truthing data (from video and still photography) from the sea floor is considered important as it provides large amounts of accurate and easily accessible information on the nature of the seabed (physical and biological) compared to traditional remote sampling techniques (e.g., grabs).
6) The ability to build a visual database (i.e., a geo-referenced, GIS-based display), and to zoom in or out to different levels of data and of detail, is key to scientific advancement of marine habitat mapping.
7) The GIS approach to habitat mapping provides (almost) limitless capability to study factors involved in defining habitats, including physical and chemical oceanographic data.
8) Boundaries developed in a habitat map must be ecologically significant; biology should not blindly follow isolines.

This analysis also identifies a number of significant challenges for the further development and application of marine habitat mapping, not the least of which will be the availability of essential data. There are also some technological challenges to collecting acoustic data and ground-truthing information in shallow waters.

7.3.2 OSPAR marine habitat mapping initiative

OSPAR convened a workshop at Stansted, UK in October 2002 to assess the feasibility of establishing habitat maps for the OSPAR area and to recommend a strategy to carry out any such mapping projects. A document was produced that outlined a mapping strategy for the OSPAR area which was presented to the OSPAR Biodiversity Committee (BDC) in Dublin in January 2003. The strategy was received with enthusiasm by the BDC and the UK was asked to initiate the proposals. The strategy, in the short term, is to produce two types of habitat maps:

1) a point distribution (geo-referenced) map of specific habitats of particular conservation interest, e.g., *Lophelia* coral reefs, for the entire OSPAR area; and
2) a holistic habitat map of the North Sea, to include a compilation of all data relevant to the North Sea area, the goal of which is to identify the spatial extent of various habitats.

The North Sea was selected on the basis that considerable research has been carried out there and that coverage was substantial. This strategy was preferred over the development of a habitat map of the entire OSPAR area in that it could be presented as a pilot-type project that could be used to better define the requirements for mapping a wider part of the OSPAR area.

ICES considers this to be a very clear mechanism for implementing a mapping programme at an international scale. Also, it is considered a logical step-wise approach that is scientifically justified. ICES can contribute to the OSPAR mapping initiatives in two main ways:

1) provision of expertise and advice on mapping techniques and the development of guidelines;
2) practical development of a North Sea map.

A previous attempt to generate a map of the southern North Sea (the HabiMap project) encountered a number of problems, not the least of which was that data from individual sources were not in compatible formats. The result was that regional maps did not correspond seamlessly when amalgamated. That experience would be of value to this new initiative under OSPAR. It was felt that the previous exercise suffered from a lack of resources and that any new mapping activity should have a transboundary component to it. It is recognized that the on-going ICES North Sea Benthos Project would benefit from the production of a habitat map of the North Sea.

7.4 Progress in marine habitat mapping specific to the Baltic Sea

A meeting specifically to address the development of a classification scheme for the Baltic Sea was organized by the European Environment Agency (EEA) in October 2002. This meeting addressed the needs for a Baltic classification and discussed various approaches to classification of the Baltic Sea ecosystem in the context of the EUNIS system.

There have been a number of previous attempts to produce classifications of the Baltic Sea habitats. Overall, there is a good correlation between the benthic communities in the Baltic and its salinity regime. Therefore, a classification scheme should place a strong emphasis on the salinity regime. In the Baltic, salinity is a stronger driving force than substratum and exposure, which are used for the Atlantic classification in EUNIS.

The Baltic Sea forms an environmental gradient with changing borders. The Baltic system is dynamic and so are the borders at the sea floor. Most habitats and communities, especially those below the pycnocline, are in a permanent state of succession after disturbance and the whole system is in a state of contemporaneous instability. As a consequence, any attempt at a rigid classification (e.g., based on limited surveys of benthic communities) is not appropriate due to the natural dynamics and variability. Temporal changes in habitat and the associated biota are a dominant force in this system. Sediments are the most conservative classification factor in the Baltic Sea.

Further development of a classification scheme for the Baltic will best be achieved through a dedicated project and ICES is well-positioned to contribute advice to any such project.

7.5 References

ICES. 2001. Report of the ICES Advisory Committee on Ecosystems, 2001. ICES Cooperative Research Report, 249: 60–66.

ICES. 2002. Report of the ICES Advisory Committee on Ecosystems, 2002. ICES Cooperative Research Report, 254: 39–41.

8 DISTRIBUTION OF COLD-WATER CORALS IN THE NORTH ATLANTIC AND THE RELATION TO FISHERIES IN THE NORTHEAST ATLANTIC

Request

The European Commission, Directorate General for Fisheries, request (letter of 5 July 2000) for the "*identification of areas where cold-water corals may be affected by fishing*".

Source of information

The 2003 Report of the Study Group on Mapping the Occurrence of Cold-Water Corals (SGCOR) (ICES CM 2003/ACE:02).

Summary

The focus of this section is on cold-water coral species, particularly *Lophelia pertusa*, that contribute to reef formation in waters colder than about 20 °C. Several publications appeared during 2002 and 2003 that have advanced our knowledge of the occurrence of cold-water corals in the Northeast Atlantic. Newly identified locations of *Lophelia* reefs have been reported by the UK, Ireland, Iceland, Norway, and Sweden. Static gear fisheries cause much less damage to these reefs than do mobile fishing gears. The use of mobile fishing gears is widespread in areas with *Lophelia*. It is noted that an Irish orange roughy trawl fishery utilizing sophisticated acoustic instruments has expanded rapidly since 2001 and poses a considerable threat to corals in the trawled areas characterized by seamounts or pinnacles. Several ICES Member Countries, including Ireland, the UK, Norway, and Canada, have recently implemented or suggested restrictions on bottom trawling in areas containing cold-water corals.

Recommendations and advice

ICES recommends as follows:

1) In order to assess the impact, and potential impact, of fishing on the destruction of corals other than *Lophelia pertusa* and other colony-forming Scleractinia corals, ICES Member Countries should provide further information on the distribution of, and threats to, large slow-growing octocorals, especially *Paragorgia arborea* and *Primnoa resedaeformis,* as well as other large forms of cold-water corals such as gorgonians.
2) In order to best tailor advice addressing actual fishing pressure, ICES Member Countries and relevant Commissions should provide access to detailed, suitably depersonalized data on the location of fishing effort, both current and anticipated, in areas known or likely to contain *Lophelia* and other cold-water corals.
3) In order to add to knowledge on the distribution of *Lophelia* and trawling impact, ICES Member Countries and relevant Commissions should ensure that by-catch recording schemes and fisheries observer reports include records of *Lophelia* and other cold-water corals.
4) In order to protect *Lophelia* reefs at the Darwin Mounds, ICES recommends that the area be closed to bottom trawling within a box with the following corner coordinates: 59° 54' N, 7° 39' W; 59° 54' N, 6° 47' W; 59° 37' N, 6° 47' W; and 59° 37' N, 7° 39' W.

Scientific background

8.1 Introduction

There is strong evidence of recent, permanent damage to cold-water coral features in the ICES area. Scientific advice is required to inform management of possible measures to avoid further damage. In this report, it is presumed that cold-water corals refer to those coral species that contribute to reef formation in waters colder than about 20 °C. In Northeast Atlantic waters, these include the azooxanthellate scleractinarian corals *Desmophyllum cristagalli*, *Enallopsammia rostrata*, *Lophelia pertusa*, *Madrepora oculata*, and *Solenosmilia variabilis*. The main reef-building species is *Lophelia pertusa*.

8.2 New information on the occurrence of cold-water corals in the North Atlantic

Several publications appeared during 2002 and 2003 that have furthered the knowledge of the occurrence of cold-water corals in the Northeast Atlantic from that reported in the 2002 ACE report (ICES, 2002).

A database of the global occurrence of *Lophelia pertusa* is being compiled (A. Freiwald, pers. comm.). Figure 8.2.1 indicates occurrences in the Northeast Atlantic.

8.2.1 Norway

Fosså and Alvsvåg (2003) have provided an updated map of the occurrence of *Lophelia pertusa* and reefs within Norwegian waters (Figure 8.2.1.1).

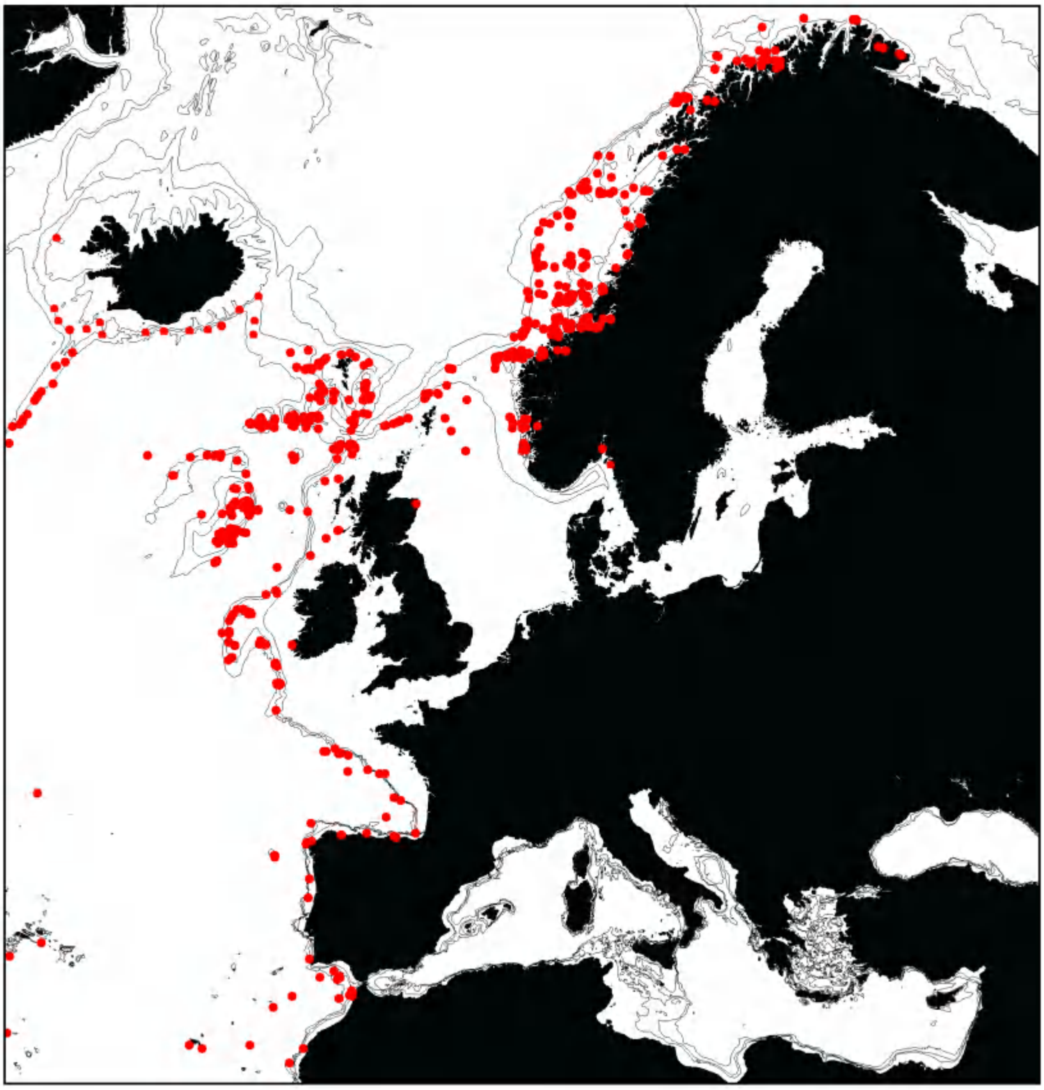


Figure 8.2.1. Distribution of hard cold-water corals tentatively identified as *Lophelia pertusa* in the Northeast Atlantic (A. Freiwald, pers. comm.).

In May 2002, a huge (initially measured at 35 km long and 3 km wide) *Lophelia* reef system (Figure 8.2.1.2) was discovered mainly between 300 m and 400 m depth on a steep and rugged part of the continental break off the archipelago of Lofoten, between 67°N and 68°N (Fosså and Alvsvåg, 2003).

A coral reef at least 1.2 km long and 200 m wide was found to the north of Tisler in Yttre Hvaler in Norway, close to the border to Sweden, in 2002 (Lundälv and Jonsson, 2003) (Figure 8.2.1.3). Living corals were found between 74 m and 160 m depth and yellow varieties of *Lophelia pertusa* were documented for the first time. There are at least two more, yet unexplored, reefs nearby. The area of habitat containing the reef extends across the Swedish/Norwegian boundary (Nilsson, 1997; Lundälv and Jonsson, 2000; Jonsson and Lundälv, 2001). The eastern parts of the area (Koster-Väderöfjorden and Singlefjorden) are situated in Swedish territorial waters and the western part (Yttre Hvaler) is situated in Norwegian territorial waters. The central position of Kosterfjorden/Yttre Hvaler is approximately 58°58.7' N and 11°01.6' E.

8.2.2 Sweden

Small amounts of reef occur in the Swedish Skagerrak adjacent to the Hvaler site described for Norway in Section 8.2.1 (J.H. Fosså, pers. comm.).

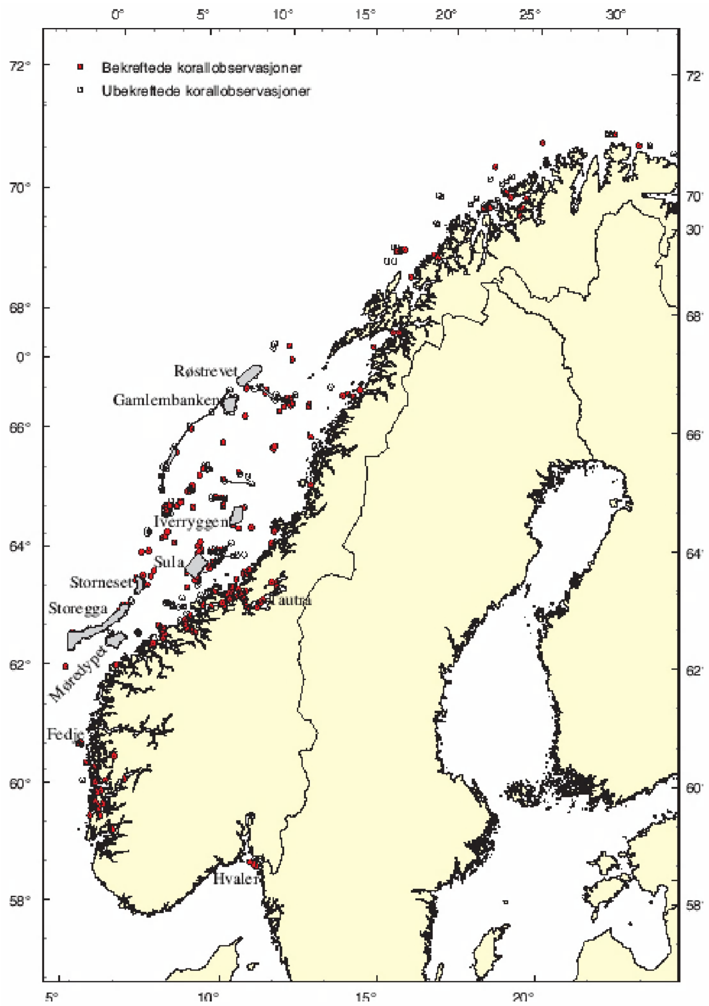


Figure 8.2.1.1. Reported occurrence of *Lophelia pertusa* in Norwegian waters at the end of December 2002. Named areas are described in detail in Fosså and Alvsvåg (2003).

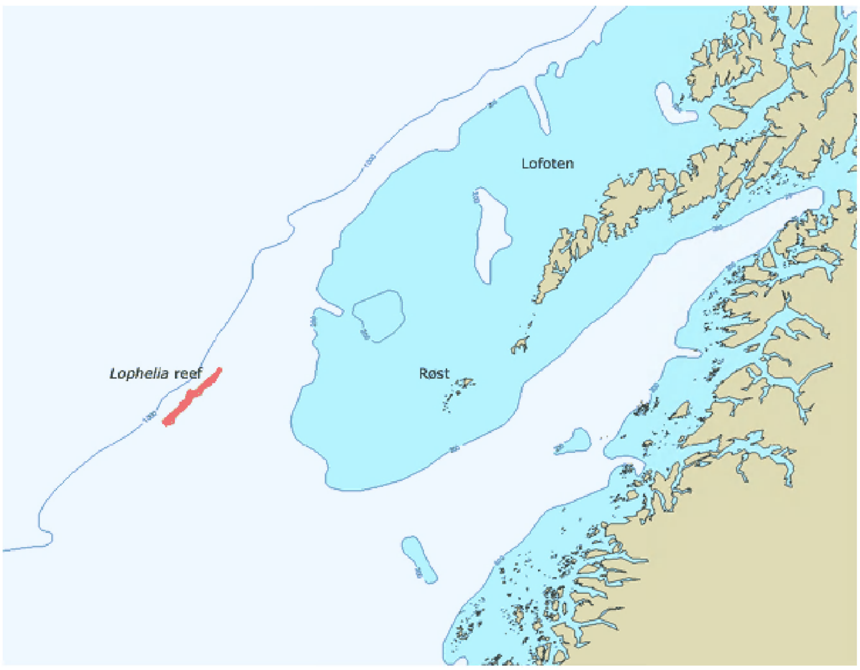


Figure 8.2.1.2. Location of the *Lophelia* reef discovered to the west of Røst, Norway in May 2002.

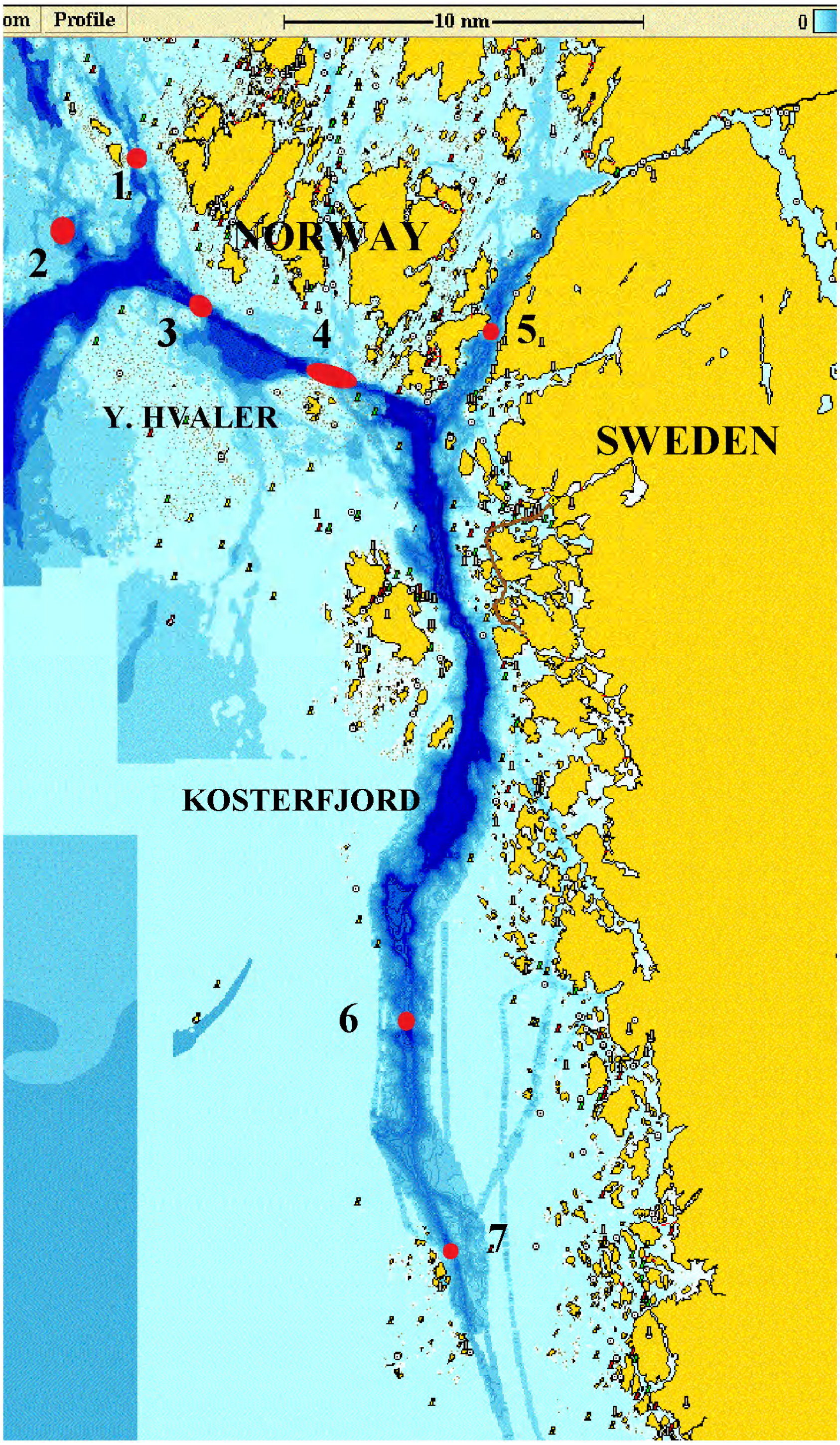


Figure 8.2.1.3. Map of known coral occurrences in the Yttre Hvaler/Kosterfjord area. Site 1 was studied with dredging techniques about thirty years ago and its present condition is unknown. Sites 2, 3, and 4 were discovered in 2002, and there are large living reefs in site 2 (Fjellknausene) and site 4 (Tisler), while site 3 (Djupekrakk) seems to have been destroyed by trawling. Site 5 (Sacken) is a small reef that has been known since the 1920s, but is badly damaged by trawling. Sites 6 and 7 contained living corals 15–20 years ago, but have since been destroyed by trawling. Small trawl exclusion areas have been created around sites 5, 6, and 7; notably, further damage occurred at site 5 after protective measures were implemented. Norwegian authorities are presently considering protective measures for sites 2 and 4.

8.2.3 Iceland

No coral surveys were conducted off Iceland in 2002/2003, but further reports have been gathered from retired fishermen on the location of coral reef areas around Iceland in the 1970s (Figure 8.2.3.1). The majority of these areas were to the south of Iceland, with some to the north and west.

8.2.4 United Kingdom

Information on the broad distribution of coral reefs on the Rockall Bank (Figure 8.2.4.1) was compiled from interviews with fishermen by J. Hall-Spencer (pers. comm.). This information broadly agrees with that presented in Wilson (1979a) and the same author's studies reported in Wilson (1979b). There has been no recent survey of this area.

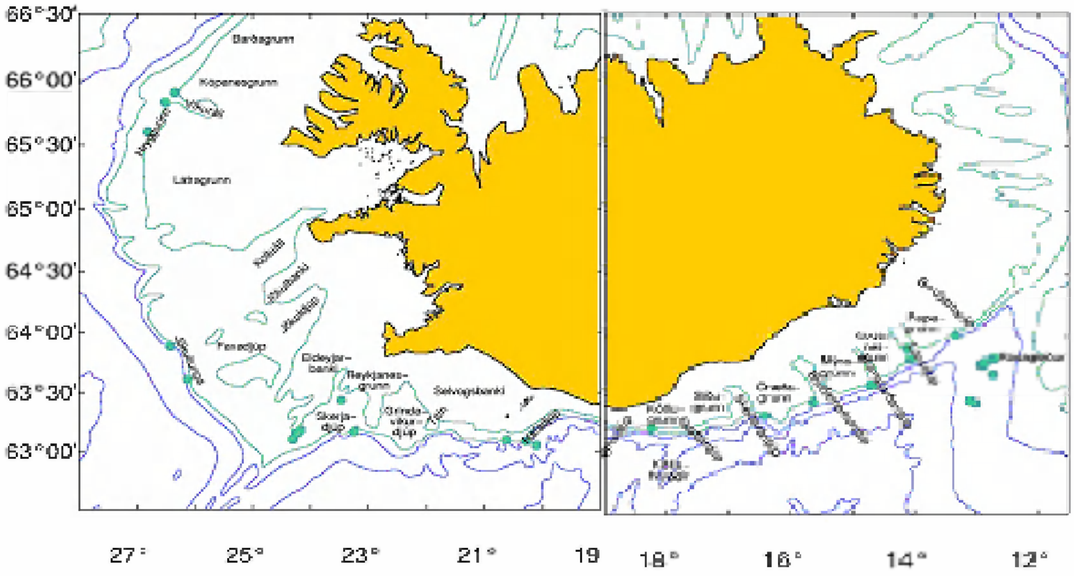


Figure 8.2.3.1. This map (based on information from retired fishermen) indicates where coral grounds (not all of which were confirmed to be *Lophelia*) occurred when trawling expanded into the shelf break area around Iceland in the 1970s (S.A. Steingrímsson, pers. comm.).

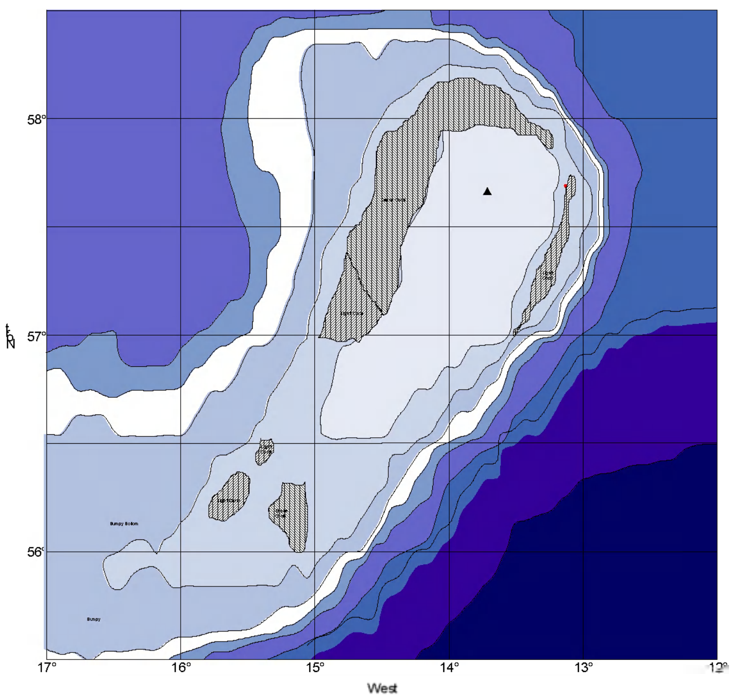


Figure 8.2.4.1. The distribution of coral reefs on Rockall Bank from fishermen's records (J. Hall-Spencer, pers. comm.). The cross-hatched areas indicate the presence of *Lophelia* reefs.

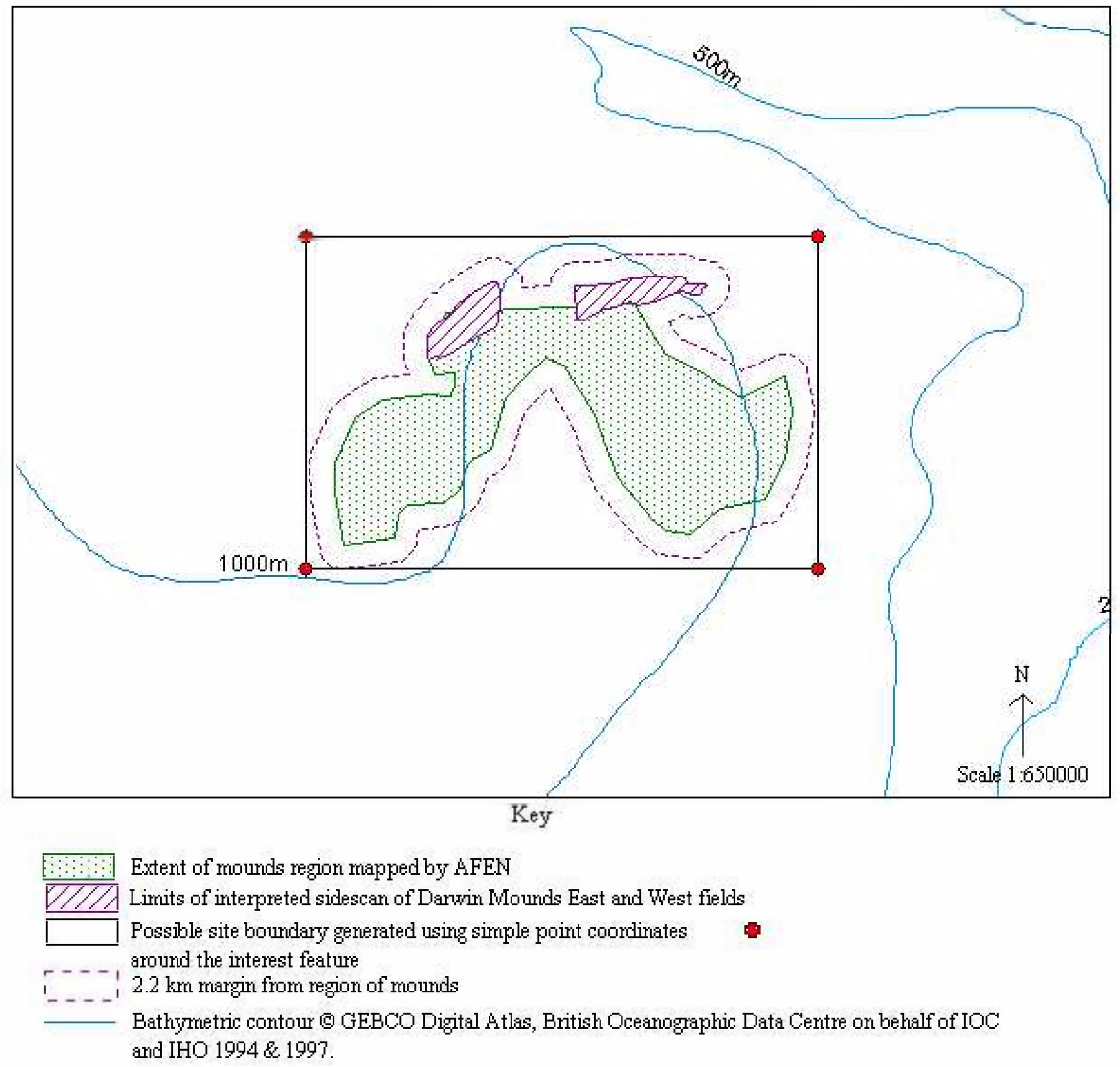


Figure 8.2.4.2. Extent of the mound features (speckled area) in the Darwin Mounds, along with boundary points around the mounds. The corner points of the boundary are 59° 54' N, 7° 39' W; 59° 54' N, 6° 47' W; 59° 37' N, 6° 47' W; and 59° 37' N, 7° 39' W.

The best known site in UK waters remains the Darwin Mounds to the north and west of Scotland. These are located in 1,000 m deep water to the south of the Wyville Thomson Ridge. The best known of these mounds were described in ICES (2002). Further analysis (Masson *et al*., 2003) has shown mound features to extend some way beyond the two best-known fields described in ICES (2002) (Figure 8.2.4.2). This area has been heavily impacted by trawl gear, with many areas typified by broken coral rubble and a reduction in biodiversity (Wheeler *et al*., 2003).

Roberts *et al*. (2003) described the distribution of *Lophelia* to the west of Scotland and the environmental controls on its occurrence.

8.2.5 Ireland

Deep-water coral reefs have been found in Irish waters associated with carbonate mounds (Hovland *et al*., 1994; Kenyon *et al*., 1998, 2003; Henriet *et al*., 1998; De Mol *et al*., 2002; Akhmetzhanov *et al*., 2003; van Weering *et al*., 2003) located to the west of Ireland. There are a number of mound clusters fringing the upper continental slope of the Rockall Trough and Porcupine Seabight (Croker and O'Loughlin, 1998). In the Porcupine Seabight, two major complex mound provinces have been identified: the Hovland/Magellan Mound Province on the northern slope of Porcupine Seabight and the Belgica Mound Province on the eastern slope of Porcupine Seabight (Figure 8.2.5.1) (De Mol *et al*., 2002). In the Rockall Trough, two major mound clusters on the southeastern and southwestern margin have been studied: the Pelagia Mound Province on the southeast Rockall Trough and the Logachev Mound Province on the southwest Rockall Trough (Kenyon *et al*., 2003). These mounds occur in water depths between 500 m and 1200 m and vary from small structures of a few metres to over 300 m in height, and occur singly or in large clusters (Kenyon *et al*., 1998; De Mol *et al*., 2002).

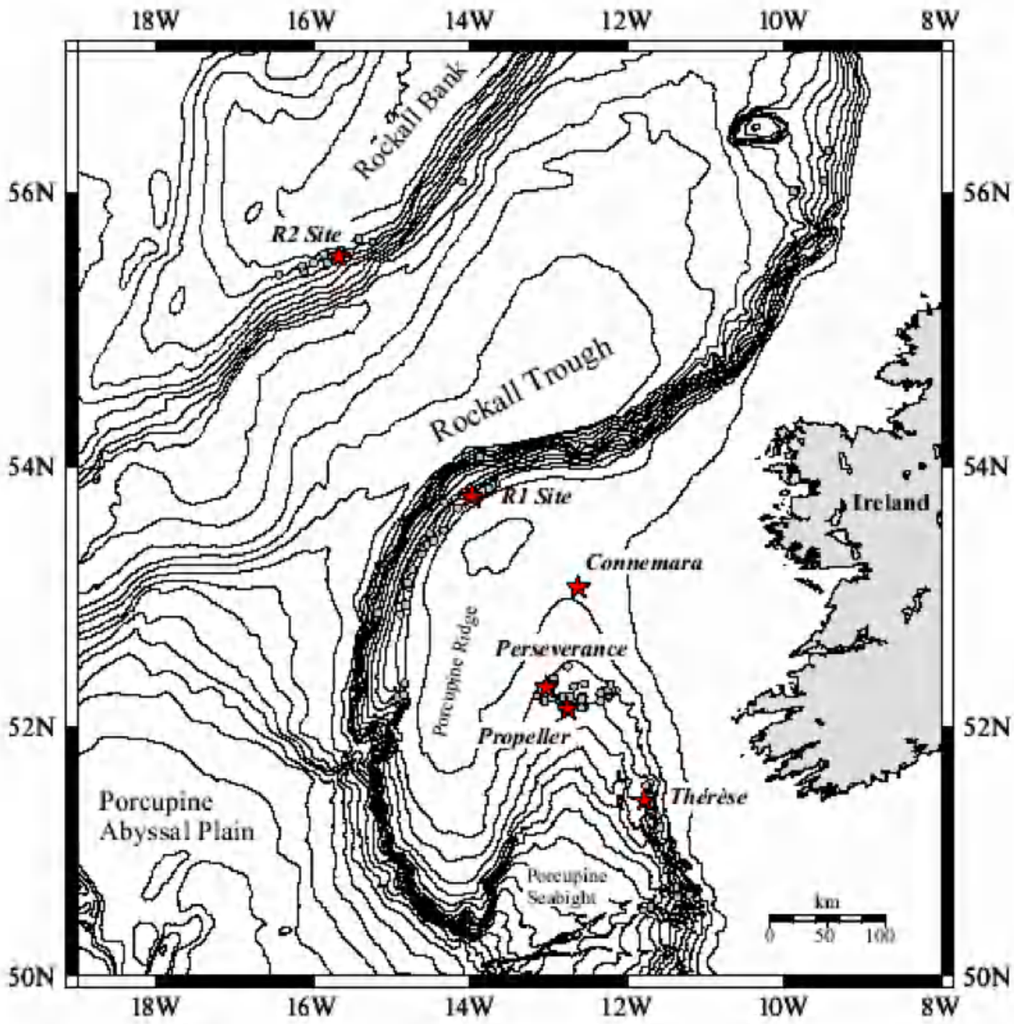


Figure 8.2.5.1. The location of carbonate mound sites investigated during the CARACOLE 2001 cruise (red stars). Circles highlight carbonate mound locations identified on the basis of seismic data (Croker and O'Loughlin, 1998). Bathymetric contour interval: 500 m (Grehan *et al.*, 2003).

During a French-Irish-EU research cruise CARACOLE (Carbonate Mound and Cold Coral Research), five coral/mound locations (Figure 8.2.5.1) were studied in detail: Thérèse Mound in the Belgica Mound Province; Propellor and Perseverance Mounds in the Hovland/Magellan Mound Province; the R1 Mound complex in the Pelagia Mound Province and the R2 Mound Complex in the Logachev Mound Province. Evidence of fishing activity included lost gillnets and tangle nets used to fish for monkfish or anglerfish (*Lophius* spp.) and hake (*Merluccius merluccius*) based on high-resolution side-scan sonar, and the use of tangle nets on the Logachev Mounds has been observed (A. Wheeler, pers. comm.). Further, trawl marks have been imaged between mounds.

8.2.6 France, Spain, and Portugal

No new information is available for these countries.

8.2.7 USA

Although not requested from the European Commission, some information is provided here on coral distribution in the western North Atlantic. An unpublished overview of reported soft cold-water corals off the eastern USA (L. Watling, pers. comm.) summarized historical observations based largely on 761 records from underwater camera tows and from dives with the submersible "Alvin". Studies to identify the distribution of hard cold-water corals are in progress.

8.2.8 Canada

There are approximately 35 species of corals identified in the waters off Nova Scotia and Newfoundland (Breeze *et al.*, 1997; MacIsaac *et al.*, 2001). Of these, seven species are Alcyonacea (soft corals), ten species are Gorgonacea (horny corals), ten species are Scleractinia (stony corals), and eight species are Pennatulacea (seapens). The corals in this area do not build reefs.

8.3 New information on the impacts of fishing on cold-water corals

As noted in 2002 advice from ICES to the European Commission (ICES, 2002), *Lophelia pertusa* reefs are particularly affected by mobile bottom-fishing gears and the use of such gears is widespread in areas holding *Lophelia*. Static gear fisheries also occur in *Lophelia* reefs but cause much less damage than do mobile fishing gears.

Evidence of the occurrence of static gears such as gillnets and tangle nets was obtained during the cruises off Ireland described above. Eight examples of lost nets were found during the video survey of the Thérèse Mound (Figure 8.3.1), along with evidence of previous scientific dredge surveys of the area (Grehan *et al.*, 2003).

An Irish orange roughy trawl fishery utilizing sophisticated acoustic instruments has expanded rapidly since 2001. This fishery uses high-resolution colour sounders and bottom sonars to locate pinnacles/seamounts prior to fishing. The headlines of the trawls are fitted with high-resolution transducers so that the actual position of the net with respect to the bottom can be monitored and controlled in real time. Orange roughy tend to aggregate near the summit of topographical highs. Fishing involves shooting the trawl to pass close to the summit of the peak, allowing the net to sink quickly to drive the orange roughy onto the sea floor, before towing off into deep water. This is a high-risk fishery as under-shooting the trawl during the initial pass will result in snagging of the net on the side of the pinnacle/seamount. In the Southern Hemisphere, particularly in Australia and New Zealand, this bottom trawl fishery has had a major impact on seamount coral ecosystems (Probert *et al.*, 1997; Koslow *et al.*, 2000, 2001). An exposure of these seamount coral ecosystems to a relatively short period of fishing would likely have significant, permanent, negative effects on the coral habitat.

Observations made in the northeast Skagerrak by Lundälv and Jonsson (2003) found that three of six sites containing or known to have contained coral have been destroyed by mobile fishing gears in relatively recent times, while two of the remaining three sites with live corals were heavily impacted. The main bottom trawl fishery in this area is for shrimp (*Pandalus borealis*). Fishermen have reported reduced catches of shrimp in the areas near corals after trawling activities, which could indicate that destruction of these coral habitats has a negative effect on associated fauna.

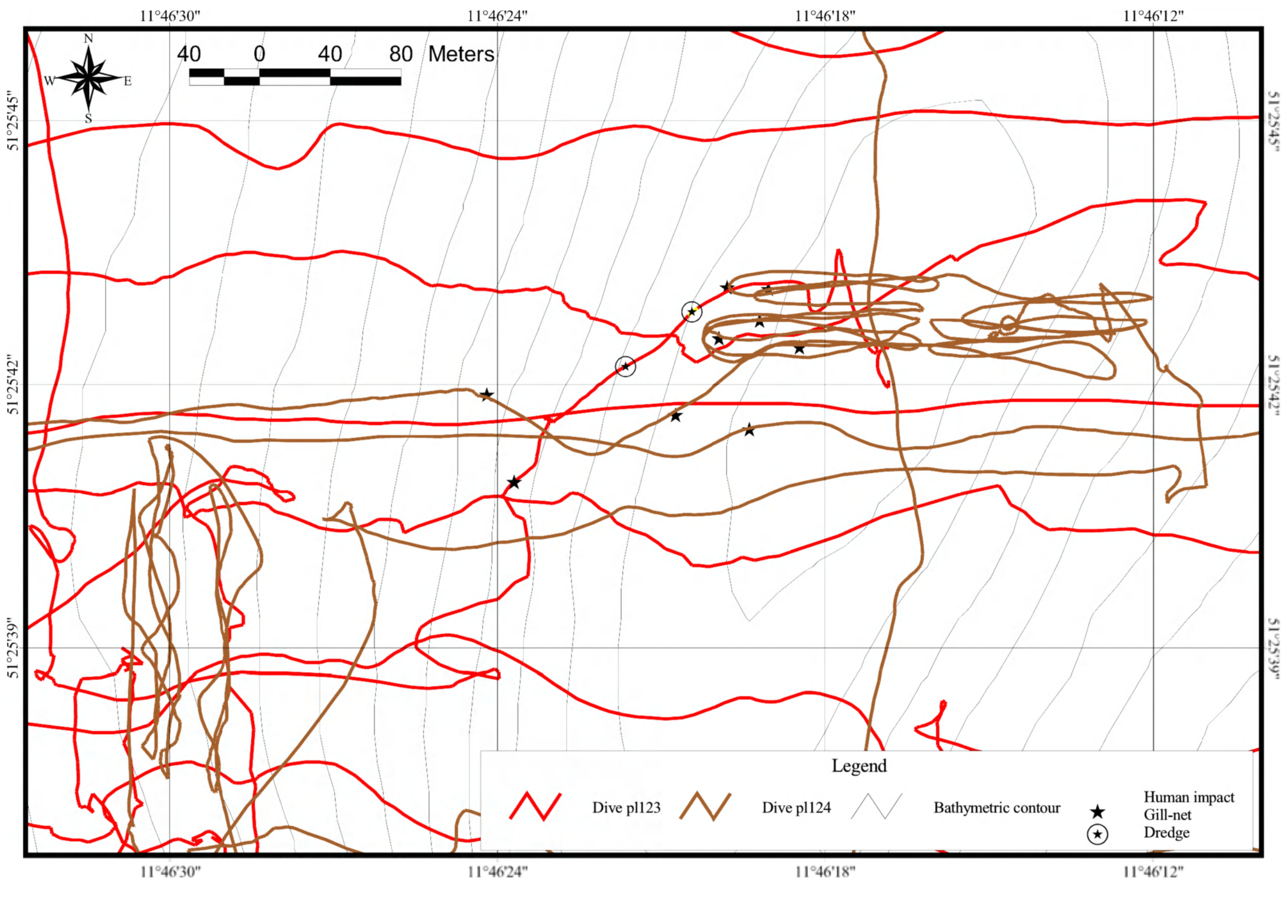


Figure 8.3.1. Enlarged section of the western flank of the Thérèse Mound showing the location of the gillnet scars (stars) observed by the downward-looking video camera (tracks in red and brown) and dredge scars (enclosed stars) (Grehan *et al.*, 2003).

8.4 Location of areas to protect from deep-water trawling

8.4.1 Norway and Sweden

Norway closed the areas of the Sula Ridge and Iverryggen reefs to towed gears in 2000. On 4 January 2003, an area 43 km long and 6.8 km wide was closed by the Norwegian Minister of Fisheries to bottom-trawl gears in order to protect the newly discovered reef to the west of Røst. The area is delineated by four coordinates: 67° 36.2' N, 009° 32.9' E; 67° 33.8' N, 009° 40.2' E; 67° 17.3' N, 008° 57.1' E; and 67° 19.8' N, 008° 49.5' E. The Worldwide Fund for Nature (WWF) has suggested that the Yttre Hvaler reef should be incorporated into a Norwegian-Swedish protected area. Some 426 km^2 of the Koster-Väderöfjorden, in the Swedish part of the area, have already been declared a Special Area of Conservation under the EU Habitats Directive. A working group, consisting of representatives from public authorities, fishing organizations and individual fishermen, has been successfully working to reduce the effects of shrimp trawling on the sensitive marine organisms in the area. Certain gear regulations have been introduced and a number of small areas within the NATURA 2000 area are being identified as protected zones where trawling is forbidden. In Norway, the area is listed as a candidate area in the national marine protection plan currently under development. Under this plan, a network of Marine Protected Areas (MPAs) will be established in Norwegian waters in 2004. The area is also being considered as a nature reserve or national park under the Norwegian Nature Conservation Act.

8.4.2 United Kingdom

In 2002, ACE noted that since *Lophelia* appears to need or favour the elevation provided by sand mounds in the Darwin Mounds area, it seems likely that damage to the corals and mounds will be permanent (ICES, 2002). This site is regarded to be at particularly high risk of further permanent damage. The UK has indicated to the European Commission that it will be proposing the Darwin Mounds site as a Special Area of Conservation under the EU Habitats Directive as soon as national regulations are in place. The UK has also drawn attention to the need for the Commission to exercise its sole competency in fisheries management in EU waters in regulating fishing in the area of the Mounds. A possible boundary for the site has been proposed (Figure 8.2.4.2). The boundary corner points are 59° 54' N, 7° 39' W; 59° 54' N, 6° 47' W; 59° 37' N, 6° 47' W; and 59° 37' N, 7° 39' W. This margin between the Mounds and the boundary is sufficiently wide to allow for the length of towing warp between any trawler and a net being fished on the seabed in this area. Fishing vessels which are bottom trawling in the region need a minimum towline length of twice the depth of the water in which they are fishing (SERAD, 2001). The Darwin Mounds are in water between approximately 1,000 m and 1,100 m deep and, therefore, minimum towline lengths are likely to be between 2 km and 2.2 km long. The suggested site boundary for the Darwin Mounds comprises the smallest rectangle based on whole degrees/minutes (to two decimal places), which will include the Mounds area plus a margin of 2.2 km to allow for possible impacts of trawling, as outlined above. The total area covered by the proposed boundary option is about 1,529 km^2.

However, if fisheries monitoring near the Mounds is to use the satellite-based VMS system (Hall-Spencer, 2003; Marrs and Hall-Spencer, 2003) currently in use in EU waters, then the boundary where fishing vessels should not go may need to be drawn wider still. This is because the VMS cycle rate is once every two hours. A further margin of at least the equivalent of 1–2 hours steaming time may therefore need to be added to the site boundary in order to ensure that fishing vessels cannot tow undetected over the site. An alternative might be to modify the VMS system to give more frequent positional updates, or to randomize the timing of positional updates such that it is impossible for any fisher to know when VMS signals might be transmitted.

8.4.3 Ireland

The Irish Coral Task Force is at the beginning of a process of scientific assessment and advice regarding possible sites for the conservation of cold-water corals. Four areas of research activity, where surveying has been conducted, have been brought to the attention of the Task Force (Figures 8.2.5.1 and 8.4.3.1). The Irish Coral Task Force has been formally asked by Duchas, the Irish Heritage Service, to assist with the designation of offshore Special Areas of Conservation to protect *Lophelia* reefs under the EU Habitats Directive (A.J. Grehan, pers. comm.).

8.4.4 Canada

Through video surveys in the Northeast Channel, Mortensen *et al.* (2003) observed signs of fishing impact in the form of broken live corals, tilted corals, and scattered skeletons. Lost fishing gear was often found entangled in corals. Broken or tilted corals were observed along 29% of the study sites. In total, 4% of the coral colonies were impacted. Based on these findings, a combined working group with members from the Department of Fisheries and Oceans and the fishing industry established a coral conservation area which centered on Romey's Peak, in the Northeast Channel (Figure 8.4.4.1). Scientific and conservation interest primarily focused on the large deep-sea gorgonian species (commonly referred to as coral "trees"). Because of their height off the seabed, large gorgonians are vulnerable to damage and disturbance from human activities such as fishing, oil and gas drilling, and trans-Atlantic cable laying. Management measures included restrictions (and conditions) on bottom-fishing gear to protect deep-sea corals in this area. This conservation area will provide scientists an opportunity to study these marine organisms in undisturbed habitats. The conservation area is approximately 424 km^2 in size.

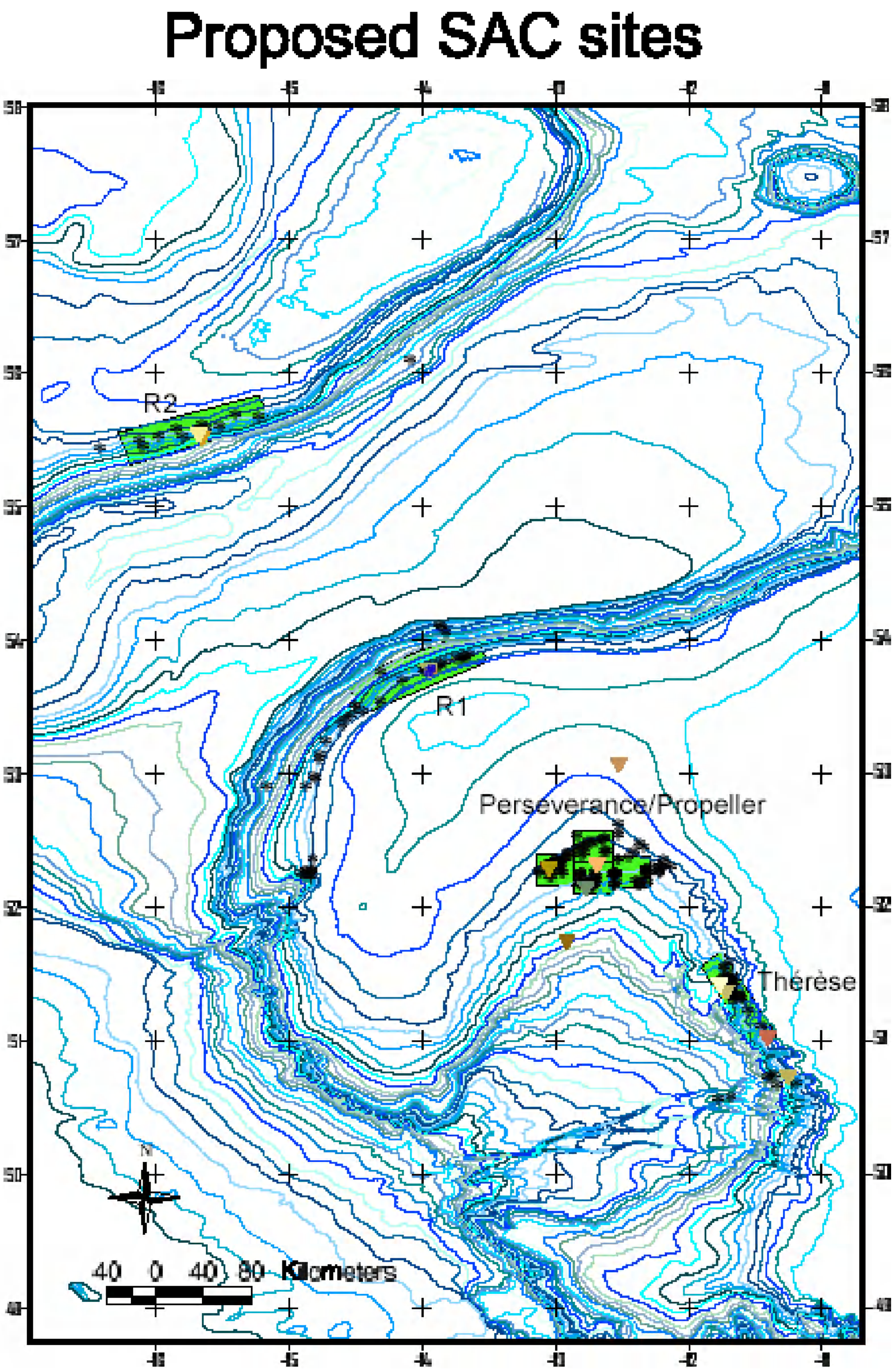


Figure 8.4.3.1. Locations where cold-water coral reefs and carbonate mounds have been studied off the west coast of Ireland. See Table 8.4.3.1 for the coordinates of these areas.

Table 8.4.3.1. Coordinates of the areas in Figure 8.4.3.1 where cold-water coral reefs and carbonate mounds have been studied off the west coast of Ireland.

Research areas		Longitude	Latitude
R2		−16.270978	55.556018
R2		−15.224803	55.83032
R2		−15.135495	55.626189
R2		−16.188049	55.313612
R1		−14.598995	53.687001
R1		−13.606095	54.076374
R1		−13.5152	53.840036
R1		−14.397819	53.466357
Thérèse Mounds		−11.903205	51.555067
Thérèse Mounds		−11.749429	51.665987
Thérèse Mounds		−11.343564	51.035761
Thérèse Mounds		−11.502381	50.933665
Perseverance/Propeller	Box 1	−12.859124	52.107253
Perseverance/Propeller	Box 1	−12.859124	52.338564
Perseverance/Propeller	Box 1	−12.569986	52.338564
Perseverance/Propeller	Box 1	−12.569986	52.107253
Perseverance/Propeller	Box 2	−12.569203	52.144644
Perseverance/Propeller	Box 2	−12.569203	52.375955
Perseverance/Propeller	Box 2	−12.280065	52.375955
Perseverance/Propeller	Box 2	−12.280065	52.144644
Perseverance/Propeller	Box 3	−13.147305	52.159463
Perseverance/Propeller	Box 3	−13.147305	52.390774
Perseverance/Propeller	Box 3	−12.858167	52.390774
Perseverance/Propeller	Box 3	−12.858167	52.159463
Perseverance/Propeller	Box 4	−12.858906	52.338841
Perseverance/Propeller	Box 4	−12.858906	52.570152
Perseverance/Propeller	Box 4	−12.569768	52.570152
Perseverance/Propeller	Box 4	−12.569768	52.338841

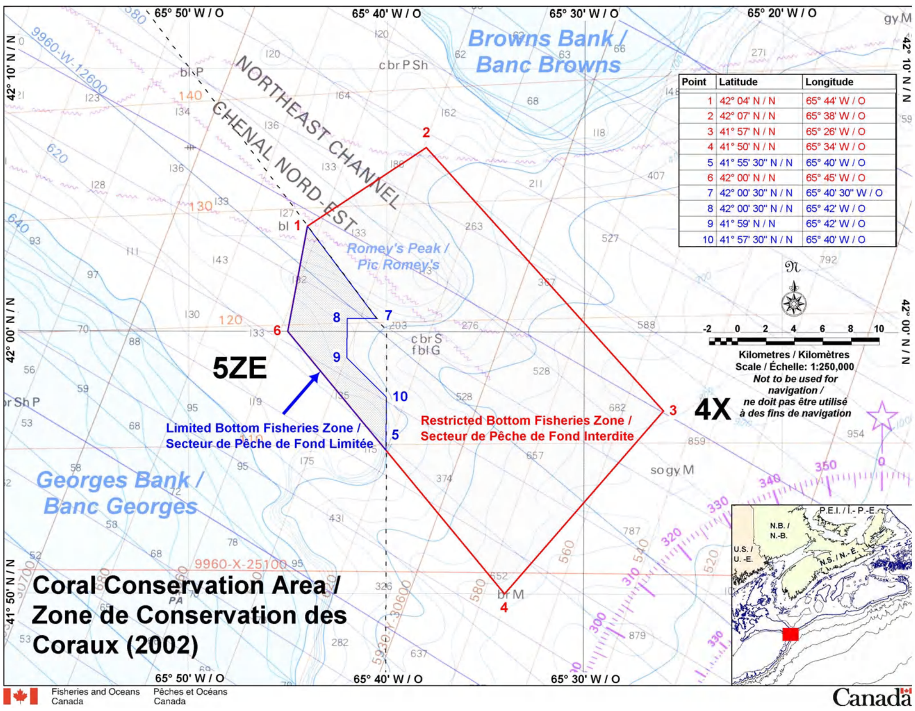


Figure 8.4.4.1. Coral conservation area in the Northeast Channel between Georges Bank and Browns Bank.

8.5 References

Akhmetzhanov, A.M., Kenyon, N.H., Ivanov, M.K., Wheeler, A.J., Shashkin, P.V., and van Weering, T.C.E. 2003. Giant carbonate mounds and current-swept seafloors on the slopes of the southern Rockall Trough. *In* European Margin Sediment Dynamics: Side-Scan Sonar and Seismic Images, pp. 203–209. Ed. by J. Mienert and P. Weaver. Springer-Verlag.

Breeze, H., Derek, D.S., Butler, M., and Vladimir, K. 1997. Distribution and status of deep sea corals off Nova Scotia. Marine Issues Committee Special Publication Number 1, Ecology Action Center, p. 158.

Croker, P.F., and O'Loughlin, O. 1998. A catalogue of Irish offshore carbonate mud mounds. *In* Proceedings of Carbonate Mud Mounds and Cold Water Reefs Conference, 7–11 February 1998, University of Gent, Belgium.

De Mol, B., van Rensbergen, P., Pillen, S., van Herreweghe, K., van Rooij, O., McDonnell, A., Huvenne, V., Ivanov, M., Swennen, R., and Henriet, J.P. 2002. Large deep-water coral banks in the Porcupine Basin, southwest of Ireland. Marine Geology, 188: 193–231.

Fosså, J.H., and Alvsvåg, J. 2003. Mapping and monitoring of coral reefs. *In* Havets Miljø 2003. Ed. by L. Asplin and E. Dahl. Fisken og Havet, Special Issue no. 2-2003, pp. 62–67.

Grehan, A.J., Unnithan, V., Olu, K., and Opderbecke, J. 2003. Fishing impacts on Irish deep-water coral reefs: making the case for coral conservation. *In* Proceedings from the Symposium on the Effects of Fishing Activities on Benthic Habitats: Linking Geology, Biology, Socioeconomics and Management. Ed. by J. Thomas and P. Barnes. American Fisheries Society, Bethesda, Maryland, USA.

Hall-Spencer, J.M. 2003. Satellite tracking trawlers to protect deep-water habitats. Proceedings of the International Society for Reefs Studies 2002, Abstracts, p 56.

Henriet, J.P., De Mol, B., Pillen, S., Vanneste, M., van Rooij, D., Versteeg, W., Crocker, P.F., Shannon, P.M., Unnithan, V., Bouriak S., and Chachkine, P. 1998. Gas hydrate crystals may help build reefs. Nature, 391: 648–649.

Hovland, M., Croker, P.M., and Martin, M. 1994. Fault-associated seabed mounds (carbonate knolls?) off Western Ireland and North-west Australia. Marine Petroleum Geology, 11: 232–246.

ICES. 2002. Report of the ICES Advisory Committee on Ecosystems, 2002. ICES Cooperative Research Report, 254: 18-30.

Jonsson, L., and Lundälv, T. 2001. Detailed mapping of a shallow occurrence of *Lophelia pertusa* (L.) in the Kosterfjord, Sweden. *In* Proceedings of the First International Symposium on Deep-sea Corals, p. 210. Ed. by J.H.M., Willison, J. Hall, S.E. Gass, E.L.R. Kenchington, M. Butler, and P. Doherty, Ecology Action Centre, Halifax, Nova Scotia.

Kenyon, N.H., Ivanov, M.K., and Akmetzhanov, A.M. 1998. Cold water carbonate mounds and sediment transport on the Northeast Atlantic margin. Preliminary results of the geological and geophysical investigations during the TTR-7 cruise of R/V "Professor Logachev" in cooperation with the CORSAIRES and ENAM2 programs, July–August, 1997. Intergovernmental Oceanographic Commission Technical Series, 52. UNESCO, Paris, France.

Kenyon, N.H., Akhmetzhanov, A.M., Wheeler, A.J., van Weering, T.C.E., de Haas, H., and Ivanov, M.K. 2003. Giant carbonate mud mounds in the southern Rockall Trough. Marine Geology, 195; 5–30.

Koslow, J.A., Boehlert, G.W., Gordon, J.D.M., Haedrich, R.L., Lorance, P., and Parin, N. 2000. Continental slope and deep-sea fisheries: implications for a fragile ecosystem. ICES Journal of Marine Science, 57: 548–557.

Koslow, J.A., Gowlett-Holmes, K., Lowry, J.K., O'Hara, T., Poore, G.C.B., and Williams, A. 2001. Seamount benthic macrofauna off southern Tasmania: community structure and impacts of trawling. Marine Ecology Progress Series, 213: 111–125.

Lundälv, T., and Jonsson, L. 2000. Inventering av Koster-Väderöområdet med ROV-teknik. Naturvårdsverket, Rapport 5079. Stockholm.

Lundälv, T., and Jonsson, L. 2003. Mapping of deep-water corals and fishery impacts in the north-east Skagerrak, using acoustical and ROV survey techniques. Abstract in 6th Underwater Science Symposium of the Society for Underwater Technology, London.

MacIsaac, K.C., Bourbonnais, E., Kenchington, E., Gordon, D., and Gass, S. 2001. Observations on the occurrence and habitat preferences of corals in the Atlantic. Proceedings of the First International Symposium on Deep-Sea Corals. Ecology Action Centre and Nova Scotia Museum, Halifax, Nova Scotia. ISBN 0-9683068-7-X.

Marrs, S., and Hall-Spencer, J.M. 2002. UK coral reefs. Ecologist, 32(4): 36–37.

Masson, D.G., Bett, B.J., Billett, D.S.M., Jacobs, C.L., Wheeler, A.J., and Wynn, R.B. 2003. The origin of deep-water, coral-topped mounds in the northern Rockall Trough, Northeast Atlantic. Marine Geology, 194: 159–180.

Mortensen, P.B., Buhl-Mortensen, L., Gordon Jr., D.C., Fader, G.B.J., McKeown, D.L., and Fenton, D.G. 2003. Effects of fisheries on deep-water gorgonian corals in the Northeast Channel, Nova Scotia (Canada). *In* Proceedings from the Symposium on the Effects of Fishing Activities on Benthic Habitats: Linking Geology, Biology, Socio-economics and Management. Ed. by J. Thomas and P. Barnes. American Fisheries Society, Bethesda, Maryland, USA.

Nilsson, P. 1997. Biological values of the Kosterfjord area – a compilation and analysis of present knowledge. Naturvårdsverket, Rapport 4749. Stockholm.

Probert, P.K., McKnight, D.G., and Grove, S.L. 1997. Benthic invertebrate by-catch from a deep-water trawl fishery, Chatham Rise, New Zealand. Aquatic Conservation of Marine and Freshwater Ecosystems, 7: 27–40.

Roberts, J.M., Long, D., Wilson, J.B., Mortensen, P.B., and Gage, J.D. 2003. The cold-water coral *Lophelia pertusa* (Scleractinia) and enigmatic seabed mounds along the north-east Atlantic margin: are they related? Marine Pollution Bulletin, 46: 7–20.

SERAD. 2001. A fishing industry guide to offshore operators. Scottish Executive, Edinburgh. 28 pp.

van Weering, T.C.E., De Haas, H., Akhmetzhanov, A.M., and Kenyon, N.H. 2003. Giant carbonate mounds along the Porcupine and SW Rockall Trough Margins. *In* European Margin Sediment Dynamics: Side-Scan Sonar and Seismic Images, pp. 211–216. Ed. by J. Mienert and P. Weaver. Springer-Verlag.

Wheeler, A.J., Bett, B.J., Billett, D.S.M., Masson, D.G., and Mayor, D. 2003. The impact of demersal trawling on NE Atlantic deep-water coral habitats: the Darwin Mounds, U.K. *In* Proceedings from the Symposium on the Effects of Fishing Activities on Benthic Habitats: Linking Geology, Biology, Socioeconomics and Management. Ed. by J. Thomas and P. Barnes. American Fisheries Society, Bethesda, Maryland, USA.

Wilson, J.B. 1979a. The distribution of the coral *Lophelia pertusa* (L.) [*L. Prolifera* (Pallas)] in the north-east Atlantic. Journal of the Marine Biological Association of the United Kingdom, 59: 149–164.

Wilson, J.B. 1979b. 'Patch' development of the deep-water coral *Lophelia pertusa* (L.) on Rockall Bank. Journal of the Marine Biological Association of the United Kingdom, 59: 165–177.

Request

The European Commission, Directorate General for Fisheries, has expressed (in a letter of 20 September 2001) its immediate interest in an "*Evaluation of the impact of current fishing practices on ... sensitive habitats, and suggestions for appropriate mitigating measures*".

Source of information

The 2003 Report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO) (ICES CM 2003/ACE:05).

Summary

Using the EUNIS system of habitat classification, information on the distribution of sensitive habitats within the ICES area was collated and summarized. Sensitivity was defined according to the OSPAR Texel/Faial criteria, but it is noted that other criteria have been used by other organizations to define habitat sensitivity. From the review of the distribution of sensitive habitats, the most urgent requirement for furthering the management of these areas is the need for broad regional-scale maps, and detailed habitat maps on a site-specific scale. These descriptions should be specific enough to allow site-based protection measures to be imposed with a high degree of accuracy, and should also have broad spatial coverage so that the entire area of the habitat is known. The impacts of fishing activities, using different gear types, on the habitats were assessed and it was concluded that the greatest physical impact on sensitive habitats is likely to be caused by towed gears such as dredgers, otter trawls, and beam trawls. Two case studies on the potential measures that could be applied to mitigate against the effects of fishing activities on contrasting habitats, deep-sea *Lophelia pertusa* reefs and coastal seapens and burrowing megafauna biotopes, are provided. It was concluded that the only proven method of preventing damage to deep-water biogenic reefs from fishing activities is through spatial closures to towed gear that potentially impacts the bottom, but the spatial extent of closures needs to be further considered. In the case of coastal seapens and burrowing megafauna biotopes, it was concluded that little is known about the population dynamics of the three seapen species, which is important if the consequences of fishing impact and organic pollution are to be mitigated effectively.

As little new information was available for consideration in 2003, relative to 2002, the overall conclusions are the same as those provided by ICES in 2002 (ICES, 2002a). Based on the available information it was concluded that:

1) More work is required to properly define "sensitive habitats" with respect to fishing activities and in this way to identify environments which require genuine management action. Consideration needs to be given to whether the definition should apply more to the structural and physical aspects of the habitat or to the individual species that occupy these substrates.
2) Detailed spatial mapping of sensitive habitats is required. In this regard, consideration needs to be given to the acquisition of the necessary physical and biological data at appropriate spatial scales.
3) Further data on the post-impact rates of recovery of different habitat types is required. Some useful progress has been reported by ICES (2001a, 2002b) in relation to the impacts of marine aggregate extraction on benthic habitats, but further detailed evaluation needs to be undertaken in relation to fishing impacts.
4) Spatial mapping of fishing effort, by gear type, is required at a scale appropriate to the habitat under consideration. In this regard, it is important that satellite-based information on fishing activity available to national inspectorates is made available (after removal of sensitive information) for scientific purposes.
5) In general, sufficient information exists in the scientific literature to predict the physical effects of the majority of existing fishing practices, particularly those involving the use of towed gears that directly contact the seabed, on a number of habitats that may be considered as proxies for sensitive habitats, and to suggest mitigating actions. Gaps mainly exist in relation to the effects of bottom longlining and tangle netting, and the type of mitigation measures that may be appropriate.

Recommendations and advice

ICES Member Countries should support, as appropriate, the proposed OSPAR habitat mapping project in the North Sea.

Scientific background

9.1 Introduction

There is ongoing work within ICES and OSPAR aimed at identifying and selecting a priority list of "threatened and declining" habitats. The criteria used to define such threatened and declining habitats are still in the process of development, but are likely to include a description of the regional importance of the habitat, the rate and extent of decline, the ecological importance of the habitat, and the sensitivity and recoverability of the habitat.

Habitat sensitivity can be defined in relation to the degree and duration of damage caused by a specified external factor. Sensitivity may refer to structural fragility of the entire habitat in relation to a physical impact, or to intolerance of individual species comprising the habitat to environmental factors such as exposure, salinity fluctuations or temperature variation.

9.2 Distribution of sensitive habitats

A number of organizations have worked over the years to compile lists of terrestrial and marine habitats that they consider to be sensitive or under threat according to different criteria. In 2002, ACE reviewed a number of different definitions of "habitat sensitivity", including that given in MacDonald *et al.* (1996), along with a range of proposed schemes to quantify habitat sensitivity (Gundlach and Hayes, 1978; Anderson and Moore, 1997; Cooke and McMath, 1998) as well as the so-called "Texel/Faial" criteria, developed by OSPAR, for the identification of habitats in need of protection, conservation and, where practical, restoration and/or surveillance or monitoring. ICES (2002a) reviewed criteria developed to quantify and rank the sensitivity of habitats and concluded that more work was required to properly define "sensitive habitats" with respect to fishing activities and in this way to identify environments which require management action. Consideration needs to be given to whether the definition should apply more to the structural and physical aspects of the habitat or to the individual species that occupy these substrates.

ICES (2001b) previously reviewed several habitat classification systems and selected EUNIS as appearing to be most appropriate for wide application. Using this classification system, and based on the OSPAR Texel/Faial definition of a "sensitive habitat", i.e., one which is "easily adversely affected by a human activity, and/or if affected is expected to only recover over a very long period, or not at all", in 2003 WGECO collated information on the distribution of sensitive habitats. The EUNIS classification system has also been adopted by other organizations, most notably by OSPAR and the Worldwide Fund for Nature (WWF), who have in turn compiled listings of marine habitats congruent with EUNIS. This approach demonstrates the potential utility for management purposes of habitat maps based on the EUNIS classification. In order to make use of this approach, however, habitat maps for the management area will be required at scales consistent with those management areas.

A comprehensive evaluation of species and habitat sensitivity has also been developed as part of the UK Marine Life Information Network (MarLIN) programme (Tyler-Walters *et al.*, 2001) administered by the Marine Biological Association of the UK. This gives biology and sensitivity key information on habitats, biotope complexes, and biotopes found within Annex I habitat types in the UK candidate marine Special Areas of Conservation (SACs) or addressed under the UK Biodiversity Action Plan.

A summary of the habitats considered to be sensitive, based on the EUNIS classification, and cross-referenced with the OSPAR, MarLIN, and WWF lists of threatened and declining species in the ICES area, is given below. This assessment should be considered as a review of the major sensitive species that have been identified by a number of organizations, but it is not definitive. It should be noted that the MarLIN list is confined to relatively inshore habitats and does not include the deep sea.

Carbonate mounds

This habitat is recognized by both OSPAR and WWF as being sensitive. While the full distribution of such mounds within the ICES area is not fully known, they are thought to be widely distributed on the eastern margin of the North Atlantic from the Iberian Peninsula to offshore Norway (Masson *et al.*, 1998). A number of precise locations are available, but information on the precise location of these habitats exists only for a few cases, and so is incomplete.

Seamounts

Seamounts occur along the Mid-Atlantic Ridge in chains and are large features, as wide as 100 km across the base. Their general location is well known and documented from the west of Portugal on the Madeira-Tore rise and the Milne seamounts to the east of the Mid-Atlantic Ridge going northwards up past the Rockall Bank (Gubbay, 1999). Some specific sites are known and the Banco Gorringe Site has been listed by WWF. While adequate information regarding the physical location of seamounts within the ICES area exists from bathymetric charts, knowledge as to the flora and fauna that inhabit them and their sensitivity to fishing exists in only a limited number of cases.

Deep-sea sponge aggregations

This habitat is recognized by OSPAR in its recent list of threatened and declining habitats. In limited areas, deep-sea sponge aggregations occur in very high densities and can make up more than 90% of the catch biomass, excluding benthic fish. It has been reported that one study off the coast of northern Norway took grab samples from an area of less than 3 m^2, yielding 4,000 sponge specimens belonging to 206 species, 26 of which had not yet been described (Konnecker, 2002). They are recognized as sensitive habitats by both OSPAR and WWF. Precise locations are known for a limited number of examples of this habitat close to the shelf break around the Faroe Islands (Klitgaard and Tendel, 2001), along the Norwegian coast up to West Spitzbergen and Bjørnya (Blacker, 1957; Dyer *et al.*, 1984; Fosså and Mortensen, 1998), and from the Porcupine Seabight (Rice *et al.*, 1990), however, they have not all been mapped.

Hydrothermal vents

Hydrothermal vents are recognized as sensitive habitats by both OSPAR and WWF. The animal communities associated with hot water vents caused by hydrothermal activity are particularly unusual. They derive their energy under conditions where photosynthesis is not possible, as well as resisting extreme temperatures and potentially toxic concentrations of various heavy metals (Tunnicliffe *et al.*, 1998). Hydrothermal vent habitats occur in areas of deep-sea tectonic activity. In the ICES area, they are confined to the Mid-Atlantic Ridge, and at the present time the locations of four vent fields, the Menez Gwen, Lucky Strike, Saldanha, and Rainbow Vents, are known to the southwest of the Azores. While adequate information exists regarding the geographical location of a number of hydrothermal vent habitats, the overall distribution of this habitat and its precise locations within the ICES area remain incomplete.

***Lophelia pertusa* reefs**

This habitat is recognized as being sensitive by OSPAR, WWF, and the UK Joint Nature Conservation Committee (JNCC), and is the subject of a specialist study group within ICES (SGCOR). *L. pertusa* is a reef-forming cold-water coral with a wide distribution ranging from 55°S to 71°N (Dons, 1944; Cairns, 1994). Several publications appeared during 2002 and 2003 that have furthered knowledge of the occurrence of cold-water coral in the Northeast Atlantic from that reported in the 2002 ACE report (ICES, 2002a). While the largest concentrations currently appear to be off the coast of Norway, there are other outcrops off the Iberian Peninsula, around the Rockall Bank off Ireland, and off the Faroe Islands. Adequate information exists regarding the geographical location of an incomplete number of *Lophelia pertusa* reef habitats. The full extent of this habitat within the ICES area remains unknown. A detailed review of the current known distribution of these cold-water coral reefs and the impacts of fishing activities on these habitats is presented in Section 8 of this report.

Sabellaria spinulosa

MarLIN lists this habitat as being covered by the UK Biodiversity Action Plan and the EU Habitats Directive. This is a rare habitat, with two records from northeast England and one from the Gower peninsula, Wales. MarLIN states that this habitat is highly sensitive to substratum loss, smothering, and physical abrasion, with low evidence of recovery in the latter case. Adequate information exists regarding the geographical location of the few *Sabellaria spinulosa* habitats that exist around the UK coastline. Knowledge of the full extent of this habitat within the ICES area, however, remains incomplete.

Sabellaria alveolata

MarLIN describes this habitat as restricted to the south and west coasts of the UK, with the eastern limit in Lyme Bay and the northern limit in the Solway Firth; it is also found in the southwest and west of Ireland. MarLIN states that this habitat is highly sensitive to substratum loss and physical abrasion. Adequate information exists regarding the geographical location of *Sabellaria alveolata* reef habitats around the UK coastline. Knowledge of the full extent of this habitat within the ICES area, however, remains incomplete.

Serpulid reefs on very sheltered circalittoral muddy sand

MarLIN describes this habitat as being extremely rare. Serpulid reefs are known from Loch Creran in Scotland and at locations in County Galway, Ireland. MarLIN also states that these reefs are very sensitive to substrate loss, abrasion, and displacement. Adequate information exists regarding the geographical location of serpulid reef habitats around the UK coastline. Knowledge of the full extent of this habitat within the ICES area, however, remains incomplete.

Maërl beds

MarLIN lists two maërl habitats under the EU Habitats Directive and the UK Biodiversity Action Plan, featuring the species *Phymatolithon calcareum* and *Lithothamnion glaciale*. The latter species is commercially dredged. Both species occur within the photic zone along the west coasts of Scotland and Ireland. Both species are listed as being highly sensitive to substratum loss and smothering, with very low rates of recovery. Adequate information exists regarding the geographical location of maërl bed habitats within the ICES area.

Ampharete falcata

Ampharete falcata forms dense stands of tubes which protrude from muddy sediments, creating a physical habitat for both itself and a range of other species. It is especially common with *Parvicardium ovale* on cohesive, muddy, very fine sand near margins of deep, stratified seas. MarLIN describes this habitat as being highly sensitive to substratum loss and smothering, although the rate of recovery is suggested as being high. MarLIN lists occurrences of this habitat off the west coast of Scotland and in the Irish Sea. Adequate information exists regarding the geographical location of the *Ampharete falcata* habitats that exist around the UK coastline. Knowledge of the full extent of this habitat within the ICES area, however, remains incomplete.

***Modiolus modiolus* beds**

Modiolus beds form a physical habitat in tide-swept areas for a number of other species, including hydroids, sponges, tubeworms, and barnacles. This habitat is highly sensitive to substratum loss and abrasion, with a low rate of recovery. MarLIN shows a number of locations of this habitat, ranging from the west coast of Scotland to Wales and Northern Ireland. Adequate information exists regarding the geographical location of the *Modiolus* habitats that exist around the UK coastline. Knowledge of the full extent of this habitat within the ICES area, however, remains incomplete.

Littoral muds

Intertidal mudflats are characterized by high biological productivity and abundance of organisms, but low diversity with few rare species (Anon., 2000). They are recognized as sensitive habitats by both OSPAR and MarLIN. The largest continuous area of intertidal mudflats in the ICES area borders the North Sea coasts of Denmark, Germany, and the Netherlands in the Wadden Sea and covers around 499,000 hectares. Adequate information exists regarding the geographical location of littoral muds in the ICES area.

Seapens and burrowing megafauna

This habitat is recognized by OSPAR, MarLIN, and the JNCC as a sensitive habitat. There has been no detailed mapping of this biotope in the OSPAR Maritime Area and, therefore, no quantifiable information on changes in extent. Geographical information on this habitat is incomplete.

Littoral chalk communities

The marine communities associated with littoral chalk habitats are generally tolerant of a high degree of turbidity. The most sensitive elements of the marine communities in these habitats are likely to be the algae found in the splash zone. This habitat is recognized as sensitive by OSPAR, MarLIN, and JNCC. Coastal exposures of chalk are rare in Europe, with the greatest proportion occurring around the coast of England. There is, however, around 120 km of chalk coastline on the French coast of upper Normandy and Picardy and some chalk exposures at the coast in Denmark. Sufficient information exists regarding the location of this habitat within the ICES area.

***Ostrea edulis* beds**

Ostrea edulis beds are recognized by OSPAR as being a sensitive habitat. The habitat is also listed on the MarLIN website as having high sensitivity to substratum loss and smothering. Information on the historic distribution of *O. edulis* beds appears good, particularly for the North Sea (Korringa, 1976) and the UK (Edwards, 1997). Adequate information exists regarding the location of this habitat within the ICES area.

***Zostera* beds**

Zostera beds are recognized by OSPAR as being a sensitive habitat on the grounds of decline, ecological significance, and sensitivity, with information also provided on threat. This habitat is also recognized by MarLIN for two species, *Zostera marina* and *Zostera noltii*. This habitat is the subject of several local recovery plans around the UK coast. Distribution of *Zostera* beds within the ICES area is well known (Davison and Hughes, 1998).

9.3 Impact of current fishing practices on sensitive habitats and suggestions for mitigation measures

The spatial scale and magnitude of impact by fishing gears have previously been reviewed by ICES (2000) who concluded that the greatest physical impact on sensitive habitats is likely to be caused by towed gears such as dredgers, otter trawls, and beam trawls. Additionally, the effects of fishing, by gear type, on selected habitat types as well as appropriate mitigation measures were reviewed by ICES (2002a) who concluded that:

- Further data on the post-impact rates of recovery of different habitat types are required. Some useful progress has been reported by ICES (2002b) in relation to the impacts of marine aggregate extraction on benthic habitats, but further detailed evaluation needs to be undertaken in relation to fishing impacts.
- Detailed spatial mapping of fishing effort, by gear type, is required.
- In general, sufficient information exists in the scientific literature to predict the physical effects of the majority of existing fishing practices, particularly those involving the use of towed gears that directly contact the seabed, on a number of habitats that may be considered as proxies for sensitive habitats, and to suggest mitigating actions. Gaps mainly exist in relation to the effects of bottom longlining and tangle netting, and the type of mitigation measures that may be appropriate.

In 2003, WGECO provided two examples, or case studies, of approaches to the application of mitigation in contrasting sensitive habitats. The emphasis is on improved knowledge of habitat distribution rather than a detailed evaluation of impact and response. The two examples of the approach are illustrated below. The contrasting examples were selected because of the different levels of information available on the local and regional scale distribution of the habitats and serve to illustrate the range of data that is required.

9.3.1 Application of mitigating measures on *Lophelia pertusa* reefs

With the decline of traditional commercial species available to fishermen, increasing fishing effort is being turned towards non-traditional, deep-water species. The preferred habitats of deep-water fish such as orange roughy (*Hoplostethus atlanticus*) include seamounts and biogenic reefs, where they shoal in large numbers and can be targeted with otter trawls. Other species targeted include roundnose grenadier (*Coryphaenoides rupestris*), black scabbardfish (*Aphanopus carbo*), tusk (*Brosme brosme*), and blue ling (*Molva dypterygia*). Deep-water shark, forkbeard (*Phycis phycis*), Greenland halibut (*Reinhardtius hippoglossoides*), and various species of redfish (*Sebastes* spp.) are targeted with a variety of gear, including deep-water driftnets, gillnets, bottom longlines, or tangle nets.

ICES (2002a) noted that *Lophelia pertusa* reefs are particularly affected by mobile bottom-fishing gears and that the use of such gears is widespread in areas holding *Lophelia*. Static gear fisheries also take place in *Lophelia* reef areas, e.g., evidence of the occurrence of static gears such as gillnets and tangle nets was obtained during cruises off Ireland (Grehan *et al.*, 2003), but the damage caused appears to be much less than that caused by mobile fishing gear (see Section 8 of this report).

ICES (2002a) advised that the only proven method of preventing damage to deep-water biogenic reefs from fishing activities is through spatial closures to towed gear that potentially impacts the bottom. While this would undoubtedly be effective in preserving the *Lophelia* habitat, the practicality of closing all such habitats to trawl fishing is open to question.

WWF has recommended the establishment of Marine Protected Areas (MPAs) on the Sula Ridge and Reef, the Røst Reef, and Kosterfjorden/Yttre Hvaler off Norway, areas of the Rockall Bank, Trough, and Channel off Ireland, and the Darwin Mounds off the northwest of Scotland. Norway closed the areas of the Sula Ridge and Iverryggen reefs to towed gears in 2000. In 2003, an area 43 km long and 6.8 km wide was closed by the Norwegian Minister of Fisheries to bottom-trawl gears in order to protect the newly discovered reef to the west of Røst. In Sweden, two reef areas in the Kosterfjord are now protected and management measures have been agreed with local fishermen, who now avoid fishing in the area.

A letter from the Scottish Fishermen's Association in IntraFish in 2002 expressed the need for better coral distribution maps so that fishermen can avoid these areas and thereby the high costs associated with damaged nets and poor fish quality. There are indications that closure of a defined area of *Lophelia* habitat in the interests of conservation may also be viewed positively by the fishing industry.

In addition, in EU waters, the Habitats Directive requires the conservation of reef habitat. This is widely interpreted to include *Lophelia* coral reefs. Few EU Member States yet have the legal powers to designate Special Areas of Conservation (SAC) beyond territorial limits (12 n.m.), but some, including the UK and Ireland, are expected to have such powers within the next year (see Section 8 of this report). The understanding in the UK is that, once a candidate SAC has been notified to the European Commission, the Member State will be obliged to protect that SAC from harm from those activities which it has exclusive competence to regulate (e.g., fisheries).

9.3.2 Application of mitigating measures on seapens and burrowing megafauna

The "seapens and burrowing megafauna" biotope occurs on sandy and muddy substrata in sheltered, fully marine conditions. It is characterized by three species of colonial anthozoans (*Virgularia mirabilis*, *Pennatula phosphorea*, and *Funiculina quadrangularis*), and other burrowing megafauna species which construct large, long-lasting burrows in the bottom sediments. The burrowing megafauna is a taxonomically diverse assemblage of crustaceans. Seapens and burrowing megafauna coexist in many sediment biotopes. However, these two groups of organisms are functionally and ecologically dissimilar and are not always associated with each other. Various megafaunal burrowers are preyed upon by fish, but very few specialist predators of seapens are known from British waters (Hughes, 1998).

The biotope complex supports a major fishery for one of its characteristic species, *Nephrops norvegicus*, and so is of considerable economic importance, and the impacts of this gear and the fishery have already been reviewed by ICES (2002c).

Collectively, the biotope complex is found widely around the British Isles and the coasts of continental Europe, especially in the Adriatic and Aegean Seas. Examples are known from a number of sites and are generally well described and mapped, for example, seapen habitat distribution for the northeast coast of England (Figure 9.3.2.1). SAC management has already protected several example habitats, although detailed information on community composition and species abundance is generally lacking (Hughes, 1998).

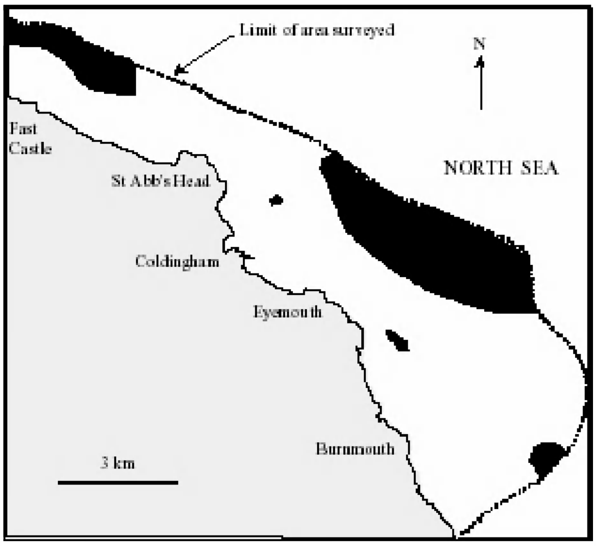


Figure 9.3.2.1. Distribution of fine silty sand with *Virgularia* and burrowing megafauna off the Berwickshire/North Northumberland coast (from Hughes, 1998).

Biotopes included within this complex occur from the shallow subtidal (<10 m) to over 100 m in depth in conditions of fully marine salinity. They are always highly sheltered from wave action and some megafaunal species are characteristic of muddy, organic-rich substrata. Megafauna are usually absent from coarse sands, probably because they are unfavourable to burrow maintenance and generally have a low organic content.

Observational evidence from towed video and diving surveys suggests that organic pollution and trawling for *Nephrops norvegicus* are the two human activities most likely to affect the biotope complex. The effects of trawling on burrowing megafauna are not fully understood, but of the three seapen species, *Funiculina quadrangularis* is likely to be the most vulnerable to trawl damage because of its brittle stalk and inability to retract into the sediment. However, experimental studies show that all three species can re-anchor themselves in the sediment if dislodged by fishing gear. Oxygen depletion caused by heavy organic pollution is probably the most damaging consequence of organic enrichment. Little is known about the population dynamics of the three seapen species, which is important if the consequences of fishing impact and organic pollution are to be mitigated effectively.

9.4 Summary conclusions

More work is required to develop criteria for evaluating and ranking the sensitivity of habitats with respect to fishing activities and in this way to identify environments which require management action. Consideration needs to be given to how the application of the criteria would use information on the structural and physical aspects of the habitat and the individual species that occupy these habitats.

In general, sufficient information exists in the scientific literature to predict the physical effects of the majority of existing fishing practices, particularly those involving the use of towed gears that directly contact the seabed, on a number of habitats that may be considered as proxies for sensitive habitats, and to suggest mitigating actions. Gaps mainly exist in relation to the effects of bottom longlining and tangle netting, and the type of mitigation measures that may be appropriate. In many cases, however, the precise nature of the impact is inferred from nearby examples where site-specific data are lacking. Although not ideal, this is normally sufficient to generate action, and can be followed up with specific studies as part of a detailed management framework.

The main problem, which is currently inhibiting the ability to select sites for protection, relates to the location of habitats. As can be seen from the review of the distribution of sensitive habitats above, the most urgent requirement for furthering the management of these areas is the need for broad regional-scale maps, and detailed habitat maps on a site-specific scale. These descriptions should be specific enough to allow site-based protection measures to be imposed with a high degree of accuracy, and should also have broad spatial coverage so that the entire area of the habitat is known. ICES notes that a mapping strategy to provide such information has been proposed for the North Sea and details are given in Section 7 of this report.

Several countries have designated, or are considering designating, locations containing sensitive habitats as Marine Protected Areas (see Section 8 of this report). The availability of habitat maps at the appropriate scale is a prerequisite for the determination of an appropriate scale of closure for such areas. In addition, consideration needs to be given to the type and extent of fishing activity as well as the ability of regulatory authorities to carry out surveillance and monitor compliance with closures. It is also important to ensure that the most appropriate proportion of the habitat is protected, while taking into consideration issues such as the spatial continuity of multiple sites on a regional basis. This will help to underpin the selection of optimum sites and areas of a habitat. In general, the scientific community is not well-equipped to advise on the science behind this aspect of site protection, yet it is fundamental both to the establishment of ecologically coherent networks of closed areas and for the study of meta-population dynamics in affected sites. Further development of this area of science will provide much-needed advice on the most appropriate scale of closure for particular habitats.

9.5 References

Anderson, S., and Moore, J. 1997. Guidance on assessment of seabed wildlife sensitivity for marine oil and gas exploration. A report to JNCC from OPRU, Neyland, UK. Report No. OPRU/18/96.

Anon. 2000. UK Biodiversity Group Tranche 2 Action Plans. Volume V – maritime species and habitats. English Nature, Northminster House, PE1 1UA. ISBN 1 85716 467 9. 242 pp.

Blacker, R.W. 1957. Benthic animals as indicators of hydrographic conditions and climatic change in Svalbard waters. Fishery Investigations (London), ser. II, 20: 1–49.

Cairns, S.D. 1994. Scleratinia of the Temperate North Pacific. Smithsonian Contributions to Zoology, 557. 150 pp.

Cooke, A., and McMath, M. 1998. SENSMAP: Development of a protocol for assessing and mapping the sensitivity of marine species and benthos to maritime activities. CCW Marine Report: 98/6/1.

Davison, D.M., and Hughes, D.J. 1998. *Zostera* species: An overview of dynamic and sensitivity characteristics for conservation management of marine SACs. Scottish Association of Marine Sciences (UK Marine SACs Project), Oban.

Dons, C. 1994. Norges korallrev. Det Kongelige Norske Videnskabers Selskabs Forhandlinger, 16: 37–82.

Dyer, M.F., Cranmer, G.J., Fry, P.D., and Fry, W.G. 1984. The distribution of benthic hydrographic indicator species in Svalbard waters, 1878–1991.

Journal of the Marine Biological Association of the United Kingdom, 64: 667–677.

Edwards, E. 1997. Molluscan fisheries in Britain. *In* The History, Present Condition and Future of the Molluscan Fisheries of North and Central America and Europe, vol. 3, Europe. Ed. by C.L. MacKenzie Jr., V.G. Burrell Jr., A. Rosenfeld, and W.L. Hobart. National Oceanic and Atmospheric Administration, NOAA Technical Report NMFS, 129.

Fosså, J.H., and Mortensen, P.B. 1998. Artsmangfoldet på *Lophelia*-korallrev langs norskekysten. Forekomst og tilstand. Fisken og Havet nr. 17. 95 pp. (in Norwegian).

Gubbay, S. 1999. Offshore Directory. Review of a selection of habitats, communities and species of the North-East Atlantic. Report for WWF-UK, North East Atlantic Programme.

Gundlach, E.R., and Hayes, M.O. 1978. Classification of coastal environments in terms of potential vulnerability to oil spill damage. Marine Technical Society Journal, 12(4): 18–27.

Grehan, A.J., Unnithan, V., Olu, K., and Opderbecke, J. 2003. Fishing impacts on Irish deep-water coral reefs: making the case for coral conservation. *In* Proceedings from the Symposium on the Effects of Fishing Activities on Benthic Habitats: Linking Geology, Biology, Socioeconomics and Management. Ed. by J. Thomas and P. Barnes. American Fisheries Society, Bethesda, Maryland, USA.

Hughes, D.J. 1998. Sea pens and burrowing megafauna. An overview of dynamics and sensitivity characteristics for conservation management of marine SACs. Scottish Association for Marine Science (SAMS), UK Marine SACs Project, Vol. III.

ICES. 2000. Report of the ICES Advisory Committee on the Marine Environment, 2000. ICES Cooperative Research Report, 241: 9–26.

ICES. 2001a. Effects of Extraction of Marine Sediments on Marine Ecosystems. ICES Cooperative Research Report, 247.

ICES. 2001b. Report of the ICES Advisory Committee on Ecosystems, 2001. ICES Cooperative Research Report, 249: 18–30, 60–66.

ICES. 2002a. Report of the ICES Advisory Committee on Ecosystems, 2002. ICES Cooperative Research Report, 254: 31–38.

ICES. 2002b. Report of the Working Group on the Effects of Extraction of Marine Sediments on the Marine Ecosystem. ICES CM 2002/E:06.

ICES. 2002c. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 2002/ACME:02.

Klitgaard, A.B., and Tendel, O.S. 2001. "Ostur" – "cheese bottoms" – sponge dominated areas in Faroese shelf and slope areas. *In* Marine biological investigations and assemblages of benthic invertebrates from the Faroe Islands, pp. 13–21. Ed. by G. Guntse and O.S. Tendal. Kaldbak. Marine Biological Laboratory, the Faroe Islands.

Konnecker, G. 2002. Sponge fields. *In* Offshore Directory. Review of a selection of habitats, communities and species of the North-East Atlantic. Ed. by S. Gubbay. WWF-UK, North East Atlantic Programme.

Korringa, P. 1976. Farming the flat oyster of the genus *Ostrea*. Developments in Aquaculture and Fisheries Science, Vol. 3. Elsevier, Amsterdam. 224 pp.

Masson, D.G., Bett, B.J., Jacobs, C.L., and LeBas, T.P. 1998. Distribution and biology of recently discovered carbonate mounds in the northern Rockall Trough. Poster presented at Atlantic Frontiers Environmental Forum, Aberdeen University, 6–7 October 1998.

MacDonald, D.S., Little, M., Eno, N.C., and Hiscock, K. 1996. Disturbance of benthic species by fishing activities: a sensitivity index. Aquatic Conservation: Marine and Freshwater Ecosystems, 6: 257–268.

Rice, A.L., Thursen, M.H., and New, A.L. 1990. Dense aggregations of a hexactinellid sponge, *Pheronema carpenteri*, in the Porcupine Seabight (northeast Atlantic Ocean) and possible causes. Progress in Oceanography, 24: 179–206.

Steingrímsson, S.A. 2002. Potential coral reefs off the south coast of Iceland. Working paper to the 2002 meeting of the ICES Working Group on Ecosystem Effects of Fishing Activities.

Tunnicliffe, V., McArthur, A.G., and McHugh, D. 1998. A biogeographical perspective of the deep-sea hydrothermal vent fauna. Advances in Marine Biology, 34: 355–442.

Tyler-Walters, H., Hiscock, K., Lear, D.B., and Jackson, A. 2001. Identifying species and ecosystem sensitivities. Report to the Department for Environment, Food and Rural Affairs, from the Marine Life Information Network (MarLIN), Marine Biological Association of the UK, Plymouth, Contract CW0826 [Final Report].

10 RELATIVE IMPORTANCE OF EXTRINSIC FACTORS ON FISH POPULATION DYNAMICS COMPARED TO FISHING

Request

The European Commission, Directorate General for Fisheries, has expressed (in a letter of September 2002) its interest in a "*consideration of the relative importance of extrinsic factors such as long-term changes in oceanographic conditions, climate change, pollution, habitat disturbance, large predators, or other factors on fish population dynamics compared to fishing*".

Source of information

The 2003 Report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO) (ICES CM 2003/ ACE:05).

Summary

The effect of the climate and oceanographic conditions on fish population dynamics has been well documented. Anthropogenic factors, particularly fishing, have an important impact, as do predation and habitat disturbance. Other factors such as pollution, disease, parasites, etc., may also affect fish population dynamics, but probably at small spatial scales.

In order to examine the relative importance of the extrinsic factors compared to fishing, three main processes in fish population dynamics were chosen: mortality, recruitment, and growth. Each of these processes is individually complex; recruitment, in particular, is the sum of many interacting processes.

Quantitative comparative analyses to contrast the relative importance of fishing and predation mortality are available for several well-studied ecosystems. The effects of climate variation and pollution have been less systematically studied, while the information on population-scale effects of the other factors is available for very few systems and not systematically available for any of them. For this reason, the comparative analysis was limited to predation, climate variation, and pollution. In the case of oceanographic and climatic factors, there are complex and multi-factorial processes. Climatic factors with a small impact in the short term, but showing a directional trend over decades, or major periodic regime shifts can have cumulative impacts much greater than estimated from analyses of data over a short series of years.

Some general statements can be made.

<u>For mortality of "adult" fish of commercially exploited species, the effect of fishing usually dominates all others</u>. However, for the main ICES areas, there is a strong tendency for the contribution of fishing to adult mortality to be less for traditional forage species than for fish that reach larger sizes. Predation is a much smaller contributor to mortality for large fish. For the forage species, however, predation mortality commonly takes 2/3 or more of the biomass annually. On a population scale, the "typical" contribution of pollution to mortality is very much less than predation and fishing. Fish kills due to oceanographic conditions appear to be even more infrequent than kills due to pollution.

<u>For recruitment, predation and oceanographic conditions are the dominant factors. The effects of oceanographic conditions are usually the dominant factor and appear to be a more dominant factor for pelagic species than for demersal species</u>. Predation mortality on pre-recruit fish is high but can be variable over time within species. The "typical" effect of fishing is much weaker. The possibility of fishing depleting spawning stock biomass (SSB) and leading to a lasting reduction in recruitment is a particularly important aspect of this relationship. The contribution of pollution to recruitment variation tends to be much less than the other factors.

<u>The relative contribution of the four factors to variation in growth is less clear.</u> There are certainly well-documented effects of temperature on growth rate, and the effects can be large. There should be an effect of fishing on growth as well, but this is considered a relatively small effect, and secondary to the effects on direct mortality and life expectancy. Pollution can undoubtedly affect the growth rate of individuals exposed to it, but again, at *population* scales it is hard to argue that it is a major effect for the "typical" condition. Finally, predation is size dependent, but is thought to be affected by variation in growth rate that occurs for other (environmental) reasons, rather than being the cause of it.

Scientific background

10.1 Introduction

Fish population dynamics are shaped by many factors. Any fish population is controlled by the balance between the number of individuals recruiting into it and the number that die. Furthermore, within a given population many processes operate differently on different age groups, and therefore population dynamics processes such as recruitment and growth are also important factors. The effect of the climate and oceanographic conditions on fish population dynamics has been well documented (e.g., Brander, 1996; Alheit and Hagen, 1997; Schwartzlose *et al*., 1999; O'Brien *et al*., 2000). Moreover, anthropogenic factors such as pollution and fishing can have an important impact on fish population dynamics (e.g., Poulard and Léauté, 2002; ICES, 2002a), as can predation by both marine birds and mammals (e.g., Bax, 1991; Overholtz *et al*., 1991) and by fish (e.g., Greenstreet *et al*., 1997; Tsou and Collie, 2001), and habitat disturbance. Other factors such as disease, parasites, etc., may also affect fish population dynamics, but probably at small spatial scales.

10.2 Approach

To make progress, it is necessary to keep the problem simple. In order to examine the relative importance of the extrinsic factors compared to fishing on fish population dynamics, three main processes in fish population dynamics were chosen. These are mortality, recruitment, and growth. Together these three processes are necessary and sufficient to describe the major trends in fish abundance and biomass (the key "dynamic" properties of fish stocks that are of interest to ICES clients). Each of these processes is individually complex; recruitment, in particular, is the sum of many interacting processes, and attempts to deal with all of the complexities would rapidly become overwhelmed with detail.

Moreover, there are many more important properties to fish populations than biomass and abundance. Biogeographical properties such as distribution and migration, for example, are ecologically important and changes to historic patterns can impact opportunities to harvest the stocks. Although biogeographical patterns are not conventionally considered "fish population dynamics" parameters, extrinsic factors can impact them in important ways. Hence, it should be kept in mind that the comparative analyses which follow only address some first-order aspects of fish population biology and life history. Many other important properties also are affected by fishing and extrinsic factors, and could be investigated fruitfully. Nevertheless, consideration of adult survivorship, recruitment variation, and growth does go to the heart of the biological properties that are at the core of annual stock assessments and, in turn, are the basis for scientific advice on management.

ICES tried to be as quantitative as possible in addressing this question. Quantitative comparative analyses that allow contrasting the relative importance of fishing and predation mortality are available for several well-studied ecosystems (the North Sea, Baltic Sea, and Barents Sea) (ICES, 1997a, 1997b, 2001a, 2002b). The effects of climate variation and pollution have been less systematically studied, and so examples of the relative importance of these extrinsic factors must be taken from many other systems. Quantitative information on population-scale effects of the other factors is available for very few systems and not systematically available for any system. So the comparative analysis is limited to four extrinsic factors: fishing, predation, climate variation, and pollution.

For the comparisons to be meaningful, the "typical" role of each factor in the variation of each population dynamics parameter must be considered. Where the effect of a factor has been quantified for many species in a system for several years, the analyses may give some indication of these "typical" roles. This is the case for fishing and predation in systems where multispecies assessments are conducted, because these necessarily include all the major species at higher trophic levels. When searching the literature widely for data on effects, there is a greater risk of finding cases that have been published just because they are extreme examples, and not typical of the role of the factor in overall fish population dynamics. This is particularly problematic for pollution and oceanographic factors, where the literature is likely to be biased, favouring reports where the factors have been shown to have large effects, and often are episodic events sometimes at local scales.

Professional judgement was applied in many places. It was necessary to arbitrarily draw some boundaries between population dynamics parameters. For example, for mortality on "mature" fish, only the fisheries and some predation estimates differentiate effects among ages, so "adult mortality" was treated very coarsely. The same considerations certainly should apply to the three other extrinsic factors (predation, climate variation, and pollution) relative to adult mortality, but are harder to apply in practice.

Similar problems were encountered when trying to evaluate the effects of fishing on recruitment dynamics. Except in a few rare cases, the main effect of fishing on recruitment, via reduction in spawning biomass, is only quantified by variation in the youngest age of recruits securely estimated by Virtual Population Analysis (VPA). Moreover, the effect of fishing on recruitment was estimated by determining the variance in recruitment that was explained by changes in spawning biomass. Such analyses assumed that all the variance in SSB was a consequence of fishing, when in reality variation in incoming cohorts does affect SSB. By attributing all the variance in SSB to fishing, we are placing an upper bound on the impact that fishing has on recruitment. The effect might be smaller, but for the comparative purposes here, the upper bound was sufficient.

Effects of oceanographic variation and pollution to a lesser extent, on the other hand, have been quantified on many components of recruitment, from adult fecundity to egg survival and transport, to larval survivorship. Predation is available by age from Multispecies VPA (MSVPA), and predation on age-one fish was taken as a representative estimate of the *relative* impact of predation on recruitment. Because of the even greater need to make arbitrary decisions about how to quantify the effects of the various extrinsic factors on recruitment, the comparisons can be considered at best to be rank-order (some processes greater than others). Whether the factor may be 50% greater in impact or 200% greater cannot be resolved from the diversity of data sources that it was necessary to use. For variation in growth due to fishing, there were not even systematic sources of information from the assessments.

Dealing with oceanographic/climate variation and its influences on the three processes is open-ended and difficult in another way, however. Oceanographic and climatic variations are complex and multi-factorial processes, operating on spatial scales from local to basin-wide and time scales from very short to multi-decadal. Analyses trying to relate variation in recruitment, growth, and mortality to "oceanographic factors" necessarily consider only a small number of surrogates that are analytically tractable, such as "temperature" or "upwelling". The true impacts of this class of extrinsic factors will be more complex than any feasible analyses will capture, so there is always a risk that the effects of these factors on recruitment, growth, and mortality will be underestimated. This risk has to be balanced against the equally unquantified (but real) risk that the published relationships represent a biased sample of all the relationships between oceanographic factors and stock dynamics. Moreover, climatic factors with a small impact in the short term, but showing a directional trend over decades, or major periodic regime shifts can have cumulative impacts much greater than estimated from analyses of data over a short series of years (O'Brien *et al*., 2000). Such situations can make the past an unreliable guide to future relationships. Taken together, these complexities mean that for any processes where oceanographic factors are found to have less influence than fishing (or other factors), this result should be considered to apply in the short and medium term. It should not be concluded that oceanographic effects will always be negligible, nor that they are fully understood.

It is important to bear in mind that no one factor acts independently, and the cumulative effects of the different factors may combine in non-linear ways to produce unpredictable outcomes. For example, if a population has already been weakened by one factor (e.g., overfishing, pollution), the influence of other factors may be greater. The extrinsic factors can also lead to secondary effects, such as an environmental variation which could lead to an increase in plankton, and thus an additional source of food for fish, leading to improved growth. This type of effect that cascades through the food web is very difficult to quantify.

In the following sections, the effect of each factor on mortality, recruitment, and growth is first considered. Then comparisons across factors are made for each of the three population dynamics factors, to evaluate their *relative* importance at each stage. The concluding section brings the information together from both perspectives and considers the implications of the various results for science and management.

10.3 Fishing

The data on fishing impacts on the three fish population dynamics parameters come from ICES assessments of stocks in the North Sea, Baltic Sea, and Barents Sea (ICES, 2001a, 2002b, 2002c).

10.3.1 Mortality

For adult mortality, the fully recruited fishing mortality (F) values from the 2002 VPAs were tabulated across species, and are presented in Table 10.3.1.1. Fs on pelagic stocks are lower than Fs on demersal stocks for both the North Sea and the Baltic Sea, but are high in all cases. For exploited stocks, the fishery commonly kills half or more of all the fully recruited demersal fish each year, and between a fifth and a third of all the fully recruited pelagic fish.

Table 10.3.1.1. Adult mortality as fully recruited F values and predation mortality as mean M2 from VPA for seven species and three ecosystems for 1985–2000.

Stock	Fishing mortality F		Predation mortality M2	
	Ages for F estim.	**Mean F 1985–2000**	**Mean M2 1985–2000 for age 1**	**Mean M2 1985–2000 for age 3**
Cod (IIIa, IV, and VIId)	2–8	0.91	0.52	0.17
Haddock (IIIa and IV)	2–6	0.94	1.06	0.08
Herring (IIIa, IV, and VIIIa)	2–6	0.52	0.45	0.17
Norway pout (North Sea)	1–2	0.63	1.60	1.24
Sandeel (IV)	1–2	0.59	0.70	1.24
Sprat (North Sea)	ND		0.73	0.30
Whiting (IV and VIId)	2–6	0.76	1.01	0.14
Baltic cod	4+	0.98	0.22*	0.20
Baltic herring (sd 25–29 and 32)	3–6	0.32	0.34	0.23
Baltic sprat	3–5	0.23	0.42	0.29
Northeast Arctic cod	4–6	0.37	1.1**	0.15**

* Age 2 is used instead of age 1 for Baltic cod.
** Mortality due to cannibalism only.

10.3.2 Recruitment

The effect of fishing on recruitment acts most directly through the reduction of the spawning biomass. We estimate the magnitude of that effect by the amount of variance captured by the stock-recruit curves calculated annually for the stocks assessed by ICES, and included in the annual reports of the Advisory Committee on Fishery Management (ACFM) (ICES, 2002a). These R^2 have both potential upward and downward biases. The upward bias occurs because for each stock ICES commonly fits at least three stock-recruit models (Beverton-Holt, Ricker, and Shepherd), and accepts the one that fits best. Such practices create the potential for goodness of fit of the functional form to be over-estimated due to the chance positioning of outliers (anomalously high recruitments), which are common for many stocks where environmental conditions allow exceptional year classes to be produced occasionally. To manage this problem, reanalysis of the data and systematic application of the Beverton-Holt relationship were carried out. The downward bias is present because fishing not just reduces SSB, but also produces a spawning biomass comprising smaller individuals and more first-time spawners. Studies with cod (*Gadus morhua*) have shown that, on a per-weight basis, larger, experienced spawners produce more eggs of higher viability than smaller, first-time spawners (Cardinale and Arrhenius, 2000; Marteinsdottir and Steinarsson, 1998; Wigley, 1999). There have, however, been too few studies to know whether this pattern will generalize to other species, so the values from the usual stock-recruitment fittings were used.

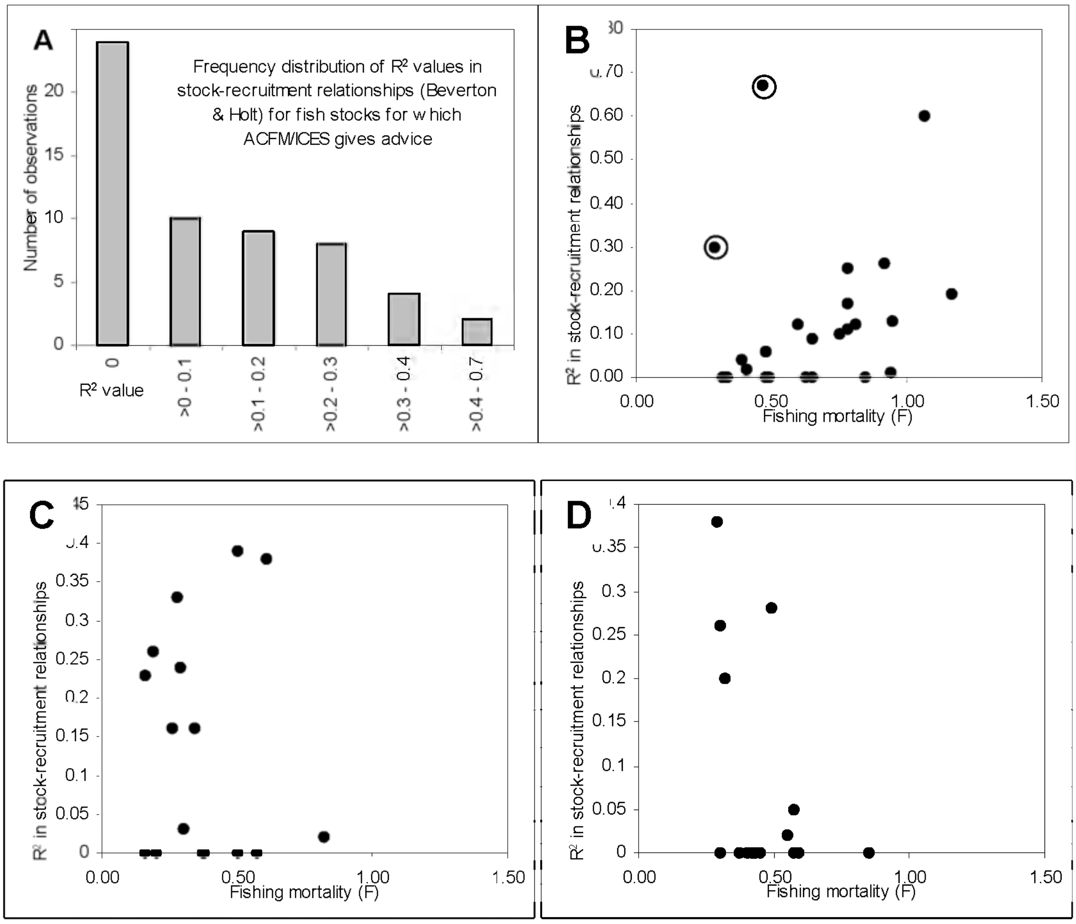


Figure 10.3.2.1. Results of stock-recruitment analyses (assuming Beverton-Holt relationships between spawning biomass and recruitment) in stocks for which ICES gives advice. Panel A summarizes the frequency distribution of R^2 values for the stock-recruitment relationship. All values are included, even those from stock-recruitment relations that are statistically non-significant. The scatterplots show the fishing mortality values versus the R^2 value for the stock-recruitment relation for demersal (panel B), pelagic (panel C), and benthic (panel D) fish. The encircled points in panel B are for two populations of hake.

The results are presented in Figure 10.3.2.1a, which shows R^2 values (the amount of variation in recruitment explained by the SSB) from the different stock-recruitment data. These values include all R^2 values, even those that were derived from stock-recruitment relations that were not statistically significant. The strength in the stock-recruitment relationships was generally quite modest, with R^2 values that were rarely greater than 0.3.

For demersal species (Figure 10.3.2.1b, cod, haddock (*Melanogrammus aeglefinus*), saithe (*Pollachius virens*), whiting (*Merlangius merlangus*), and hake (*Merluccius merluccius*), the group for which there are the most data and which is exposed to the highest fishing mortality), a statistically significant correlation was derived (Kendall's tau=0.33, $p<0.03$). Data for hake were clear outliers, and when removed from the statistical analyses, the significance increased dramatically (Kendall's tau=0.54, $p<0.005$). No clear patterns were found for pelagic fish (Figure 10.3.2.1c, herring (*Clupea harengus*), sprat (*Sprattus sprattus*), sandeel (*Ammodytes* spp.), horse mackerel (*Trachurus trachurus*), and mackerel (*Scomber scombrus*)) or benthic fish (Figure 10.3.2.1d, Greenland halibut (*Reinhardtius hippoglossoides*), megrim (*Lepidorhombus whiffiagonis*), plaice (*Pleuronectes platessa*), and sole (*Solea solea*)).

These results indicate that, at least for demersal species, high fishing mortality not only reduces spawning biomasses, but also reduces the recruitment potential.

10.3.3 Growth

As fishing reduces a population from B_{zero} (virgin biomass), the demographics of the population change to include more younger, faster-growing individuals. Hence, for the *population*, modest levels of fishing can increase growth. For higher intensities of fishing, however, fish can be caught before they have achieved their full potential. This phenomenon, referred to as "growth overfishing" (Gulland, 1983), also reflects an effect of fishing on growth at the *population* scale, but in a negative direction. No systematic tabulation of current fishing mortality relative to the F associated with growth overfishing, and relative to enhanced demographic growth, has been done (but it could be, if a priority). However, given the current F levels of most fully exploited stocks, it is likely that the detrimental effects of growth overfishing dominate over the demographic growth benefits of light exploitation.

It is well-known (Gulland, 1983) that the parameters of the von Bertallanfy growth equation for annual growth rate and maximum length are correlated. Part of the correlation is a statistical artefact, but as fishing reduces the maximum length that a population reaches, the initial growth rate commonly does increase. On the scale of year-to-year variation in growth, the effect of change in growth rate due to reduced maximum length is very small, and cannot be partitioned from a variety of other effects on observed length-at-age. The magnitude of this effect at decadal time scales has been studied. This effect is accompanied by a reduction in maximum length, however, such that the total biomass accumulated per recruit is less, even though the biomass accumulated faster.

More importantly, size-selective fishing has been shown to be selective at the genetic level. These relationships were examined in 2002 by WGECO (ICES, 2002d). It was concluded that there were lasting genetic effects of size-selective fishing on Northeast Arctic cod. Compared with the yield estimated for the stock prior to the onset of intensive fishing, the genetic effects of fishing could have reduced yield by as much as 10^5 tonnes annually. However, this effect is shared among genetically based changes in maximum length, maturation rate, and growth rates, in ways that have not been fully partitioned.

In summary, fishing unquestionably has some effect on growth. However, it is mediated to some extent by the effects of fishing on age of maturation, and maximum length in the population. The effect on growth itself is poorly partitioned from the other inter-related effects, but is not negligible.

10.4 Predation

10.4.1 Mortality

The data on predation impacts on adult mortality come from the Multispecies Assessment Working Group reports for the North Sea (ICES, 1997a) and the Baltic Sea (ICES, 2001a). Estimates of cannibalism in Northeast Arctic cod are also produced by the Arctic Fisheries Working Group.

From these sources, predation mortality estimated on age-three prey is used as an upper boundary on the rate of predation mortality experienced by adult fish. This is not absolutely true because the MSVPA does not include all predators; but for the North Sea, estimates of predation by seabirds and marine mammals are included, so most major known predators are represented. These values are considered an upper boundary because estimates of predation rate only decline with age after age three for all but a very few pelagic stocks (where older ages are uncommon anyway, and their mortality rates are poorly determined).

Predation mortality on the small pelagic sandeel and Norway pout is very high, indicating that more than 2/3 of the population is eaten each year by predators. For the Baltic stocks, Northeast Arctic cod, and North Sea sprat, predation mortality on age-three fish is around 0.3, which is noteworthy, but represents less than a quarter of the stock being eaten. For the other demersal stocks, the mortality rates are not negligible, but are even lower (see Table 10.3.1.1).

10.4.2 Recruitment

Predation on 0-group fish is thought to be very high, but also quite variable. Most quantifications of predation on 0-groups have been local studies. It has proved harder to quantify predation mortality rates for larvae on the scale of full stocks or populations. Moreover, predation is commonly size-selective, and both food supply and temperature can affect growth rates. Therefore, it is hard to disentangle multiple causal factors in very small, potentially fast-growing fish larvae. By age 1, the factors are somewhat better differentiated, and MSVPA also produces estimates of predation mortality rates for those ages. These, too, are tabulated in Table 10.3.1.1. For all stocks they are substantial, and for some stocks, particularly small pelagics again, they exceed 1.0. These values are all averages over fifteen years, and individual extreme values can be much higher.

10.4.3 Growth

No studies were found that clearly link changes in growth rate to predation. Predation is commonly known to be highly size-selective, but in fish population dynamics the causality is generally presented as going in the other direction. Growth rate varies owing to either variation in food supply or temperature. As growth varies, the time that small fish spend in the size-preference window for predators increases or decreases correspondingly. This translates into more or less predation mortality in the face of a stable predator field. Hence, predation mortality is commonly thought to be related to growth variability, but is not considered to be a major cause of it.

10.5 Climate variation

In the ICES area, the main effects of climate studied are the effects of temperature on recruitment and growth and the relationship between the North Atlantic Oscillation (NAO) index and recruitment. Relationships between oceanographic variables, and growth and recruitment exist for several species, but systematic quantifications across a number of species and years have not been undertaken for any areas. General patterns of relationships have been reported for individual oceanographic variables, however, it was not possible to tabulate the effects of climate on population dynamics parameters as they were for Table 10.3.1.1 and Figure 10.3.2.1a. This places severe limitations on the ability to conduct the necessary general comparisons.

10.5.1 Climate and recruitment

Effects of temperature variation on recruitment vary with the position of the stock relative to the range of the species (Planque and Frédou (1999) and O'Brien *et al.* (2000) for cod, Mueter *et al.* (2002) for Pacific salmon). As a generalization, oceanographic variation will have a more pronounced effect on the recruitment dynamics of the southern and northern stocks of a species than the central stocks, and this effect can be large.

The effects of upwelling on recruitment are more qualitative, and are usually given as positive or negative changes. In the ICES area, upwelling occurs mainly in Portuguese, Spanish, and French waters. In the Bay of Biscay, the upwelling intensity and the shelf stratification explained 75% of the recruitment variability of anchovy (*Engraulis encrasicolus*) between 1987 and 1996 (Allain *et al.*, 2001). In Portuguese waters, there is a spring-summer seasonal upwelling that favours sardine (*Sardina pilchardus*) and horse mackerel (*Trachurus trachurus*) recruitment survival except when a winter upwelling is observed, corresponding to the spawning season (Santos *et al.*, 2001). The variance in recruitment explained by the upwelling index is 86% for horse mackerel and 96% for sardine. This mechanism is correlated with increasing northward winds during a positive winter NAO index (Borges *et al.*, 2003). In this last case, the recruitment of both species decreases, and the total variance in recruitment is explained by upwelling intensity and NAO index.

The North Atlantic climatic variability has been largely driven by atmospheric forcing related to the North Atlantic Oscillation (NAO). The changes in climate resulting from changes in the NAO index appear to have had substantial impacts on marine ecosystems, in particular on fish productivity, with the effects varying from region to region. An examination of several species and stocks, e.g., gadoids, herring, and plankton in the Northeast Atlantic and cod and shellfish in the Northwest Atlantic, indicates that there is a link between long-term trends in the NAO and the productivity of various components of the marine ecosystem (Parsons and Lear, 2001). Quantitative relationships between the NAO index and recruitment are well documented.

10.5.2 Climate influence on growth and adult survivorship

The relationship between survival or mortality and oceanographic factors is less well documented. For cod in the northern Gulf of St. Lawrence, fish condition (Fulton's K) is lower in cold years, while mortality is higher with a range of variation in K from 0.6–0.7 (cold) to 0.85–1.0 or even higher in warm years (Dutil and Lambert, 2000). Moreover, it has been shown for ten cod stocks in the North Atlantic that K increases with the mean temperature of their habitat (Rätz and Lloret, 2003). One could then speculate that temperature-dependent mortality for cod is a more important factor for cold-water stocks. However, in the best quantified cases, such mortality only was observed to occur in extremely cold years, and its magnitude was difficult to quantify (DFO, 2003).

Variation of growth related to temperature is rather well documented for cod. As a generalization, growth always increases at warmer temperatures. Across sets of cod stocks, 73% percent of the total variance in growth rate can be explained by temperature (Brander, 1995, 2000; Dutil *et al.*, 1999).

10.6 Pollution

There are very few studies on the effects of pollutants on fish population dynamics. Two main pollutant groups were defined in order to synthesize the information available. These are xenobiotics (chemical compounds introduced into the environment by human activity, either deliberately or unintentionally, which are capable of affecting biological processes) and eutrophication and its associated low oxygen levels. Some of the effects of pollution on the three fish population dynamics processes are described schematically in Table 10.6.1.

It is rarely possible to link community-level effects to contaminant inputs. Bioassays are generally not very sensitive and biomarker responses may be linked to contaminant inputs, but are poor predictors of ecological effects. ICES has considered strategies to improve knowledge of population-scale effects of chemical contaminants (ICES, 2001b), but it is not expected to be possible in the near term to tabulate quantitative estimates of effects of pollution on recruitment, growth, or adult mortality at the scale of stocks or species.

10.6.1 Eutrophication

There is good evidence that eutrophication has affected Baltic fish communities, but evidence for population-scale effects elsewhere was not found.

First, eutrophication is thought to be one of several factors that have led to an increase in overall fish biomass in the Baltic Sea during the 20th century (Thurow, 1997). This effect is presumably a consequence of higher levels of primary production propagating up through the food web.

Second, eutrophication has contributed to a decrease in oxygen levels in the cod reproductive habitat (deep, saline layers of the Baltic) over the past century (Thurow, 1997). This, in turn, has led to reduced reproductive and recruitment success. However, the effects of eutrophication on oxygen levels and recruitment are confounded by other factors. These include a reduction in the frequency and intensity of major inflows of saline, oxygen-rich water from the North Sea-Skagerrak (HELCOM, 2002): between 1897 and 1977 such events occurred almost every year, but since 1977 there have only been two major events (1993 and 2003) (HELCOM, 2002; http://www.boos.org). In addition, an intensive fishery has decreased the size composition of the spawning stock. Since eggs produced by small Baltic cod are less buoyant than eggs produced by larger Baltic cod (Vallin and Nissling, 2000), a larger proportion of the egg production is now more susceptible to anoxic conditions than in the past.

To some extent, eutrophication-induced recruitment problems have accentuated the decline in spawner biomass caused by fishing.

Table 10.6.1. Known effects of pollution on population dynamics processes.

	Mortality	Growth	Recruitment
Xenobiotics	Pink salmon eggs experimentally exposed to "Exxon Valdez"-like PAHs: exposed and unexposed tagged juveniles were released. After two years, exposed fish experienced a 15% decrease in marine survival compared to unexposed fish (Heintz *et al.*, 2000).	Experimental exposure of English sole to PAH-contaminated sediments. Impacts on growth are linked with PAH concentration (Johnson *et al.*, 2002). Pink salmon eggs experimentally exposed to "Exxon Valdez"-like PAHs: survivors show a delayed effect on growth (Heintz *et al.*, 2000). Feeding of English sole with PAH-contaminated polychaetes: growth was lower than reference in all but one of eight groups (Rice *et al.*, 2000).	Results mainly come from *in vitro* or laboratory exposure experiments. Decrease or failure may occur due to endocrine disruption in adults or increased larval mortality (Arukwe, 2001). Field: Only 2% of the spawned eggs were affected but 25–32% of the Pacific herring embryos were damaged after "Exxon Valdez" oil spill (Carls *et al.*, 2002).
Eutrophication	N/A	Eutrophication is thought to have contributed to an overall increase in the total population production and fish biomass in the Baltic Sea, over the last century (Thurow, 1997).	Cod: Laboratory experiments correlate cod egg survival with oxygen concentration (Wieland *et al.*, 1994). Central Baltic cod: Oxygen has a significant impact on egg survival but explains very little of the variability observed in the survival between larval and 0-group stages (Köster *et al.*, 2001).

10.7 Synthesis of the relative importance of the different factors

Some general statements can be made. These are intended to describe general patterns and not to explain individual cases. Exceptions to each generalization exist, and stocks which do show the general patterns summarized below can occasionally experience large impacts of other factors.

10.7.1 Mortality

For mortality of "adult" fish of commercially exploited species, the effect of fishing usually dominates all others. However, for the main ICES areas, there is a strong tendency for the contribution of fishing to adult mortality to be less for traditional forage species (whose maximum size is generally less than about 30 cm) than for fish that reach larger sizes. Fs for the larger fish are commonly between 0.5 and 0.9, suggesting that the fishery is removing from over one third to over half of the biomass annually. Predation is a much smaller contributor to mortality for large fish, apparently rarely taking more than 10–15% of the fish of sizes that are recruiting to the fisheries. For the forage species, however, predation mortality commonly exceeds 1.0, taking 2/3 or more of the biomass annually. Pollution has been associated with local fish kills, but these are infrequent, even when eutrophication and oxygen depletion are included. On a population scale, the "typical" contribution of pollution to mortality is very much less than predation and fishing. Fish kills due to oceanographic conditions (anomalous temperatures, etc.) appear to be even more infrequent than kills due to pollution.

10.7.2 Recruitment

For recruitment, predation and oceanographic conditions are the dominant factors. Predation mortality on pre-recruit fish is high, regardless of species, but can be variable over time within species. Likewise, the effects of oceanographic conditions on recruitment can be highly variable, and can be high for some species. The effects of oceanographic conditions are especially important, and usually are the dominant factor, when accounting for the occurrence of anomalously strong or weak year classes, and tend to be a more dominant factor for pelagic species than for demersal species. Also, some of the impact of predation on recruitment may be mediated by oceanographic conditions which change the vulnerability of eggs, larvae, or juveniles to predators.

The "typical" effect of fishing is much weaker. The possibility of fishing depleting SSB and leading to a lasting reduction in recruitment is a particularly important aspect of this relationship. In mixed species fisheries, it is also possible that fishing targeted at older age classes can inflict heavy fishing mortality on larger than average individual recruit year classes (ICES, 2002a). The contribution of pollution to recruitment variation tends to be much less than that of the other factors. Nevertheless, the possibility remains that reduced fecundity has followed from pollution events, and may have been widespread, but not well documented.

10.7.3 Growth

The relative contribution of the four factors to variation in growth is less clear. There are certainly well-documented effects of temperature on growth rate, and the effects can be large. Fisheries population dynamics theory indicates that there should be an effect of fishing on growth as well, but this is considered a relatively small effect, and secondary to the effects on direct mortality and life expectancy. It has proved difficult to find estimates of the actual magnitude of the effect in the literature. Pollution can undoubtedly affect the growth rate of individuals exposed to it, but again, at *population* scales it is hard to argue that it is a major effect for the "typical" condition. Finally, predation is size-dependent, but is thought to be affected by variation in growth rate that occurs for other (environmental) reasons, rather than being the cause of it.

10.7.4 An example: cod in the North Sea, Baltic Sea, and Barents Sea

One way to compare the relative importance of several extrinsic factors on population dynamics is to investigate how much of the variation of a given population abundance or biomass can be explained by each of the factors. Two components of the population will be considered: recruitment and spawning stock.

Three cod populations from contrasting situations were chosen: the North Sea, the Baltic Sea, and the Barents Sea. Both the North Sea and the Barents Sea are on the edges of the temperature range of this species. Moreover, in the Baltic Sea, the main predator is cod (cannibalism), while it is due to other species in the other two populations. The aim of these case studies is to calculate the ratio of the variance of the spawning stock biomass (SSB) and the ratio of the variance of recruitment explained by fishing, predation, and climate (temperature) for each population and then being able to rank these factors according to the strength of their impacts on SSB and on recruitment.

The results are summarized in Table 10.7.4.1. The main impact on both SSB and recruitment is fishing. Fishing accounts for 21% to 52% of the variance. Climate and predation account for a smaller part of the variation: from 0% for predation in the Barents Sea to 18% for climate in the North Sea. In the North Sea, climate and predation have the same impact on recruitment. Then from only these three examples, we can rank the extrinsic factors by decreasing order of importance as follows: fishing, climate, and predation.

Table 10.7.4.1. Ratio of SSB and growth and recruitment variance explained by the extrinsic factors: fishing, predation, and climate; (−) means a negative effect of the factor on the population and (+) means a positive effect.

Species	Population	Factor	Population parameter	
			SSB	Recruitment
Cod	North Sea	Fishing	(−) 0.52	(−) 0.26
		Predation	(−) 0.12	(−) 0.16
		Climate		(−) 0.18
	Baltic Sea	Fishing	(−) 0.21	(−) 0.43
	Barents Sea	Fishing	(−) 0.25	(−) 0.33
		Predation	0	0
		Climate		(+) 0.18

10.7.5 Collapsed stocks

The above generalizations are based on analyses of as many stocks as possible, and hence generally refer to stocks that have not been severely depleted. For the special case of stocks that have collapsed due to overfishing, there are additional considerations. If fishing (including by-catches and illegal harvests) has been completely terminated, then fishing is not a major factor in the *current* adult mortality, and extrinsic factors are necessarily more important. However, that should not detract from the fact that fishing remains the reason that the stock is collapsed to begin with, and even small harvests of depleted stocks can represent a significant mortality source (DFO, 2003). Additionally, impacts of predators that are negligible mortality sources on healthy stocks can represent significant mortality on depleted stocks (DFO, 2003), and even small variations in recruitment or growth due to environmental factors can be large population signals. These considerations are not intended to alter the main generalizations for healthy stocks, but to warn that the recovery of depleted stocks may be more difficult than implied by the roles of extrinsic factors and even small fishery harvests on healthy stocks.

10.8 Conclusions

1) Fishing dominates all other factors with regard to impact on mortality of adult fish. Fishing often explains relatively little variation in recruitment for pelagic stocks, but usually does have some effect on the recruitment of demersal stocks through its effects on SSB size and age composition.

2) Oceanographic variation usually has an important effect on recruitment, and the effect is usually the dominant one for pelagic species. It can also have a major effect on growth, but variation in growth rate usually contributes less to variation in fish population dynamics than does variation in recruitment and adult mortality.

3) Predation is an important factor on recruitment variation, on a scale comparable to and often larger than the effect of fishing, but usually less than the effect of oceanographic variation. For small forage species, predation can be as important an influence on adult survivorship as is fishing, but is a minor factor for species with maximum sizes greater than about 30 cm.

4) No documented cases were found of lasting population-scale effects of pollution on the main population dynamics parameters. This does not mean that episodic pollution events are inconsequential for fish populations, but they usually occur on space and time scales that differ from the influences of fishing, oceanographic factors, and predation.

10.9 References

Alheit, J., and Hagen, E. 1997. Long-term climate forcing of European herring and sardine populations. Fisheries Oceanography, 6: 130–139.

Allain, G., Petitgas, P., and Lazure, P. 2001. The influence of mesoscale ocean processes on anchovy (*Engraulis encrasicolus*) recruitment in the Bay of Biscay estimated with a three-dimensional hydrodynamic model. Fisheries Oceanography, 10: 151–163.

Arukwe, A. 2001. Cellular and molecular responses to endocrine-modulators and the impact on fish reproduction. Marine Pollution Bulletin, 42: 643–655.

Bax, N.J. 1991. A comparison of the fish biomass flow to fish, fisheries and mammals in six marine ecosystems. ICES Marine Science Symposia, 193: 217–224.

Borges, M.F., Santos, A.M.P., Crato, N., Mendes, H., and Mota, B. 2003. Sardine regime shifts off Portugal: a time series analysis of catches and wind conditions. SAP Symposium Proceedings, Scientia Marina, 67(1): 235–244.

Brander, K. 1995. The effect of temperature on growth of Atlantic cod (*Gadus morhua* L.). ICES Journal of Marine Science, 52: 1–10.

Brander, K. 1996. Effects of climate change on cod (*Gadus morhua*) stocks. *In* Global warming: implications for freshwater and marine fish, pp. 255–278. Ed. by C.M. Wood and D.G. McDonald. Cambridge University Press.

Brander, K. 2000. Effects of environmental variability on growth and recruitment in cod (*Gadus morhua*) using a comparative approach. Oceanologica Acta, 23: 485–496.

Cardinale, M., and Arrhenius, F. 2000. The relationship between stock and recruitment: are the assumptions valid? Marine Ecology Progress Series, 196: 305–309.

Carls, M.G., Marty, G.D., and Hose, J.E. 2002. Synthesis of the toxicological impacts of the Exxon Valdez oil spill on Pacific herring (*Clupea pallasi*) in

Prince William Sound, Alaska, USA. Canadian Journal of Fisheries and Aquatic Sciences, 59: 153–172.

DFO. 2003. Stocks Status Reports for Cod in NAFO Division 2J3KL, 3Pn4RS, and 4TVn. Department of Fisheries and Oceans, Canada. Canadian Science Advisory Secretariat website, Reports A2-1/2003, A4-1/2003, and A3-1/2003.

Dutil, J.D., Castonguay, M., Gilbert, D., and Gascon, D. 1999. Growth, condition and environmental relationships in Atlantic cod (*Gadus morhua*) in the northern Gulf of St Lawrence and implications for management strategies in the Northwest Atlantic. Canadian Journal of Fisheries and Aquatic Sciences, 56: 1818–1831.

Dutil, J.D., and Lambert, Y. 2000. Natural mortality from poor condition in Atlantic cod (*Gadus morhua*). Canadian Journal of Fisheries and Aquatic Sciences, 57: 826–836.

Greenstreet, S.P.R., Bryant, A.D., Broekhuizen, N., Hall, S.J., and Heath, M.R. 1997. Seasonal variation in the consumption of food by fish in the North Sea and implications for food web dynamics. ICES Journal of Marine Science, 54: 243–266.

Gulland, J.A. 1983. Fish stock assessment: a manual of basic methods. Wiley, New York. xii, 223 pp.

Heintz, R.A., Rice, S.D., Wertheimer, A.C., Bradshaw, R.F., Thrower, F.P., Joyce, J.E., and Short, J.W. 2000. Delayed effects on growth and marine survival of pink salmon *Oncorhynchus gorbuscha* after exposure to crude oil during embryonic development. Marine Ecology Progress Series, 208: 205–216.

HELCOM. 2002. Environment of the Baltic Sea area 1994–1998. Baltic Sea Environment Proceedings, 82B: 24–26.

ICES. 1997a. Report of the Multispecies Assessment Working Group. ICES CM 1997/Assess:16.

ICES. 1997b. Report of the Northern Pelagic and Blue Whiting Fisheries Working Group. ICES CM 1997/Assess:14. 192 pp.

ICES. 2001a. Report of the Baltic Fisheries Assessment Working Group. ICES CM 2001/ACFM:18.

ICES. 2001b. Report of the Working Group on Biological Effects of Contaminants. ICES CM 2001/E:03. 52 pp.

ICES. 2002a. Report of the ICES Advisory Committee on Fishery Management, 2002. ICES Cooperative Research Report, 255.

ICES. 2002b. Report of the Workshop on MSVPA in the North Sea. ICES CM 2002/D:04.

ICES. 2002c. Report of the Arctic Fisheries Working Group. ICES CM 2002/ACFM:18. 529 pp.

ICES. 2002d. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 2002/ACE:03, pp. 134–142.

Johnson, L.L., Collier, T.K., and Stein, J.E. 2002. An analysis in support of sediment quality thresholds for polycyclic aromatic hydrocarbons (PAHs) to protect estuarine fish. Aquatic Conservation Marine and Freshwater Ecosystems, 12: 517–538.

Köster, F.W., Hinrichsen, H.H., St. John, M.A., Schnack, D., MacKenzie, B.R., Tomkiewicz, J., and Plikshs, M. 2001. Developing Baltic cod recruitment models. II. Incorporation of environmental variability and species interaction. Canadian Journal of Fisheries and Aquatic Sciences, 58: 1534–1556.

Marteinsdottir, G., and Steinarsson, A. 1998. Maternal influence on the size and viability of Iceland cod *Gadus morhua* eggs and larvae. Journal of Fish Biology, 52: 1241–1258.

Mueter, F.J., Peterman, R.M., and Pyper, B.J. 2002. Opposite effects of ocean temperature on survival rates of 120 stocks of Pacific salmon (*Oncorhynchus* spp.) in northern and southern areas. Canadian Journal of Fisheries and Aquatic Sciences, 59: 456–463.

O'Brien, C.M., Fox, C.J., Planque, B., and Casey, J. 2000. Climate variability and North Sea cod. Nature, 404: 142.

Overholtz, W.J., Murawski, S.A., and Foster, K.L. 1991. Impact of predatory fish, marine mammals and seabirds on the pelagic fish ecosystem of the north-eastern USA. ICES Marine Science Symposia, 193: 198–208.

Parsons, L.S., and Lear, W.H. 2001. Climate variability and marine ecosystem impacts: a North Atlantic perspective. Progress in Oceanography, 49: 167–188.

Planque, B., and Frédou, T. 1999. Temperature and the recruitment of Atlantic cod (*Gadus morhua*). Canadian Journal of Fisheries and Aquatic Sciences, 56: 2069–2077.

Poulard, J.C., and Léauté, J.P. 2002. Interaction between marine populations and fishing activities: temporal patterns of landings of La Rochelle trawlers in the Bay of Biscay. Aquatic Living Resources, 15: 197–210.

Rätz, H.J., and Lloret, J. 2003. Variation in fish condition between Atlantic cod (*Gadus morhua*) stocks, the effect on their productivity and management implications. Fisheries Research, 60: 369–380.

Rice, C.A., Myers, M.S., Willis, M.L., French, B.L., and Casillas, E. 2000. From sediment bioassay to fish biomarker - connecting the dots using simple trophic relationships. Marine Environmental Research, 50: 527–533.

Santos, A.M.P., Borges, M.F., and Groom, S. 2001. Sardine and Horse mackerel recruitment and upwelling off Portugal. ICES Journal of Marine Science, 58: 589–596.

Schwartzlose, R.A., Alheit, J., Bakun, A., Baumgartner, T.R., Cloete, R., Crawford, R.J.M., Fletcher, W.J., Green-Ruiz, Y., Hagen, E., Kawasaki, T., Lluch-Belda, D., Lluch-Cota, S.E., MacCall, A.D., Matsuura, Y., Nevarez-Martinez, M.O., Parrish, R.H., Roy, C., Serra, R., Shust, K.V., Ward, M.N., and Zuzunaga, J.Z. 1999. Worldwide large-scale fluctuations of sardine and anchovy populations. South African Journal of Marine Science-Suid-

Afrikaanse Tydskrif Vir Seewetenskap, 21: 289–347.
Thurow, F. 1997. Estimation of the total fish biomass in the Baltic Sea during the 20th century. ICES Journal of Marine Science, 54: 444–461.
Tsou, T.S., and Collie, J.S. 2001. Estimating predation mortality in the Georges Bank fish community. Canadian Journal of Fisheries and Aquatic Sciences, 58: 908–922.
Vallin, L., and Nissling, A. 2000. Maternal effects on egg size and egg buoyancy of Baltic cod, *Gadus morhua*. Implications for stock structure effects on recruitment. Fisheries Research, 49: 21–37.
Wieland, K., Waller, U., and Schnack, D. 1994. Development of Baltic cod eggs at different levels of temperature and oxygen content. Dana, 10: 163–177.
Wigley, S.E. 1999. Effects of first-time spawners on stock-recruitment relationships for two groundfish species. Journal of Northwest Atlantic Fishery Science, 25: 215–218.

11 ECOSYSTEM IMPACTS OF INDUSTRIAL FISHING

Request

The request from the European Commission, Directorate General for Fisheries, of 31 July 2002, concerning the ecosystem impacts of industrial fishing states:

We would like ICES to review the state of the knowledge about the impacts on the ecosystem of the current industrial fisheries in the ICES area. The study should cover the following areas:

1. *The removal of very large quantities of biomass from the sea and its effect on the global economy of the ecosystem;*
2. *The effects on the availability of food for important predators (other commercial fish important for human consumption, sensitive species such as marine mammals, birds, etc.);*
3. *The estimated by-catch of other commercial species and its effect on the sustainability of the stocks;*
4. *The economy (in energetic and value terms) of fishing for protein to convert it into more valuable protein in farming processes;*
5. *The relative contribution of industrial fisheries to habitat degradation as compared to other fishing activities.*

The request of 31 July 2002 was further qualified following correspondence between the EC Directorate General for Fisheries and the General Secretary of ICES:

ICES is requested to:

1. *Evaluate the effect of industrial fisheries at recent levels of fishing mortality on:*
   - *Yield and stock size of relevant and commercially important human-consumption fish,*
   - *Fauna such as marine mammals and seabirds.*

   Effects should be evaluated in terms of predation, food availability and growth limitation, predation on larvae, competition, mortality due to incidental by-catches and any other effects that may be considered significant.

2. *Evaluate the relative benefits (in terms of economic and of ecological efficiency) of fishing "industrial" fish for fish meal and using the product as feed, or of not fishing these species and obtaining higher yields from commercial fisheries.*
3. *Comment on any major structural changes in marine ecosystems, including changes to habitat, that may be caused by fishing for industrial species, and significant consequences for fisheries or the marine environment.*
4. *For the purposes of the above, "industrial fish" means sandeel, blue whiting, Norway pout and sprat. "Human consumption" fish means all other species covered by TACs in the north-east Atlantic.*

The revised request from the European Commission included a request to "*Evaluate the relative benefits (in terms of economic and of ecological efficiency) of fishing 'industrial' fish for fish meal and using the product as feed, or of not fishing these species and obtaining higher yields from commercial fisheries.*" This request was only addressed in part and WGECO (2003) recommended that this work should be completed next year.

Source of information

The 2003 Report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO) (ICES CM 2003/ ACE:05).

Summary

Industrial fisheries remove more fish biomass from European waters than fisheries targeting species for direct human consumption. The blue whiting fishery on the shelf slope is the largest industrial fishery (by landings) and more than half of the industrial landings from the North Sea are sandeels. Given the scale of these industrial fisheries, it is expected that they have impacts on those human consumption species that are also taken in the catch and on food webs. However, the impacts of industrial fishing that have been identified are relatively small in comparison with the effects of directed fisheries for human consumption species.

Most catches in the North Sea industrial fisheries that target specific species also contain a fraction of other industrial species. The North Sea Norway pout fishery has a high fraction of blue whiting. The "sprat" industrial fishery in the North Sea also catches 9% by weight of other species, predominantly herring, sandeel, and a small fraction of gadoid species.

Based on available catch data, the North Sea small-mesh fisheries targeting sandeel, sprat, and Norway pout take very low proportions of haddock and whiting. Based on the assessment data, the average total (human consumption, discards, and industrial by-catch) fishing mortalities for the 0- and 1-group haddock have been 0.02 and 0.12, respectively, over the past ten years. Most of the 0-group fishing mortality is attributed to industrial catch, but for 1-group haddock it accounts for less than

half the fishing mortality. In comparison, natural mortalities for 0- and 1-group haddock are 2.05 and 1.65, respectively. For whiting, the average fishing mortality was 0.11 (and natural mortality 0.95) for the 1-group. Approximately half of the fishing mortality is from the industrial catch. For fish older than one year, the partial fishing mortalities from industrial catch were 0.005 to 0.07 for haddock and 0.002 to 0.11 for whiting over the past ten years. Most haddock and whiting catches were attributable to the Norway pout fishery, and this fishery has declined in size over the past thirty years. The fractions of haddock and whiting in the sandeel fishery were typically lower than those in the Norway pout fishery.

Although catches of human consumption species in the Norway pout fishery are generally small, it is possible that they may be further reduced by increasing the size or changing the position of the "Pout box" area, which is already intended to reduce catches of human consumption species in the industrial fishery. A reassessment of the value of the Pout box requires detailed data on the spatial distribution of catches of other gadoids. ICES encourages the collection of more comprehensive spatially disaggregated catch data in the Norway pout fishery.

There is almost no information on the catches of other species in the blue whiting fisheries of the Northeast Atlantic. Given the scale of this fishery, such information needs to be obtained, as does information on the food-web effects of this fishery.

The effects of variation in the sizes of most industrial stocks on their predators are also poorly known. A better understanding of the interactions between industrial species and their fish predators is needed, and the historically low stock sizes of predators such as North Sea cod may have reduced their overall dependence on the species targeted by industrial fisheries.

Interactions between industrial species and seabird predators are well known in some areas. In one of the most intensively studied seabird-sandeel interactions, certain seabirds, particularly Arctic terns (*Sterna paradisaea*) and black-legged kittiwakes (*Rissa tridactyla*), 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. However, fisheries were unlikely to be the direct cause of the decline in sandeel abundance as the availability of sandeels was primarily driven by recruitment variation. If fishing did affect the recruitment of sandeels, as a result of its impacts on the target stock, then fishing would have had an indirect effect on the seabirds, but the evidence for such an indirect effect was weak.

Recent EC-funded projects such as ELIFONTS and IMPRESS have also examined the effects of variation in sandeel abundance on their predators. The findings from ELIFONTS were considered in advice to the EU Council's decision to prohibit fishing for sandeels in a 20,000 km^2 band of sea down the east coast of Britain in 2000. The black-legged kittiwakes in the region showed an increase in breeding success comparable to pre-fishery levels, but it was difficult to determine whether recovery was due to the cessation of fishing or climatic drivers. Updated analyses showed that kittiwake chick productivity did not respond primarily to sandeel abundance and was not directly and tightly linked to fishing levels.

Based on available analyses, the indirect (food web) effects of the North Sea industrial fisheries appear to be relatively small when viewed at large spatial scales, though small-scale effects may be significant and detailed small-scale analyses are often not available. There are also a number of industrial fisheries for potential "forage" species and for which there is little or no information on either the direct or indirect effects of fishing.

Recommendations and advice

The preceding summary gives a short review of the information requested and the "Scientific background" provides a longer and fully referenced review of the same information. In addition, ICES (2002a) has previously identified a number of industrial stocks for which ecological dependence may need to be considered in management advice. These are typically "forage fish" stocks for which quantitative assessments may or may not be available and which, on the basis of existing observations on the distribution and abundance of associated predators, may be important prey species. There is still relatively scant information on the effects of fisheries targeting these stocks and further analysis of the ecological impacts of these fisheries is required.

ICES has not found information on catch rates of other species in the large blue whiting fishery in the Atlantic nor on the ecological impacts of this fishery. The absence of such data may reflect the low catches of other species or that they are composed primarily of non-commercial species. The food-web effects of fishing for blue whiting are also unknown, but given the large apparent size of the blue whiting stock, ICES would expect it to support a range of predators. However, there is little or no information on the predators that target the different life history stages of blue whiting. These predators need to be identified, as does the significance of changes in blue whiting abundance on their dynamics.

Studies of the local interactions between industrial fisheries and foraging seabirds or predatory fish are still required in areas where industrial fisheries occur and such relationships have not already been considered. Spatially resolved by-catch data for the North Sea Norway pout fishery would assist with improving the protection afforded to the young of other gadoid species and improving the design of the Norway pout box, if necessary.

Scientific background

Industrial fisheries remove a significant proportion of fish biomass from the marine environment. Industrial catches may include the young of species that are targeted in other fisheries or remove individuals that would otherwise be eaten by birds, marine mammals, or predatory fish.

Here, ICES provides a review of the potential effects of industrial fishing on the ecosystem. ICES examines and evaluates the quantity and quality of evidence for the effects, and identifies aspects of the request which require further work.

ICES reviews the effects of industrial fishing on: a) fish (including effects on the target species), b) seabirds, c) marine mammals, and d) habitats/benthos. In considering effects, these are divided between direct effects (mostly direct mortality) and indirect effects (mostly effects on food chains). ICES does not review "downstream" ecosystem effects—the effects of the products of industrial fishing on the environment; these include usage in aquaculture. Reviews of these effects have been published elsewhere (e.g., Iwama, 1991; Findlay *et al.*, 1995; Findlay and Watling, 1997; Pearson and Black, 2001). In addition, ICES does not review the ecosystem effects of industrial fisheries outside ICES waters that provide products for the European market (e.g., anchovy (*Engraulis encrasicholus*) fisheries off South America).

For the purposes of this report, industrial fisheries are defined as fisheries taking fish primarily for reduction to fish meal and oil. The primary target species of industrial fisheries in EU and adjacent Atlantic waters are sandeel (*Ammodytes* sp.), sprat (*Sprattus sprattus*), blue whiting (*Micromesistius poutassou*), and Norway pout (*Trisopterus esmarkii*). It should be noted that blue whiting and sprat are fished in some parts of their range for human consumption purposes, while other species normally targeted for human consumption purposes are occasionally consigned to reduction. These fisheries (or uses of fish) are not considered here.

The request from the EC also included a request to "evaluate the relative benefits (in terms of ecological efficiency) of fishing 'industrial' fish for fish meal and using the product as feed, or of not fishing these species and obtaining higher yields from commercial fisheries". This request was partially addressed by WGECO in 2003, but they recommended that the work should be completed next year and it has not been included in this report.

11.1 Effects on fish

11.1.1 Catches of human consumption species in small-meshed fisheries

Small-meshed fisheries for industrial species can also take by-catches of species targeted for human consumption. Usually these are the juveniles of target species, but some individuals above the minimum landing size may also be caught. These larger individuals may be sorted from the catch intended for reduction and landed for the human consumption market.

This section summarizes data on the catches of target and non-commercial species in industrial fisheries and discusses the implications of these catches for the population biology and management of the target species and for the ecosystem. Since all species caught in the industrial fisheries are usually landed, there are no ecological effects of discarding in this fishery.

11.1.1.1 Sampling of catches

The sampling of Norway pout and sandeel landings is described in detail by Dalskov (2002) and by ICES (1996, 2003a). The sampling methods vary between countries.

Denmark

Landings in the Danish industrial fisheries are sampled to estimate species, and length and age composition. Sampling is conducted by fisheries officers at the time of landing as part of monitoring and enforcement activities. Landings are selected at random and one sample of 10–15 kg is taken per landing, resulting in approximately one sample per 650 tonnes landed (the target sampling regime is one sample per 1000 tonnes landed). A greater proportion of a catch is sampled (up to 120 kg) if the composition of the initial sample gives rise to suspicion that catch composition regulations are being violated. Total landings are estimated per statistical rectangle based on total catch estimates from sales slip and logbook data, together with data on species composition and biological data. The catch composition on a few boats in the Danish sandeel fishery has also been monitored by Scottish observers (Newton *et al.*, 2002).

Norway

In Norway, the sampling system since 1993 has been based on catch samples from three market categories: E02 (sandeel, if mainly sandeel), D13 (blue whiting, if not sandeel and catch was taken west of 0°E), and D12 (Norway pout, if not sandeel and catch was taken east of 0°E). The samples are raised to total landings on the basis of sales slip information on landed categories. Effort is estimated from the total number of trips and an estimate of the average days out at sea per trip.

EU industrial fisheries are regulated by mesh size and catch composition as well as quotas and TACs. For a given codend mesh size, the retained catch must meet certain species composition targets; in most cases a maximum of 5% of any mixture of cod (*Gadus morhua*), haddock (*Melanogrammus aeglefinus*), or saithe (*Pollachius virens*) is permitted (EC Council Regulation 850/98). With the mixed, bulk nature of the catches (up to 70 tonnes per haul) and the small size of the individuals of the young of human consumption species, it would be very difficult to systematically discard these species. There is no evidence in the ICES working group reports of misreporting of the target species landings. There has been some misreporting of herring catches in industrial fisheries recently, but in those cases the whole catch was reported incorrectly. Misreporting that is detected is included in the assessments.

11.1.1.2 Catch trends

Catch trends for target species in the Danish and Norwegian industrial fisheries in the North Sea (sandeel, Norway pout, sprat, herring (*Clupea harengus*), and blue whiting) are shown in Figure 11.1.1.2.1. Tables 11.1.1.2.1 and 11.1.1.2.2 show the total landings of target species and the catch of human consumption species for these fisheries. ICES could not account for the differences between the totals in these tables.

Figures 11.1.1.2.2 and 11.1.1.2.3 show mean catches at age for North Sea haddock and whiting for three time periods: 1987–1991, 1992–1996, and 1997–2001. The catch is divided into three categories: industrial "by-catch", discarded catch, and retained catch in the human consumption fishery. These results indicate that the majority of the industrial catches of human consumption species ("by-catch" in the figures) consist of fish of age 3 and less. There is also variability in the number of 0-group haddock. 0-group whiting are not included in Figure 11.1.1.2.3.

11.1.1.3 Catch data

Industrial catch data on haddock and whiting are used, together with human consumption and discard data, in the assessments of these stocks in the North Sea (haddock in Sub-area IV and Division IIIa and whiting in Sub-area IV and Division VIId; ICES, 2003a). Catch data from all North Sea industrial fisheries are included; there is no division by target species. For North Sea cod (Sub-area IV, Divisions IIIa and VIId), only data on human consumption catches are used in the assessments; no data from the industrial by-catch or from the discards in the human consumption fishery are used. Some monitoring programmes have quantified cod by-catches in these industrial fisheries, and the by-catches are reported to be low. North Sea haddock and whiting are the only species included in the industrial catch for which there are data on both the industrial catch and catches for human consumption (discarded and retained).

11.1.1.4 Relative impact of the industrial and human consumption fisheries

Partial fishing mortalities for haddock ages 2 to 6 and whiting ages 2 to 6 are described in ICES (2003a). These results indicate a partial fishing mortality due to industrial fishing of 0.005 to 0.07 for haddock and 0.002 to 0.11 for whiting over the past ten years. This compares with much higher fishing mortalities from the human consumption fisheries (both retained and discarded catches) of F = approximately 1.0 for haddock and F = 0.43–0.82 for whiting over the same period. Fishing mortality for these age groups due to industrial fishing is much smaller than mortality due to the human consumption fishery.

The average total (human consumption, discards, and industrial catch) fishing mortality for the 0- and 1-group of haddock has been 0.02 and 0.12, respectively, over the past ten years. Most of the 0-group fishing mortality is from the industrial catch. For the 1-group, less than half of the mortality is due to industrial catch. For comparison, the assessment uses a natural mortality of haddock of 2.05 and 1.65 for the 0- and 1-groups, respectively. For whiting, the average F was 0.11 (and M 0.95) for the 1-group. Approximately half of the F is from the industrial catch. There is no estimate available for the 0-group.

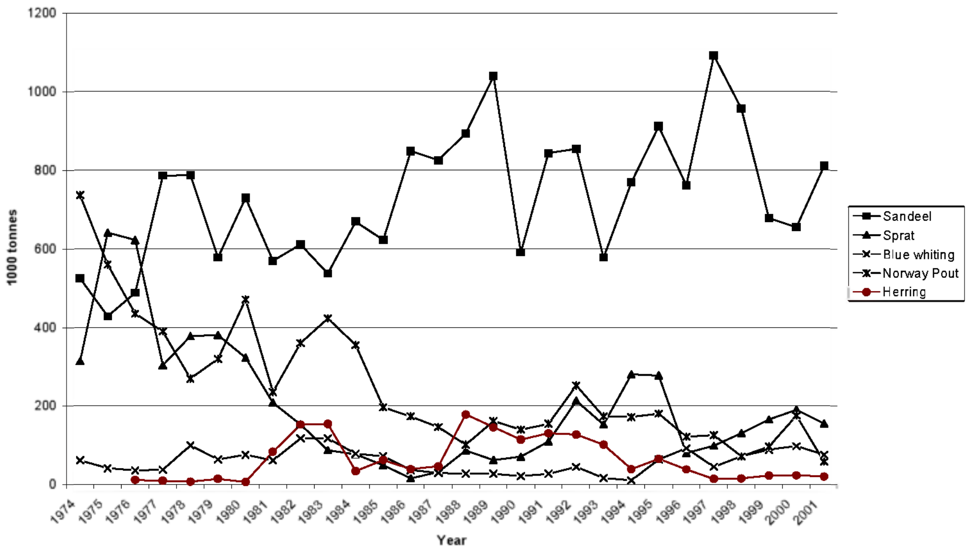


Figure 11.1.1.2.1. Total Danish and Norwegian catches of the main industrial fishery target species in the North Sea, 1974–2001 (ICES, 2003a).

Table 11.1.1.2.1. Total catch (1000s of tonnes) of the main target species (sandeel, sprat, blue whiting, and Norway pout) and other species in Danish and Norwegian North Sea industrial fisheries 1974–2001 (ICES, 2003a). For the species composition of "others", see Table 11.1.1.2.2.

Year	Target Species	Other species					
		Herring	Haddock	Whiting	Saithe	Cod	"Others"
1974	1637		48	130	42		
1975	1671		41	86	38		
1976	1581	12	48	150	67		
1977	1518	10	35	106	6		
1978	1535	8	11	55	3		
1979	1342	15	16	59	2		
1980	1599	7	22	46			
1981	1076	84	17	67	1		
1982	1242	153	19	33	5		
1983	1166	155	13	24	1		
1984	1180	35	10	19	6		
1985	942	63	6	15	8	0.5	65
1986	1075	40	3	18	1	0.7	32
1987	1035	47	4	16	4	1.1	72
1988	1110	179	4	49	1	1.4	43
1989	1292	146	2	36	1	3.0	56
1990	824	115	3	50	8	2.9	37
1991	1136	131	5	38	1	0.6	37
1992	1365	128	11	27		1.0	29
1993	922	102	11	20	1	1.1	26
1994	1233	40	5	10		0.9	18
1995	1434	66	8	27	1	1.0	14
1996	1057	39	5	5	0	0.4	14
1997	1362	15	7	7	3	1.7	20
1998	1231	16	5	3	3	1.3	27
1999	1030	23	4	5	2	0.5	40
2000	1120	24	8	8	6	0.4	21
2001	1101	21	6	7	3	0.2	12

Table 11.1.1.2.2. Composition of catch labelled as "others" in Table 11.1.1.2.1 (data from ICES, 2003a).

Common name	Latin name	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001
Trigla sp.	*Trigla* sp.	0	888	45342	5394	9391	2598	5622	4209	1593	1139	2091	897	2618	1015	2566	1343	2293
Dab	*Limanda limanda*	187	3209	4632	3781	7743	4706	5578	3986	4871	528	1028	1065	2662	6620	4317	441	1441
Smelt sp.	*Argentina* spp.	8714	5210	3033	1918	778	2801	3434	2024	2874	2209	292	3101	2604	5205	3580	333	397
Long rough dab	*Hippoglossoides platessoides*	59	718	1173	946	2160	1673	1024	1694	1428	529	617	339	1411	2229	1272	493	431
Hake	*Merluccius merluccius*	349	165	261	242	290	429	28	359	109	10	-	3625	2364	33	211	231	167
Poor cod	*Trisopterus minutus*	0	68	0	5	48	121	79	111	36	0	9	30	181	261	922	518	0
Ling	*Molva molva*	51	1	40	39	37	13	65	10	28	0		0	31	31	125	19	49
Witch	*Glyptocephalus cynoglossus*	236	132	341	44	255	251	1439	195	246	40	-	97	394	860	437	154	246
Silvery pout	*Gadiculus argenteus*	1210	729	3043	2494	741	476	801	0	0	0	-	7	248	248	387	532	942
Others [1,2]		31715	3853	3604	3670	3528	3154	4444	4553	4106	5141	5158	50	749	5405	17931	8927	301
Total		42521	14973	61469	18533	24971	16222	22514	17141	15291	9596	9195	9211	13262	21907	31748	12991	6267

[1]Danish cod (*Gadus morhua*) and mackerel (*Scomber scombrus*) included in "others" in 1985.
[2]Danish catches of witch (*Glyptocephalus cynoglossus*) included in "others" for 1985 and 1989–1992.

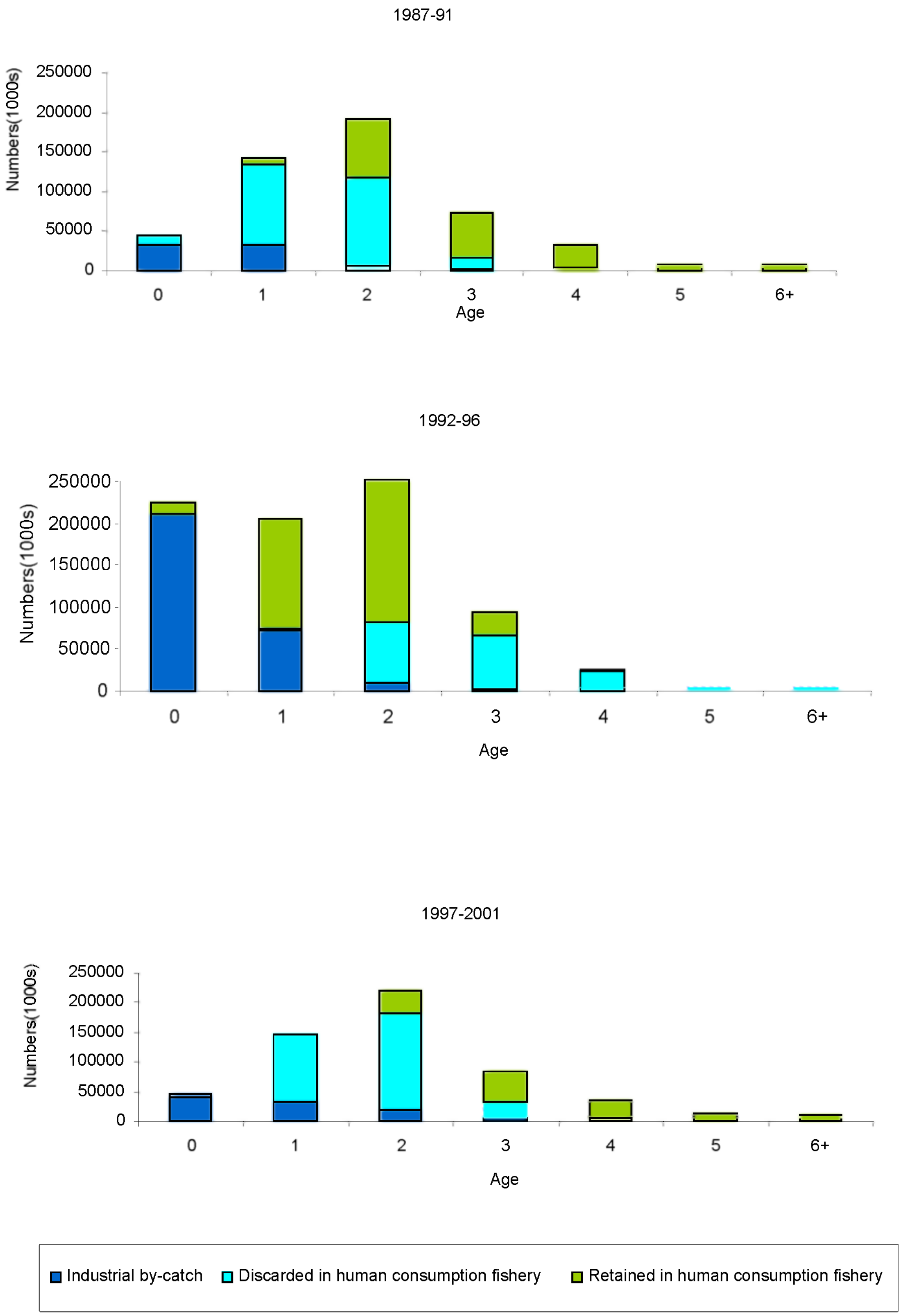


Figure 11.1.1.2.2. Mean numbers of North Sea haddock (Sub-area IV and Division IIIa) by age captured per year by time period in three categories: industrial "by-catch", discarded, and retained in the human consumption fishery (data from ICES, 2003a).

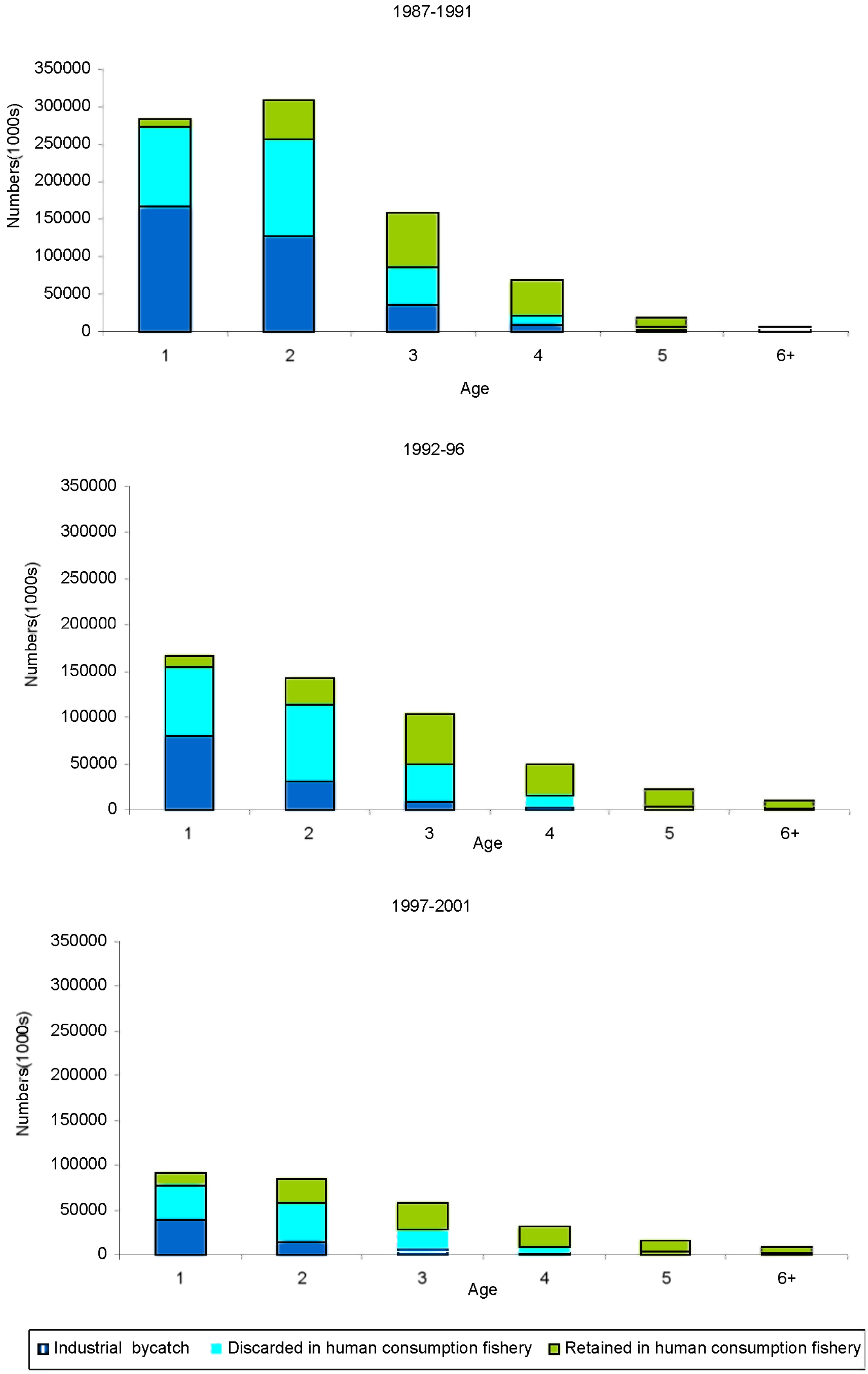


Figure 11.1.1.2.3. Mean numbers of whiting (Sub-area IV and Division VIId) by age captured per year by time period in three categories: industrial "by-catch", discarded, and retained in the human consumption fishery (data from ICES, 2003a).

Thus, although the overall partial fishing mortality due to industrial fishing was low on these stocks, there was some variation. ICES (2003a) also indicated that partial fishing mortality of the industrial fisheries on these stocks had been higher in the past (before 1990). Further investigations of these data could elucidate past spatial and temporal patterns.

These results accord with those of ICES (2003a). Of the species assessed in ICES (2003a), only whiting was regarded as being subject to significant catch in the industrial fisheries. This catch has been reduced in recent years. Fishing mortality on haddock due to the industrial fisheries has been very low throughout the 1990s (ICES, 2003a).

11.1.2 Fisheries for sandeel

The majority of sandeel fishing takes place in the central and eastern North Sea (Figure 11.1.2.1). The fishery is very patchy, some areas being preferred in some years; this preference is driven by catch rates and market conditions. Nearly the entire fishery occurs in the second and third quarters of the year.

Sandeel landings from 1974–1985 fluctuated between 428,000 tonnes and 787,000 tonnes, with a mean of 611,000 tonnes. During 1986–2000 the landings increased to a higher level from 591,000–1,091,000 tonnes, with a mean of 819,000 tonnes. The combined Danish and Norwegian landing in 1997 was more than 1 million tonnes, and was the highest ever recorded (Figure 11.1.1.2.1). The effort of the sandeel fleet gradually decreased from 1989 to 1994, increased from 1994 to 1998, and decreased again from 1998–2000. In 2001, there was a small increase in effort compared to 2000 (ICES, 2003a).

11.1.2.1 Direct effects

Effects on sandeel stocks

The direct effects of fishing for sandeels are assessed and advised upon by ACFM (ICES, 2002b). For the North Sea and Shetland, this assessment is made every year, for Division VIa less frequently, and never for Division IIIa. In 2002, the North Sea stocks were "within safe biological limits" (SSB above B_{pa} of 600,000 t), but no fishing mortality reference points have been set. Safe biological limits have not been defined for the Shetland "stock", but fishing mortality was believed to be well below natural mortality.

There is evidence that there is considerable geographic structuring within the sandeels of the North Sea. It appears that lesser sandeels (*Ammodytes marinus*) do not inter-mix to any significant degree over distances >200 km (Procter *et al.*, 1998; Wright *et al.*, 1998), though other sandeel species (e.g., *Gymnammodytes semisquamatus*) are less structured (Fehervari and Naevdal, 1995). Too little is known about the stocks in the North Sea to comment on possible local depletion or any possible ecosystem effects.

Catch of other species

The catch of other species in recent sandeel fisheries is very small in comparison with the landings of sandeels from the North Sea (Table 11.1.2.1.1). The catch of commercial species in the sandeel fishery mainly consists of herring and saithe in the area north of 57°N and herring and whiting in the area south of 57°N. The average catch in weight of saithe, whiting, haddock, herring, and other species was, in the period 1997–2001, 2.8% and 1.6% of the catches of sandeels in the area north and south of 57°N, respectively.

11.1.2.2 Indirect effects

Reduction in predation by sandeels

No evaluation has been made on the consequences of fishing on sandeels for their main prey, which comprises phytoplankton and zooplankton including juvenile fish and eggs (Macer, 1966).

Reduction in prey for fish predators of sandeels

The annual consumption of sandeels by commercial fish, seabirds, and other fish/marine mammals in the North Sea has been estimated as 1.9 million tonnes, 0.2 million tonnes, and 0.3 million tonnes, respectively (ICES, 1997a). Cod, haddock, whiting, mackerel, saithe, grey gurnard (*Eutrigla gurnardus*), and starry ray (*Raja radiata*) are the most important predators of sandeels (Pope and Macer, 1996; ICES, 1997b). Of the commercially exploited fish species in the North Sea, sandeels comprise 40–60% of the fish biomass consumed and 15–25% of the total biomass (ICES, 1997a). Changes in the size of the sandeel stocks in the North Sea clearly have potential implications for predators.

The feeding behaviour of sandeel predators was recorded in the ELIFONTS study. This study extended over two years (1997 and 1998) in the Wee Bankie area off the Firth of Forth where a sandeel fishery was present (Harwood *et al.*, 2000). The total biomass of sandeels in 1998 was 15% less than in 1997. In both years, the sandeel population was dominated by a very strong (1996) year class. Total removals from the sandeel population were similar in both years (69,000 tonnes in 1997 and 65,000 tonnes in 1998). Fish were the most important natural predator in both years. The fishery was responsible for 68% of all removals in 1998, compared to 34% in 1997.

In the Wee Bankie area, there was also a large change in the diet of the predatory fish when sandeel abundance changed (prey switching), with fewer fish of all species being consumed in 1998, although the statistical significance of this could not be assessed. The fish consumed by haddock were almost entirely sandeels in

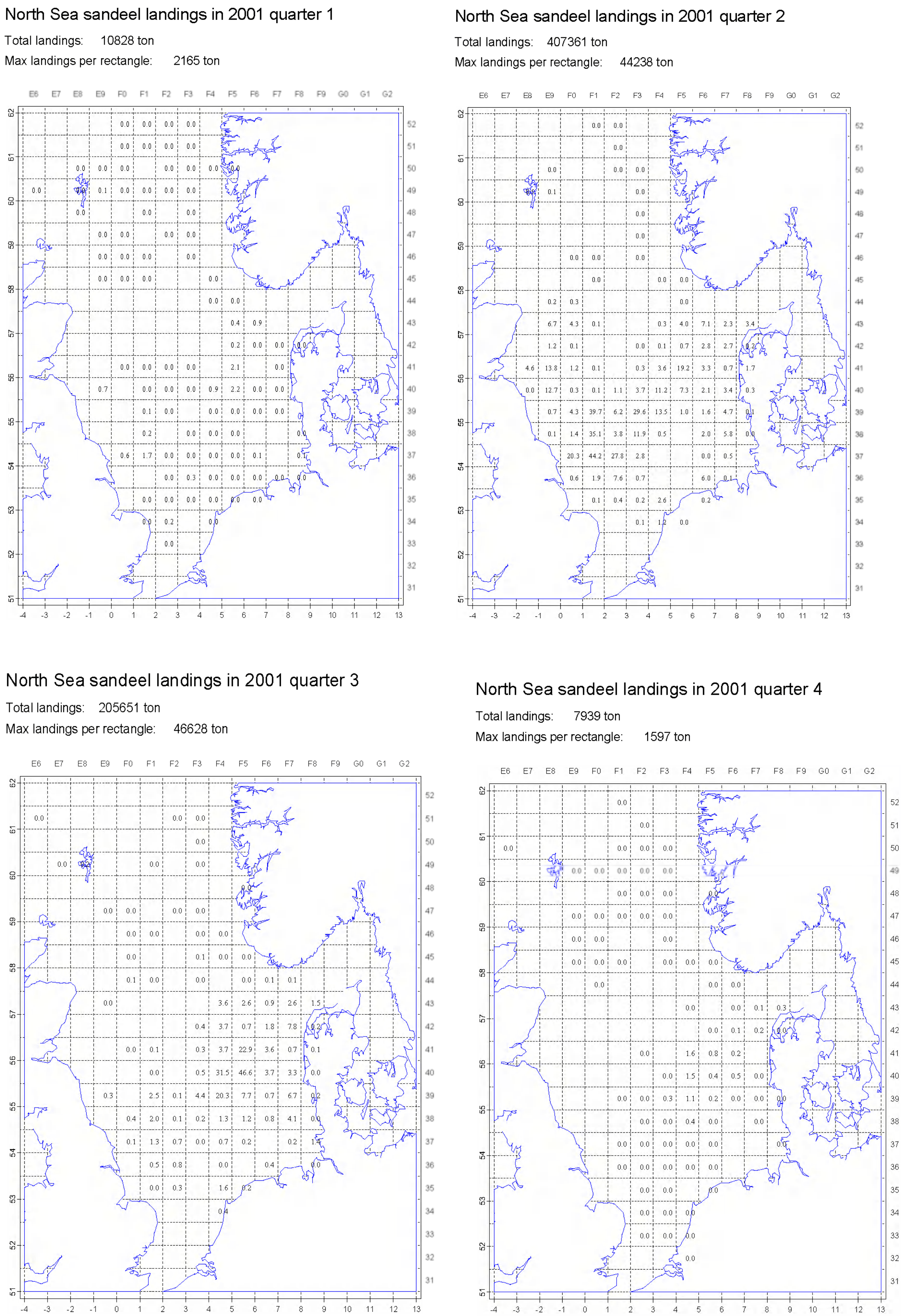


Figure 11.1.2.1. Quarterly catches (Denmark, Norway, and Scotland) of sandeel by ICES rectangle in 2001 ('000 tonnes) (ICES, 2003a).

both years, but they consumed more invertebrates in 1998. The proportion of sandeels in the diet of whiting was lower in 1998, when clupeids and gadoids replaced them. The proportion of fish that were feeding when caught, the quantity of food consumed per day, body condition factors, and growth rates were all lower in 1998 than in 1997, perhaps evidence that whiting and haddock experienced a food shortage in 1998.

It is important to understand the interaction between the foraging strategy of each predator and the behaviour and life cycle of the sandeel in order to understand the effects of changes in sandeel abundance. The predators in the ELIFONTS study showed measurable responses to the change in age structure of the sandeel population between 1997 and 1998. The main factors determining sandeel availability for most of the predators were recruitment and the effects of environmental variation on behaviour. For predatory fish, the precise time at which 0-group sandeels become available, and their average size, also appeared to be important factors. While neither size nor timing are apparently affected immediately by the activities of the commercial fishery, if sandeel recruitment to the local stock is heavily dependent on the local spawning stock, then fishing of the 1+-group fish could affect the abundance of 0+-group fish in subsequent years.

11.1.3 Fisheries for sprat

The majority of the sprat fishery occurs in the central southern North Sea in winter (third, fourth, and first quarters), with an eastwards movement in fishery distribution in the third quarter (Figures 11.1.3.1 and 11.1.3.2).

11.1.3.1 Direct effects

Effects on sprat stocks

The direct effects of fishing for sprat are assessed and advised upon by ACFM (ICES, 2002b). Fishing occurs in ICES Divisions IIIa, VIId and e, Sub-area IV, and Baltic Sub-divisions 22–32. The North Sea stock is believed to be in good condition; fishing mortality for the Baltic stocks was above F_{pa} in 2002 (ICES, 2003c).

Catch of other species

Compared with demersal fisheries, sprat catches are relatively “clean” of other species with the (intermittent) exception of herring. The North Sea sprat fishery in the area north of 57°N has in recent years been small compared to that in the area south of 57°N. In the area south of 57°N, the average catch of other species was about 9% in weight of the landings of sprat in the period 1997–2001 (Table 11.1.3.1.1). The catch of other species in the sprat fishery consists mainly of herring, followed by sandeel and others.

Table 11.1.2.1.1. Average catches (’000 t) in the time period 1997–2001 in the Danish and Norwegian sandeel fishery by species. Information on the composition of “others” was not available (ICES, 2000, 2001b, 2002b, 2003a).

Sandeel	Blue whiting	Haddock	Herring	Norway Pout	Saithe	Sprat	Whiting	Others
800	1	3	5	2	+	8	2	7

Table 11.1.3.1.1. Average catches (’000 t) in the time period 1997–2001 in the Danish and Norwegian sprat fishery by species (ICES, 2000, 2001a, 2002b, 2003a).

Sprat	Blue whiting	Haddock	Herring	Norway pout	Saithe	Sandeel	Whiting	Others
144	+	+	10	+	+	7	1	2

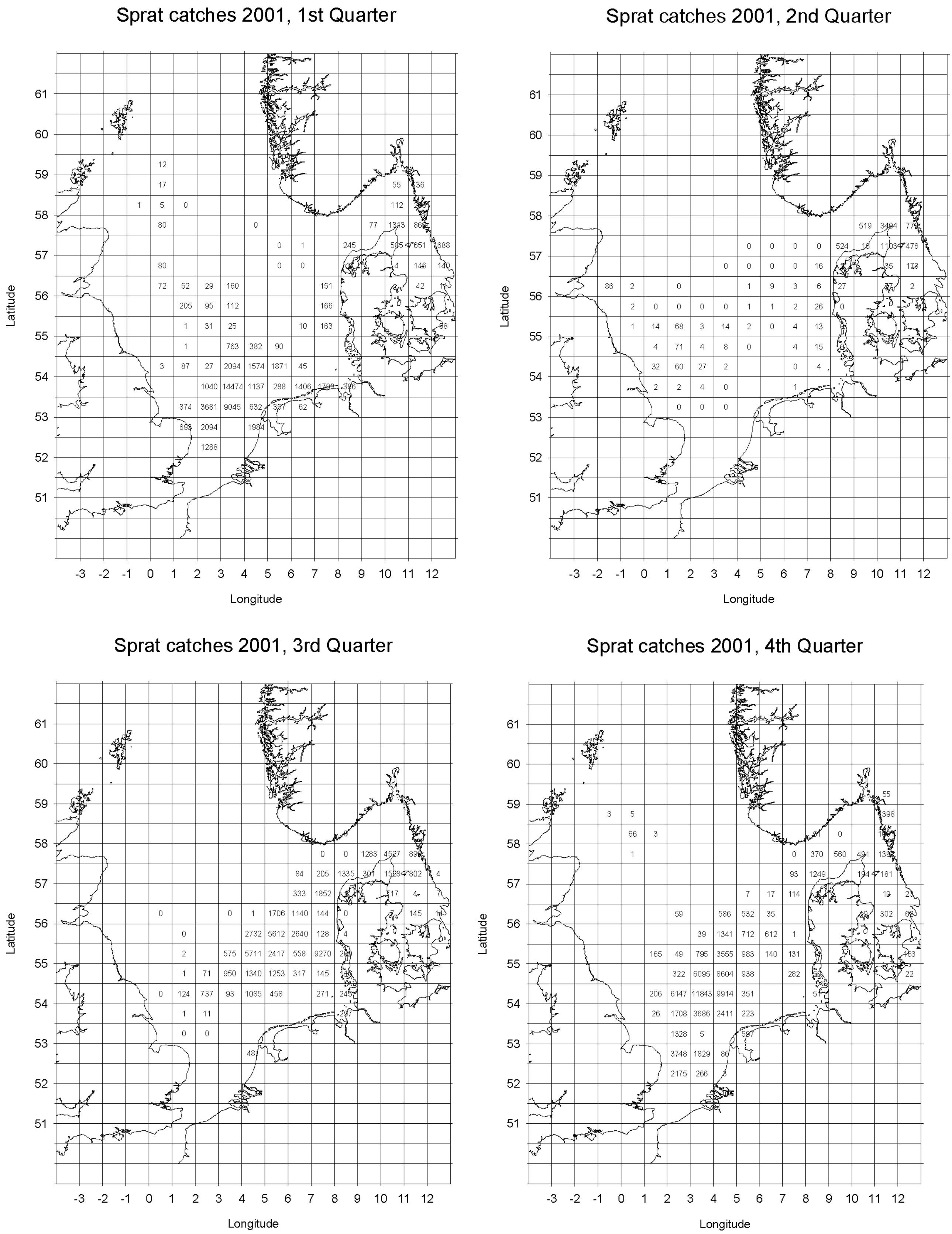


Figure 11.1.3.1. Quarterly catches (working group estimates) of sprat by ICES rectangle in 2001 (tonnes) (ICES, 2001b).

11.1.3.2 Indirect effects

Reduction in predation by sprat

No evaluation has been made on the consequences of fishing on sprat for their main prey, which comprises planktonic crustaceans (e.g., *Pseudocalanus* spp., *Temora* spp., and *Acartia* spp.).

Reduction in prey for fish predators of sprat

Sprat are consumed by cod, haddock, whiting, mackerel, saithe, grey gurnard, and starry ray (Pope and Macer, 1996; ICES, 1997a). No evaluation has been made of the possible effects of a reduction of sprat stock size on these predators.

During the 1970s collapse of the herring stocks, sprat increased, leading to suggestions that they had partially replaced herring in the North Sea (Hall, 1999). Corten (1986, 1990) rejects this theory as geographical patterns in the decline and recovery of both stocks occurred on smaller scales where there was no evidence of one species temporarily occupying the geographical niche of the other. Other reviews conclude that the evidence supporting species replacement is not conclusive (e.g., Daan, 1980; Duplisea *et al*., 1997).

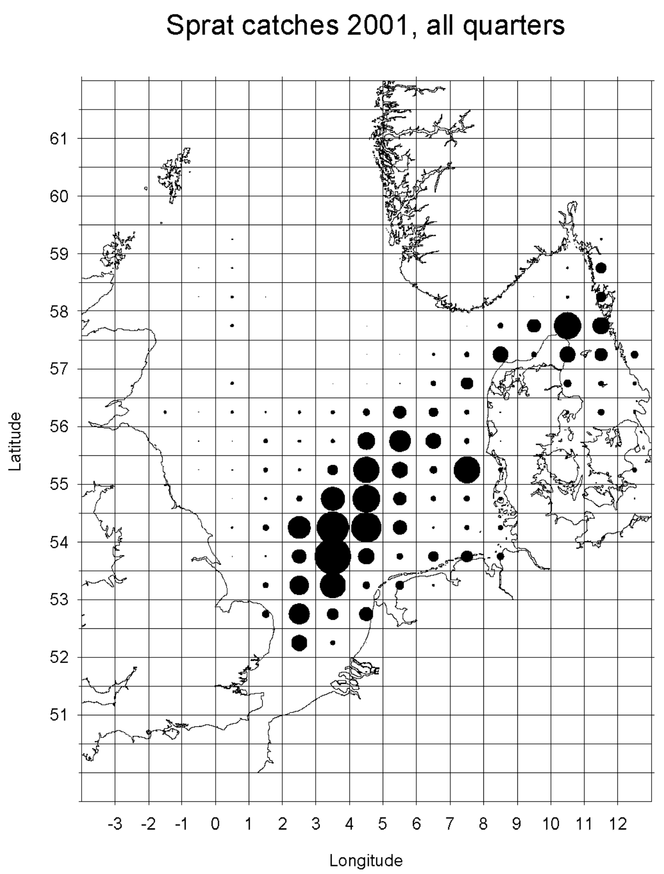


Figure 11.1.3.2. Spatial distribution of 2001 catches of sprat (working group estimates) by ICES rectangle (ICES, 2001b). Circle area is proportional to catch in tonnes.

11.1.4 Fisheries for blue whiting

In contrast to the other fisheries, the main blue whiting fishery occurs in deeper shelf slope waters to the west of Scotland, Ireland, around the Faroes, and towards Iceland. In the North Sea, nearly the entire fishery occurs in the Norwegian Trench to the south and west of Norway (Figures 11.1.4.1 and 11.1.4.2). The majority of this fishery is for adults. Substantial tonnages of smaller fish are also landed with Norway pout (see Section 11.1.5).

11.1.4.1 Direct effects

The main oceanic distribution is considered to represent a single stock and is managed accordingly (ICES, 2002b). The blue whiting fishery is directed on spawning fish in ICES Divisions Vb, VIa,b, VIIb,c, VIIg–k, and Sub-area XII (ICES, 2000). The fishery is mainly carried out by Norway, with Russia, the UK, and the Netherlands also catching significant amounts. About 40% of this fishery takes place in ICES Division VIa along the continental slope to the west of Hebrides. Norway takes more than 70% of the total catch.

The spawning stock biomass (SSB) for 2001 at the spawning time (April) was inside safe biological limits, while the SSB for 2002 was expected to be below B_{pa}. Fishing mortality has increased rapidly in recent years, and is estimated at 0.82 in 2001 (ICES, 2002b). Total landings in 2001 were almost 1.8 million tonnes.

Some data were available on the catch of other species in the industrial blue whiting fishery in the North Sea (Table 11.1.4.1.1). There appears to be no published information on catches of other species in the larger Northeast Atlantic fishery, though the small quantity of other species caught in the North Sea component of the fishery suggests that catches may be relatively "clean".

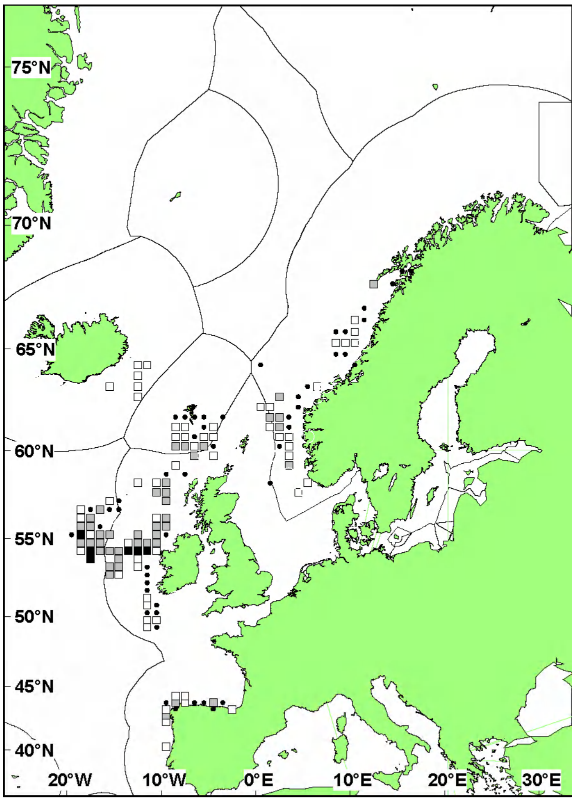
11.1.4.2 Indirect effects

Reduction in predation by blue whiting

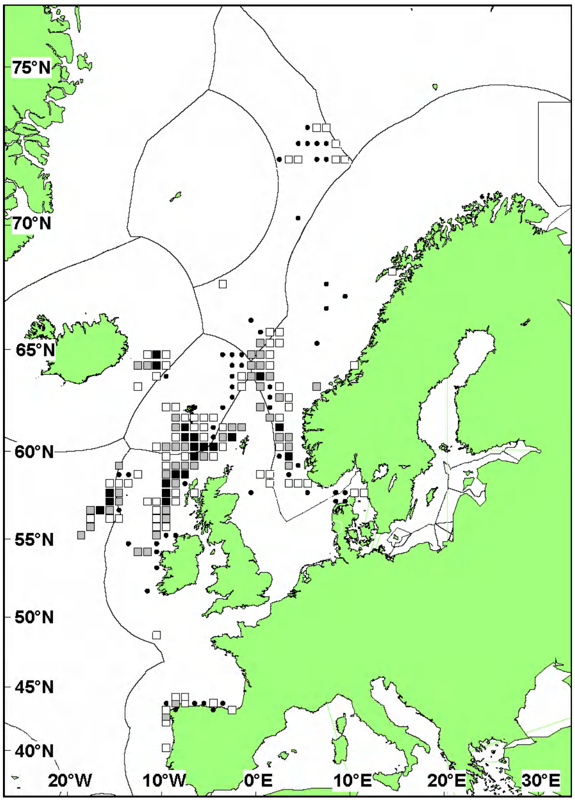
No evaluation has been made on the consequences of fishing on blue whiting for their main prey, which comprises mostly zooplankton, micronekton and, particularly in larger individuals, small finfish. It has been speculated that blue whiting is an important competitor with (and/or predator on) herring in the Norwegian Sea (Zilanov, 1968).

Reduction in prey for fish predators of blue whiting

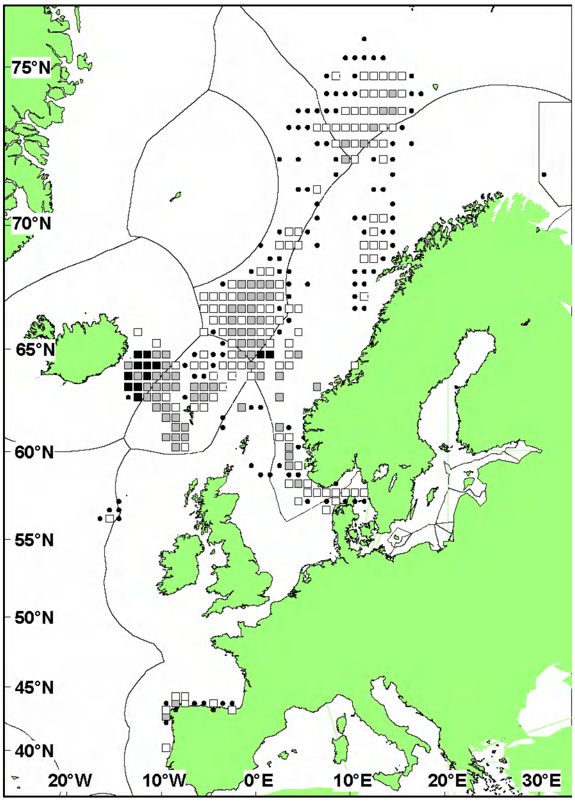
Blue whiting are consumed by a range of piscivores, and the species is an important item in the diet of some fish, e.g., cod (Du Buit, 1995) and Greenland halibut (*Reinhardtius hippoglossoides*) (Bjelland *et al*., 2000).



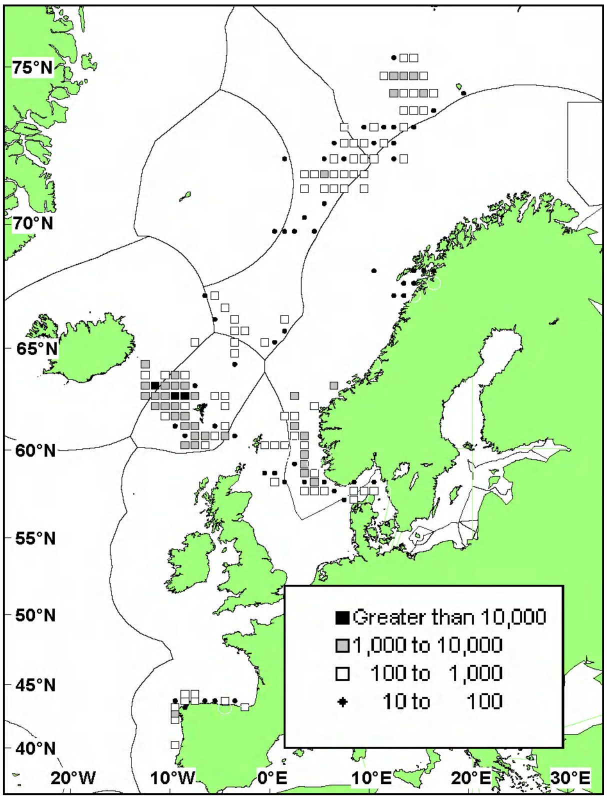
Q1



Q2



Q3



Q4

Figure 11.1.4.1. Total catches of blue whiting in 2001 by quarter and ICES rectangle. Grading of the symbols: small dots 10–100 t, white squares 100–1000 t, grey squares 1000–10,000 t, and black squares > 10,000 t. The 200 n.m. fishing zones are also shown (ICES, 2003b).

Further known fish predators of blue whiting are black scabbardfish (*Aphanopus carbo*), blue hake (*Antimora rostrata*), blue ling (*Molva dipterygia*), European conger (*Conger conger*), European hake (*Merluccius merluccius*), grey gurnard, John dory (*Zeus faber*), ling (*Molva molva*), Arctic eelpout (*Lycodes frigidus*), spurdog (*Squalus acanthias*), saithe, swordfish (*Xiphias gladius*), and tusk (*Brosme brosme*). Silva *et al.* (1997), in a study in Portuguese waters, also recorded lesser spotted dogfish (*Scyliorhinus canicula*) and monkfish (*Lophius* spp.) as blue whiting predators.

Silva *et al.* (1997) identified blue whiting as a component of hake diet in Portuguese waters. In Portuguese waters, hake prey on blue whiting, mackerel, chub mackerel (*Scomber japonicus*), anchovy, and sardine (*Sardina pilchardus*). There is some correspondence between the seasonal and spatial variation in abundance of prey species in the ecosystem and the proportion of these prey in the diet (Silva *et al.*, 1997). Blue whiting is always one of the most important species, both in terms of availability in the environment and as a proportion of the diet. Hake, however, is an opportunistic feeder (Du Buit, 1996; Hill *et al.*, 1999) and when one prey species becomes less abundant, it will switch to another. Hill and Borges (2000) found that hake can switch from feeding on blue whiting to feeding on sardine off the northern Portuguese coast.

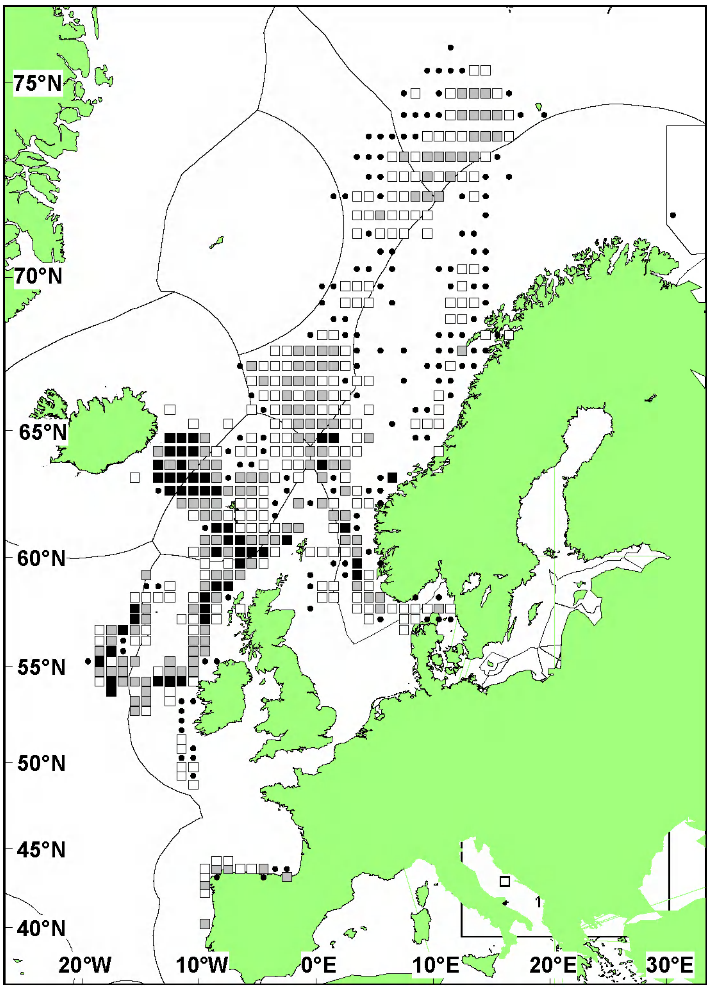


Figure 11.1.4.2. Total catches of blue whiting in 2001 by ICES rectangle. Grading of the symbols: small dots 10–100 t, white squares 100–1000 t, grey squares 1000–10,000 t, and black squares > 10,000 t. The 200 n.m. fishing zones are also shown (ICES, 2003b).

Table 11.1.4.1.1. Average catches ('000 t) in the time period 1997–2001 in the Danish and Norwegian North Sea small-mesh blue whiting fishery in the North Sea by species (ICES, 2000, 2001a, 2002b, 2003a).

Blue whiting	Haddock	Herring	Norway pout	Saithe	Sandeel	Sprat	Whiting	Others
6	0	0	+	0	0	0	0	1

Blue whiting have not exceeded 5% by wet weight of the prey of Northeast Arctic cod in the last 18 years (ICES, 2003b), but this global figure may hide some geographical variation. In shallower seas, such as the Barents Sea, blue whiting may live close to the seabed, perhaps making them more available to demersal species (Heino and Godø, 2002).

Blue whiting and herring are considered to be abundant in the deeper waters of the continental slope of the Norwegian Sea. Greenland halibut seem to preferentially take blue whiting as opposed to other slope and mesopelagic fishes (Bjelland and Bergstad, 1998). These authors further suggest that blue whiting corpses are important to deep-living scavengers, e.g., Arctic eelpout.

The high abundance of blue whiting suggests that it might be an important species in the pelagic ecosystem of the Atlantic. As a mesopelagic species, it occupies deeper water than other commercially important pelagic species such as the sandeel and may play an important trophic role in pelagic ecosystems.

11.1.5 Fisheries for Norway pout

Most Norway pout fishing occurs during the winter (quarters 1 and 4) in the northern North Sea and the Skagerrak (Figure 11.1.5.1) and is undertaken by Danish and Norwegian fleets. This is a mixed fishery that includes substantial quantities of smaller blue whiting that are also landed for industrial purposes. In order to reduce the scale of by-catch of young, commercially important species, fishing for Norway pout is not allowed (EC Regulation No. 3094/86) within the "Pout Box" in the northwestern North Sea (Figure 11.1.5.2).

11.1.5.1 Direct effects

Effects on Norway pout stocks

The direct effects of fishing for Norway pout are assessed and advised upon by ACFM (ICES, 2002b). Stocks within Division IIIa and the North Sea are within safe biological limits. Fishing mortality has generally been lower than the natural mortality and fishing mortality has generally decreased in recent years to well below the long-term average. There is no information on the stocks in Division VIa.

Norway pout catches showed a downward trend in the period 1974–1988. Between 1989 and 1997, catches fluctuated around a level of 150,000 tonnes, until 1998–1999, when the lowest catches of 100,000 tonnes were recorded. In 2000, landings of Norway pout increased to around 200,000 tonnes due to the strong year class in 1999. However, landings in 2001 decreased to around 65,000 tonnes, which are the lowest yearly landings since 1966.

Effort in the Norway pout fleet has decreased gradually from 1993–2001, apart from a small increase in 1998–2000. In 2001, the effort on Norway pout reached a historic low (ICES, 2003a).

Catch of other species

One of the direct impacts of Norway pout industrial fishing is the by-catch of juveniles of saithe, whiting, haddock, and herring that are taken in the catches (Table 11.1.5.1.1).

11.1.5.2 Indirect effects

Reduction in predation by Norway pout

No evaluation has been made on the consequences of fishing on Norway pout for their main prey, which comprises planktonic crustaceans (copepods, euphausiids, shrimps, amphipods), small fish, various eggs and larvae (Sparholt *et al.*, 2002).

Reduction in prey for fish predators of Norway pout

Cod, whiting, and saithe are by far the main predators on Norway pout of age 1 and older (Bergstad, 1990; Sparholt *et al.*, 2002). In the North Sea, the fishing mortality on Norway pout was estimated to be 40–60% of the natural mortality of 1.6 (ICES, 1997a).

11.2 Effects on seabirds

Several studies have shown that variations in prey availability can have profound effects on the population parameters of seabirds, including breeding success and over-wintering survival. Sandeels are an important, lipid-rich food source for many seabirds, especially during the breeding season, and during stock collapses, breeding success has fallen (Danchin, 1992; Phillips *et al.*, 1996). Similarly, a collapse in the North Sea sprat stock was considered to be partly responsible for a mass mortality of auks in the spring of 1983 (Blake, 1984). Harris and Bailey (1992) found that the winter survival rates of auks were correlated with the abundance of sprat. However, it has not been possible to isolate the contribution of fisheries to these collapses, and in most cases recruitment failure appears to have been the most important factor. Short-lived pelagic fish species are particularly at risk from recruitment failures as the stock biomass is composed typically of one or two year classes. Failures have occurred in the absence of any fishery (e.g., Southward *et al.*, 1988), but fishing may increase the frequency, duration or extent of a stock collapse and depress locally important fish stocks.

11.2.1 Effects of fisheries for sandeels on seabirds

11.2.1.1 Direct effects

There has been no systematic study of the by-catch of seabirds in fisheries for sandeel. However, common

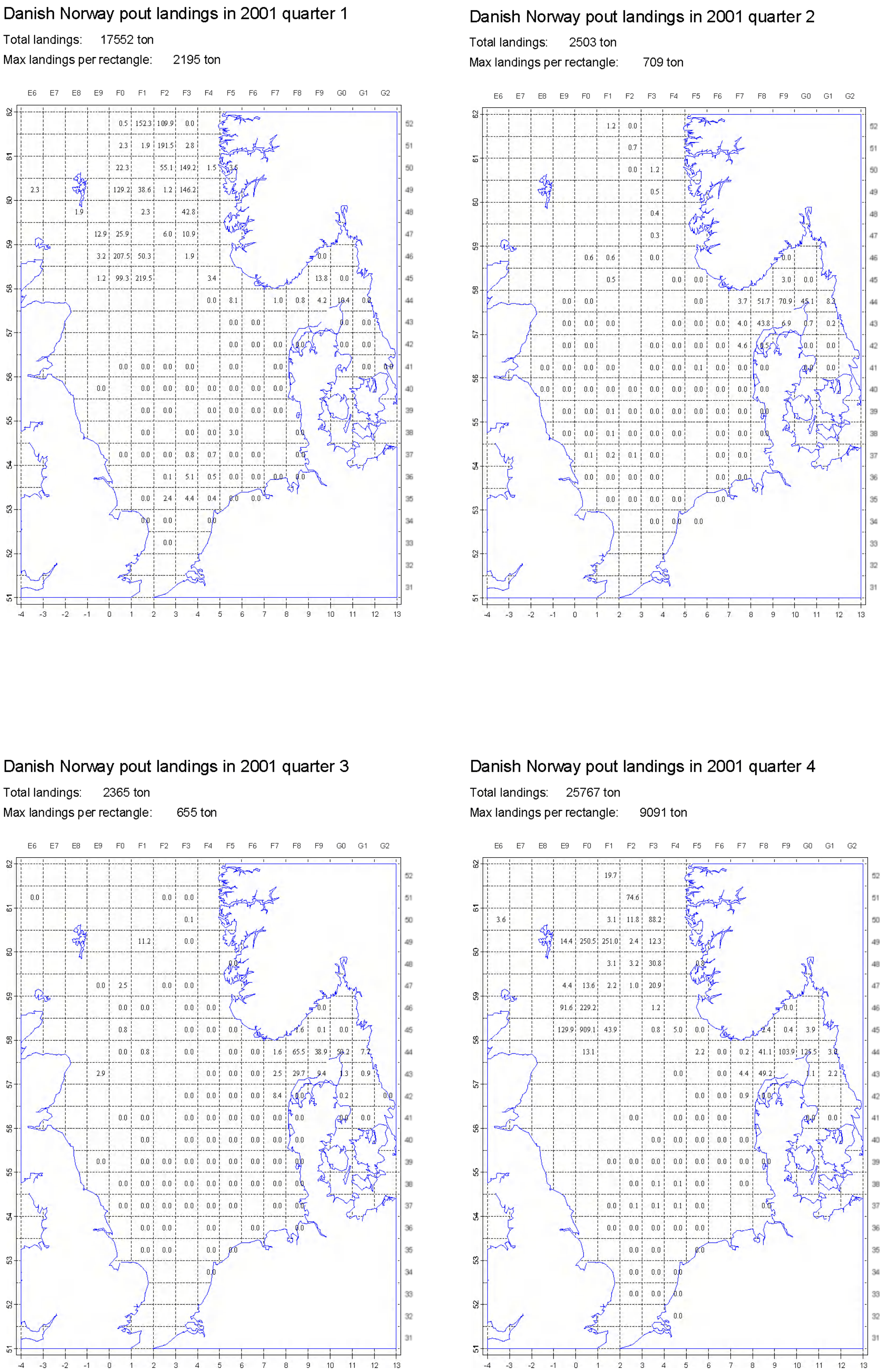


Figure 11.1.5.1. Plot of landings (tens of tonnes per ICES rectangle) in 2001 from the Danish Norway pout fishery by ICES statistical rectangle and quarter of the year (ICES, 2003a).

Table 11.1.5.1.1. Average catches ('000 t) in the time period 1997–2001 in the Danish and Norwegian Norway pout fishery by species (ICES, 2000, 2001a, 2002b, 2003a).

Norway pout	Blue whiting	Haddock	Herring	Saithe	Sandeel	Sprat	Whiting	Others
107	50	4	5	4	+	+	3	7

guillemots (*Uria aalge*) have been recorded in sandeel trawl nets in the North Sea, where fishing was occurring in the feeding area of a colony (Tasker *et al.*, 2000). Five hauls were recorded and a total of twenty-two birds were caught (others may have been missed due to the method of bulk handling of the catch).

11.2.1.2 Indirect effects

Sandeels are a preferred prey species of a wide range of seabirds (Tasker and Furness, 1996). The scale of the sandeel fishery has led to concerns about the impact on seabirds through competition for the same resource. It is important though to consider the spatial and temporal overlap of usage of sandeels by the fishery and by seabirds.

During the breeding season, seabirds in the North Sea feed predominantly close to their colonies in the northwest. If food supply is short immediately beside the colony, some seabirds can feed at greater distance, but at a cost in terms of reproductive output, and possibly in adult survival (Monaghan *et al.*, 1992). Following the breeding season, the birds are not so constrained in terms of colony location—some migrate out of the North Sea, while others move to other parts such as the Skagerrak and Kattegat, or further offshore in the central North Sea. The ICES Study Group on Effects of Sandeel Fishing (ICES, 1999) computed an index of sensitivity of breeding success of different seabird species to reduced abundance of sandeels in the vicinity of colonies. This index was multiplied by seabird density and plotted to illustrate the sensitivity of seabirds to changes in sandeel abundance in May and June (the main part of the seabird breeding season) (Figure 11.2.1.2.1).

The fisheries exploit the North Sea differently. With the exception of a comparatively small (relative to the whole North Sea fishery) catch near Shetland, sandeels are taken mostly offshore in the central North Sea with, in some past years, exploitation of banks off the east coast of Scotland. Broadly, the fishery is carried out in areas outside those used most by breeding seabirds.

Jensen *et al.* (1994) studied the overlaps between the distributions of fish (sandeel and sprat), three auk species (common guillemot, razorbill (*Alca torda*), Atlantic puffin (*Fratercula arctica*)), and the fisheries for 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, at a large scale, that some fisheries and some birds were exploiting the same fish species in the same place, but at different times of the year. This study also illustrated 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.

Wright and Begg (1997) examined the overlap among sandeels, their fisheries, and common guillemots in the northwestern North Sea at a much finer scale. They found that there were some areas important for both common guillemots and fisheries, but that most guillemots were found in areas where sandeels were not exploited.

However, as mentioned above, the sandeel fishery in the North Sea varies in location from year to year, probably

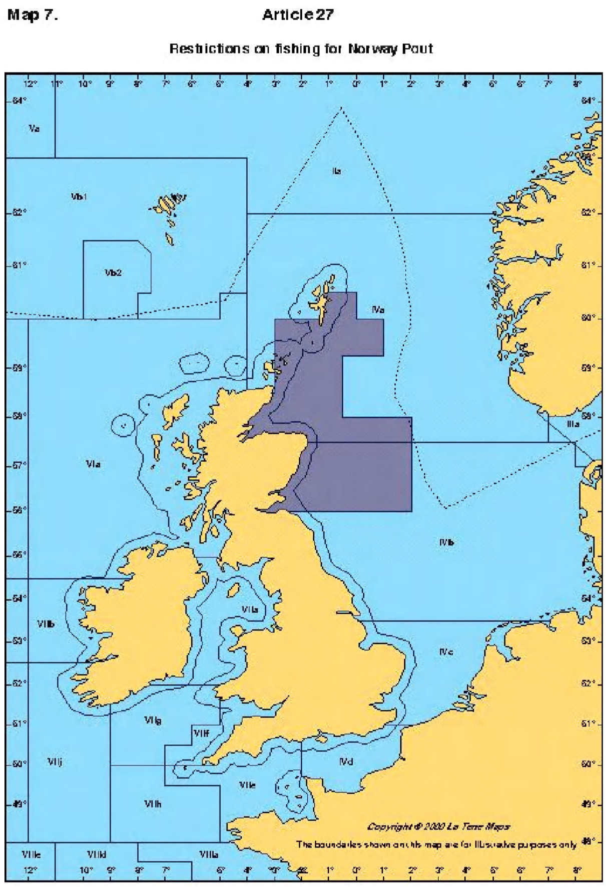


Figure 11.1.5.2. The "Pout Box" where fishing for Norway pout is not allowed.

in relation to the profitability of fishing at the various potential locations. Such a change in location occurred in 1990, when a very rapid expansion in the catch from the area off the Firth of Forth occurred. This coincided with a decline in breeding performance of black-legged kittiwakes (*Rissa tridactyla*) on nearby coasts, a decline not reflected elsewhere. If, as seems likely, the local sandeel population is relatively isolated and is relied upon by seabirds during the breeding season, fishing pressure may have a disproportionately large effect. Wanless *et al*. (1998) compared the amounts of sandeels taken from this area by fisheries and by birds and concluded that fisheries were more likely to affect seabirds than vice versa.

The studies of Jensen *et al*. (1994) and Wright and Begg (1997) demonstrate that spatial scale is critical to understanding seabird-sandeel interactions and that it is essential to understand the fish stock in these terms. For many years it was assumed, for fisheries assessment purposes, that the North Sea sandeel "stock" could be split into a northern and a southern component, with a small isolated group near Shetland. Recent research has indicated that there may be many more subdivisions within the stock (Wright *et al*., 1998).

The ELIFONTS (Effects of Large-Scale Industrial Fisheries on Non-target Species) study (Harwood *et al*., 2000) examined the diets and breeding success of common guillemots, European shags (*Phalacrocorax aristotelis*), and black-legged kittiwakes, as well as the biology of sandeels in the feeding area of the Isle of May colony, in the period 1997 and 1998. The total biomass of sandeels in 1998 was 15% less than in 1997. In both years, the sandeel population was dominated by a very strong (1996) year class. Total removals from the sandeel population were similar in both years (69,000 tonnes in 1997 and 65,000 tonnes in 1998). Fish were the most important natural predator in both years. The fishery was responsible for 68% of all removals in 1998, compared to 34% in 1997. The proportion of sandeels in the diet of common guillemots declined by 70% in 1998 compared with 1997, with a switch to clupeids. The diet of European shags and black-legged kittiwakes showed much less change and was dominated by sandeels in both years. The two diving seabirds (common guillemots and European shags) appeared to work harder in 1998. Common guillemots spent less time in the colony, left their chicks unattended, and increased their foraging range. Both common guillemots and European shags spent more time diving and proportionally less time at the surface. In contrast, the surface-feeding black-legged kittiwakes did not, or could not, change their foraging behaviour. Kittiwakes suffered an almost complete breeding failure in 1998, whereas the productivity of common guillemots and European shags was only slightly reduced. The breeding success of common guillemots, European shags, and black-legged kittiwakes in the period 1990 to 1998 was strongly correlated with the abundance of 1+ sandeels as estimated by CPUE for catches made in June in ICES rectangle 41E8, and negatively correlated with the ratio of May CPUE to June CPUE.

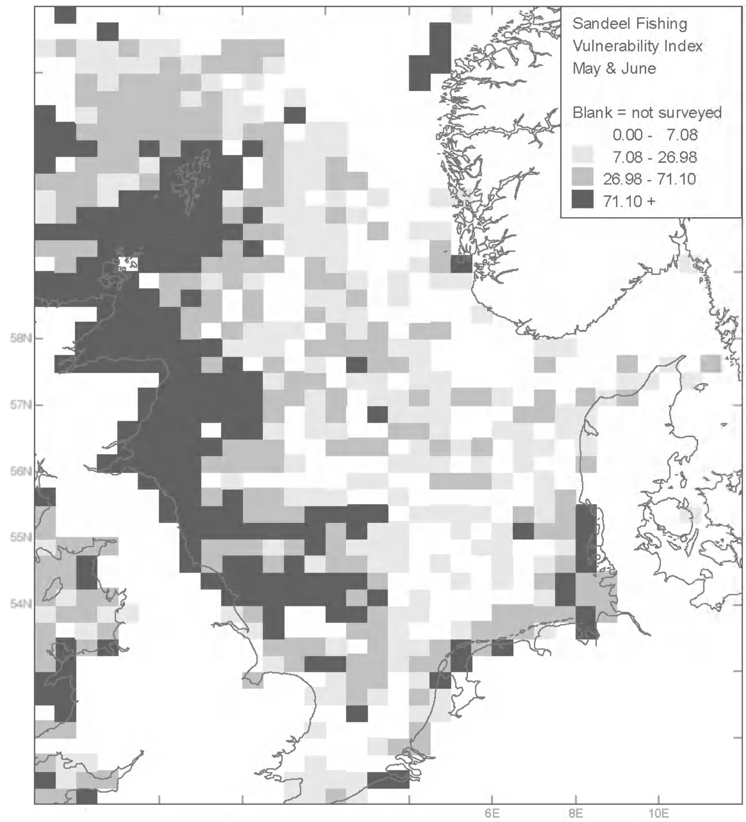


Figure 11.2.1.2.1. Sensitivity of seabirds to changes in sandeel abundance in May and June. The darker the shading, the higher the sensitivity to change (ICES, 1999).

The ELIFONTS results suggest that relatively small changes in the timing of peak sandeel availability during June are a major determinant of seabird breeding success on the Isle of May. It appears that the precise timing of two key events in the sandeel life cycle is critical: the onset of burying behaviour and the arrival of 0-group fish on the seabirds' feeding grounds. Both occur in June, when the seabirds have the highest energy demands and are thus extremely sensitive to reductions in food availability. Although the commercial fisheries do not influence the timing of these events, most catches are taken in June and the fishery could exacerbate a difficult situation for seabirds by further reducing the biomass of available age 1+ fish. This appears to have happened in 1993.

The results of the ELIFONTS project show that it is necessary to understand the interaction between the foraging strategy of the birds and the behaviour and life cycle of the sandeel in order to understand the effects of changes in sandeel abundance. All of the predators in this study showed measurable responses to the change in age structure of the sandeel population between 1997 and 1998, but each responded differently. For the bird predators, it was possible to demonstrate a relationship between sandeel availability (at a relatively small scale) and breeding performance. The main factors determining sandeel availability for most of the predators were recruitment and the effects of environmental variation on behaviour. For seabirds the precise time at which 0-group sandeels become available, and their average size, also appeared to be important factors. None of these are likely to be affected by the activities of the commercial fishery at the time of the fishing operations. However, if sandeel recruitment to the study area is heavily dependent on the local spawning stock, then fishing could affect the abundance of 0+-group fish in subsequent years. Years when there is a relatively low abundance of 1+-group sandeels and the 0+-group fish arrive late are particularly bad for seabirds. Large catches during June in these years could make this situation worse.

IMPRESS (Interactions between the marine environment, predators and prey: implications for sustainable sandeel fisheries) grew out of the ELIFONTS project. The findings from ELIFONTS were considered in advice to the EU Council's decision to prohibit fishing for sandeels in a 20,000 km^2 band of sea down the east coast of Britain in 2000 (Figure 11.2.1.2.2). The black-legged kittiwakes in the region showed an increase in breeding success comparable to pre-fishery levels, but it was difficult to determine whether recovery was due to the cessation of fishing or climatic drivers.

Thus, IMPRESS has been designed to take a bottom-up approach to determine the effect of climate and hydrography on temporal and spatial patterns in sandeel abundance and the performance of seabirds and other predators with respect to sustainable fisheries.

Currently, work is focused on:

- recording of foraging behaviour of seabirds and marine mammals and multispecies foraging associations;
- defining foraging habitats of prey species based on environmental parameters and technical development to assist in this process;
- understanding marine food-web processes as part of an ecosystem approach to ensuring sustainable sandeel fisheries in the North Sea.

In one of the most intensively studied seabird-sandeel interactions, certain seabirds, particularly Arctic terns (*Sterna paradisaea*) 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 associated 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.

11.2.2 Effects of fisheries for sprat on seabirds

11.2.2.1 Direct effects

There have been no studies and no records of direct interaction with this fishery.

11.2.2.2 Indirect effects

In the study of Jensen *et al.* (1994), there was an indication that the fishery for sprat in the southern North Sea in the autumn overlapped spatially with the common guillemot exploitation of this stock/area in winter. Blake (1984) considered that the periodic late winter (non-oiling) mass mortalities of common guillemot were most likely to be caused by difficulties in finding enough sprat to eat at this time of year. If the sprat fishery is affecting the sprat stock, there might then be an indirect effect of the fishery on common guillemots.

11.2.3 Effects of fisheries for blue whiting on seabirds

11.2.3.1 Direct effects

There have been no studies and no records of direct interaction with this fishery.

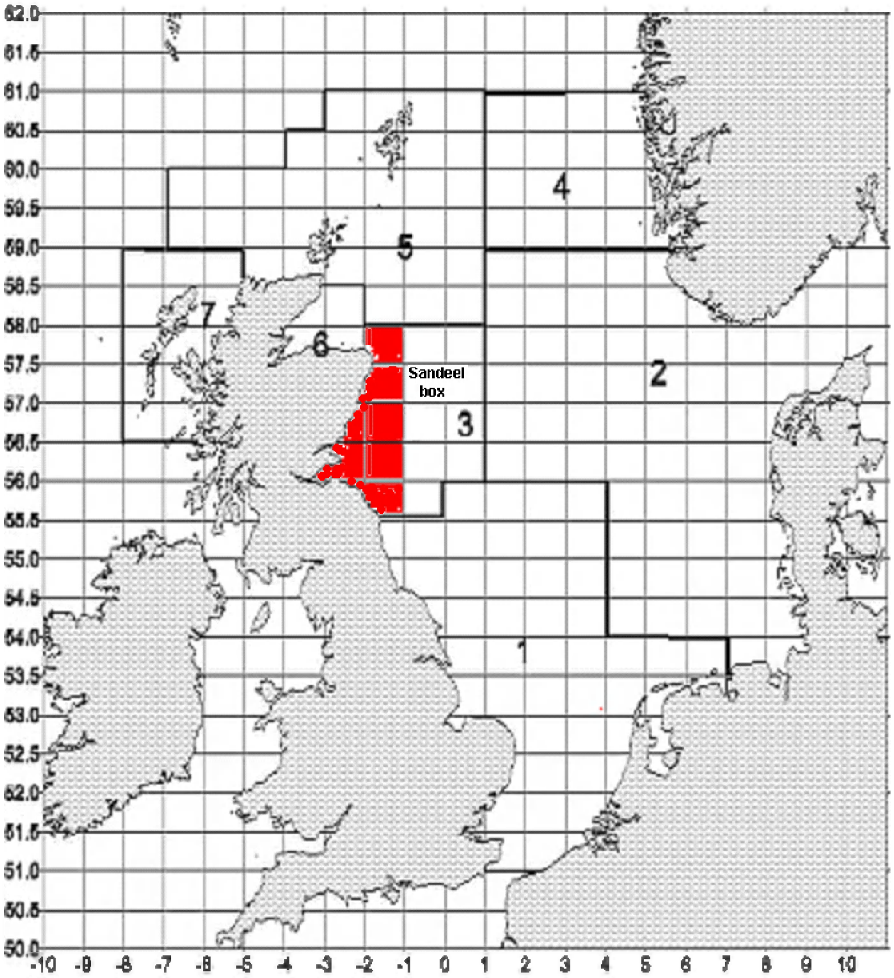


Figure 11.2.1.2.2. The area closed for sandeel fishing off the east coast of Scotland from 2000 onwards.

11.2.3.2 Indirect effects

Blue whiting are rare in seabird diets in EU waters. It seems unlikely that there would be a large indirect effect on seabirds from this fishery.

11.2.4 Effects of fisheries for Norway pout on seabirds

11.2.4.1 Direct effects

There have been no studies and no records of direct interaction with this fishery.

11.2.4.2 Indirect effects

Norway pout do not appear to form a large proportion of the diet of any seabird species. The maximum proportion in the diet of any seabird species in the North Sea for gadoid fish is about one third (Tasker and Furness, 1996), and this category includes all gadoid species, not just Norway pout. In addition, the area of closure of part of the northwestern North Sea to Norway pout fisheries coincides with much of the main area of seabird predation in the North Sea (Tasker and Furness, 1996). It seems unlikely that any depletion of Norway pout, if it were happening through industrial fishing, would affect seabirds in any significant fashion.

11.3 Effects on marine mammals

11.3.1 Direct effects of industrial fisheries on marine mammals

No records could be found of marine mammal by-catch in fisheries for sandeel, sprat, blue whiting or Norway pout, though this may be due to a lack of study or reporting. A by-catch of dolphins (mostly common dolphin (*Delphinus delphis*)) occurs in pelagic trawl fisheries for mackerel/horse mackerel off southwestern Ireland and the UK. A portion of the products from this and similar fisheries is used for reduction purposes, but this fishery is outside the current term of reference. It seems likely that the overall direct effects of industrial fisheries on marine mammals are small.

11.3.2 Indirect effects of fisheries for sandeel on marine mammals

The grey seal (*Halichoerus grypus*) population of the Isle of May was included in the ELIFONTS study (Harwood *et al.*, 2000). This study extended over two years (1997 and 1998) in the Wee Bankie area off the Firth of Forth where a sandeel fishery was present. The total biomass of sandeels in 1998 was 15% less than in 1997. In both years, the sandeel population was dominated by a very strong (1996) year class. Total removals from the sandeel population were similar in both years (69,000 tonnes in 1997 and 65,000 tonnes in 1998). Fish were the most important natural predator in both years. The fishery was responsible for 68% of all removals in 1998, compared to 34% in 1997.

Sandeels appeared to be less available to predators in 1998. Sandeels (mainly 1-year-old and 3-year-old fish) made up nearly 50% of the diet of grey seals in 1997, but only around 10% in 1998 (and in this year they were mostly 2-year-old fish). More cod and whiting were consumed by grey seals in 1998. The proportion of female grey seals not breeding in a particular year at the Isle of May between 1990 and 1997, and the number of breeding failures amongst marked animals, was correlated with sandeel CPUE in the southern North Sea. Female body condition was positively correlated with CPUE for the local stock area. None of these relationships had a measurable effect on the total number of pups born at the Isle of May, which increased steadily over the study period. There is, therefore, a relationship between sandeel abundance and seal breeding success at a large scale, but it is difficult to demonstrate a relationship between sandeel abundance and the commercial fisheries.

The diets of seals are reasonably well known from studies at rookeries and haul-out sites, and information continues to be collected.

11.3.3 Indirect effects of fisheries for blue whiting on marine mammals

In northern waters, blue whiting have been recorded in the diet of long-finned pilot whales (*Globicephala melas*) (Desportes and Mouritsen, 1993), harp seal (*Phoca groenlandica*), and minke whale (*Balaenoptera acutorostrata*) (Bogstad *et al.*, 2000). Off Portugal, common dolphins have been recorded eating this species. The ecosystem effects of any fishing on this interaction have not been evaluated.

11.4 Effects on seabed habitats and benthos

There have been no direct studies on the effects of industrial fisheries on seabed habitats and benthos, although it is unlikely that they differ significantly from those attributable to other bottom-fishing gears as reviewed by Hall (1999). This section is therefore based on the application of knowledge from general literature to the fishing gear and seabed habitats and benthos in the areas of the fisheries.

Purse seines and light otter trawls are used in the industrial fisheries to capture small pelagic fish. Purse seines have no direct impact on the sea floor as they operate within the water column. However, otter trawls disturb the benthos as the boards used to hold open the net disturb and may penetrate the sea floor. The penetration depth is dependent upon the weight of the boards and the substrate; the boards will penetrate finer substrates to a greater depth than coarse substrates (Brylinsky *et al.*, 1994). Large amounts of sediment are resuspended during otter trawling. It is estimated that an otter trawl which penetrates coarse sand to a depth of approximately 1 cm will resuspend 39 kg sec^{-1}, whilst in muddy sediments, the boards may penetrate to 4 cm, resuspending 112 kg sec^{-1} (Churchill, 1989). A single tow of an otter trawl may resuspend the upper layer of sediment from an area of about 144,000 m^2 (Hall, 1994).

The effect of this disturbance on the more dynamic habitats, such as on sand, is less important than when the disturbance occurs in areas of lower energy such as on muddy substrates and in deep water, as the level of natural disturbance in the more dynamic areas is likely to be greater than that caused by fishing (Kaiser *et al.*, 1998). Tracks were visible for up to one year after trawling the sandy sea floor of the Grand Banks of Newfoundland (Schwinghamer *et al.*, 1998), and up to 18 months on muddy substrates in the Irish Sea (Ball *et al.*, 2000). The effect of this disturbance is likely to be important to fragile and sensitive species which cannot tolerate physical disturbance (e.g., *Lophelia*) or smothering by sediment (e.g., oysters). However, the sandeel fisheries tend to occur on sandbank areas which encounter relatively high levels of natural disturbance. Also, since the sandeel fisheries occur in areas of hard sandy substrates, there is an incentive for the fishermen to keep gear contact with the seabed to a minimum to avoid damage, so the gears have only sporadic contact with the sea floor. The Norway pout fishery is likely to have a greater effect on the benthos as the fishery occurs primarily in the central northern North Sea, generally in deeper water and over relatively soft sediments, and the otter boards are operated in closer contact with the sea floor.

Although the impact to the seabed and benthos by each individual tow may be less than with comparable demersal otter trawling operations as the gears are lighter, the way the fishery operates suggests that local impact on the seabed and invertebrate communities may be quite intense. This is because the same trawl path tends to be fished repeatedly over a period of several days by several boats operating in any particular region. In mitigation, however, is the fact that these fisheries are seasonal. The local impact may be intense, but it is followed by long periods of recovery.

11.5 Relative benefits of industrial fishing and aquaculture versus capture fisheries

The ecological efficiency of industrial fishing and aquaculture versus capture fisheries could be compared in many ways. A direct comparison could be made of the efficiency of converting small industrial fish into fish readily eaten by humans, either in the wild (e.g., cod, whiting) or in captivity (e.g., Atlantic salmon (*Salmo salar*)). Such a comparison is complex, however, because prey species and size classes vary in energy content and energy:nutrient conversion ratios vary between species. Carnivorous species need to eat 2.5 g to 5.0 g of wild fish for every 1.0 g of tissue they produce (conversion ratio 2.5:1 or 5.0:1), while the organisms feeding at a lower trophic level require less wild fish input (Table 11.5.1). The amount of wild fish biomass required for culturing commercially valuable species depends on the type of mariculture and aquaculture. Comparisons are thus complex.

The environmental effects of the two methods of fish protein production for human consumption differ markedly. The effects of industrial fishing are reviewed earlier in this section and by Hall (1999) and Frid and Dobson (2002), and the effects of aquaculture are widely reviewed elsewhere (Hanley and Couriel, 1992; Beveridge *et al.*, 1994; Soley *et al.*, 1994; Findlay *et al.*, 1995; McIntyre, 1995; Findlay and Watling, 1997; Tasker *et al.*, 2000; Chamberlain *et al.*, 2001; Pearson and Black, 2001). A major impact of aquaculture is the transfer of organic matter from one system to another, e.g., diminution of nutrients in one area and increase of nutrients in another (Fischer *et al.*, 1997). On average, 25% of the nutrients found in fish feed are converted into biomass (UBA, 1996), and the remainder impacts adjacent systems. Excess nutrients lead to increased algal growth, potentially leading to eutrophication, with the concomitant problems of anoxia and, in some cases, the growth of toxic phytoplankton species (OSPAR, 2000).

Table 11.5.1. Global inputs of wild fish to fish and shellfish production in 1997 (after Naylor *et al.*, 2000). Marine finfish include flounder, halibut, sole, cod, hake, haddock, redfish, sea bass, congers, tuna, bonito, and billfish. Fed carp refers to carp species that are sometimes fed compound feeds. Filter-feeding carp are silver carp, bighead carp, and catla. It was assumed that compound feed was used to produce farmed fish, that the conversion ratio for fishmeal was 5:1, and that one-sixteenth of fishmeal was produced from by-products of processing.

Farmed Fish	Total production ('000 t)	Percentage produced with compound feeds (by weight)	Production with compound feeds ('000 t)	Percentage fishmeal in feed	Percentage fish oil in feed	Average feed conversion ratio	Wild fish used for fishmeal ('000 t)	Ratio of wild fish: fed farmed fish
Marine finfish	754	50	377	50	15	2.2	1,944	5.16
Eel	233	50	117	50	10	2	546	4.69
Marine shrimp	942	77	725	30	2	2	2,040	2.81
Salmon	737	100	737	45	25	1.5	2,332	3.16
Trout	473	100	473	35	20	1.5	1,164	2.46
Tilapia	946	35	331	15	1	2	466	1.41
Milkfish	392	20	78	10	3	2	74	0.94
Catfish	428	82	351	10	3	1.8	296	0.84
Fed carp	6,985	35	2,445	8	1	2	1,834	0.75
Filter-feeding	5,189	0	0	-	-	-	-	-
Molluscs	7,321	0	0	-	-	-	-	-
Total	24,400		5,634				10,695	1.90

11.6 Research priorities

ICES (2002a) previously identified a number of industrial stocks for which ecological dependence may need to be considered in management advice. These are typically "forage fish" stocks for which quantitative assessments may or may not be available and which, on the basis of existing observations on the distribution and abundance of associated predators and (in some cases) their diets, may have ecologically dependent predators. The stocks were:

Capelin in the Iceland-East Greenland-Jan Meyen area;

Sandeel in Division IIIa;

Norway pout in Sub-area IV and Division IIIa;

Sandeel in Sub-area IV;

Norway pout in Division VIa;

Sandeel in Division VIa.

There is still relatively scant information on the effects of fisheries targeting these stocks and further analysis of the ecological impacts of these fisheries is required. In addition, we have highlighted the minimal understanding of the effects of the blue whiting fishery. This is the largest (by landings) industrial fishery in the ICES area, and yet the rates of by-catch and the ecological impacts of this fishery are not known. The following specific questions need to be asked about this fishery:

What else is caught in the blue whiting fishery?

ICES has been unable to find information on the catches of other species in the blue whiting fishery. It may be that such catches are insignificant, but it may also be because the catch is primarily composed of non-commercial species. To understand the ecosystem effects of this large fishery, the catch of these non-commercial species needs to be recorded.

What eats blue whiting?

Given the apparent size of the blue whiting stock, ICES would expect it to form an important component of the diet of top predators (at least in its younger stages). ICES has found little information on this aspect of blue whiting ecology.

Is blue whiting a keystone species?

In the shallow waters of the North Sea, sandeels are the prey of many predators and are important consumers of the plankton; sandeels are often described as a keystone species. Given the size of the blue whiting stock, ICES would expect this stock to be key above the shelf slope of the Atlantic. Research on its role in the ecosystem is needed.

11.7 References

Ball, B., Munday, B., and Tuck, I. 2000. Effects of otter trawling on the benthos and environment in muddy sediments. *In* Effects of fishing on non-target species and habitats: biological conservation and socio-economic issues, pp. 69–82. Ed. by M.J. Kaiser and S.J. de Groot. Blackwell Science, Oxford.

Bergstad, O.A. 1990. Ecology of the fishes of the Norwegian Deep: distribution and species assemblages. Netherlands Journal of Sea Research, 25: 237–266.

Beveridge, M.C.M., Ross, L.G., and Kelly, L.A. 1994. Aquaculture and biodiversity. Ambio, 23: 497–502.

Bjelland, O., and Bergstad, O.A. 1998. Trophic ecology of deepwater fishes associated with the continental slope of the eastern Norwegian Sea. ICES CM 1998/O:51. 26 pp.

Bjelland, O., Bergstad, O.A., Skjæraasen, J.E., and Meland, K. 2000. Trophic ecology of deep-water fishes associated with the continental slope of the eastern Norwegian Sea. Sarsia, 85: 101–117.

Blake, B.F. 1984. Diet and fish stock availability as possible factors in the mass death of auks in the North Sea. Journal of Experimental Marine Biology and Ecology, 76: 89–103.

Bogstad, B., Haug, T., and Mehl, S. 2000. Who eats whom in the Barents Sea? NAMMCO Scientific Publication, 2: 98–119.

Brylinsky, M., Gibson, J., and Gordon, D.C. 1994. Impacts of flounder trawls on the intertidal habitat and community of the Minas Basin, Bay of Fundy. Canadian Journal of Fisheries and Aquatic Sciences, 51: 650–661.

Chamberlain, J., Fernandes, T.F., Read, P., Nickell, T.D., and Davies, I.M. 2001. Impacts of biodeposits from suspended mussel (*Mytilus edulis* L.) culture on the surrounding surficial sediments. ICES Journal of Marine Science, 58: 411–416.

Churchill, J.H. 1989. The effect of commercial trawling on sediment resuspension and transport over the Middle Atlantic Bight continental shelf. Continental Shelf Research, 9: 841–864.

Corten, A. 1986. On the causes of the recruitment failure of herring in the central and northern North Sea in the years 1972–78. Journal du Conseil International pour l'Exploration de la Mer, 42: 281–291.

Corten, A. 1990. Long-term trends in pelagic fish stocks of the North Sea and adjacent waters and their possible connection to hydrographic changes. Netherlands Journal of Sea Research, 25: 227–235.

Daan, N. 1980. A review of replacement of depleted stocks by other species and the mechanisms underlying such replacement. Rapports et Procès-Verbaux des Réunions, Conseil International pour l'Exploration de la Mer, 177: 405–421.

Dalskov, J. 2002. Description of the Danish monitoring scheme for the small meshed fishery in the North Sea, Skagerrak and Kattegat. Report of the Danish Institute for Fisheries Research, Charlottenlund.

Danchin, E. 1992. Food shortage as a factor in the 1988 kittiwake *Rissa tridactyla* breeding failure in Shetland. Ardea, 80: 92–98.

Desportes, G., and Mouritsen, R. 1993. Preliminary results on the diet of long-finned pilot whales off the Faroe Islands. Reports of the International Whaling Commission, Special Issue 14: 305–324.

Du Buit, M.H. 1995. Blue whiting in food webs in the Celtic Sea. Journal of Fish Biology, 45A: 245.

Du Buit, M.H. 1996. Diet of hake (*Merluccius merluccius*) in the Celtic Sea. Fisheries Research, 28: 381–394.

Duplisea, D.E., Kerr, S.R., and Dickie, L.M. 1997. Demersal fish biomass size spectra on the Scotian shelf, Canada: species replacement at the shelf wide scale. Canadian Journal of Fisheries and Aquatic Sciences, 54: 1725–1735.

Fehervari, Z., and Naevdal, G. 1995. A pilot study of inter- and intraspecific variations in sandeels (Fam. Ammodytidae). Institutt for fiskeri- og marinbiologi rapport, 1995, no. 5. 19 pp.

Findlay, R.H., and Watling, L. 1997. Prediction of benthic impact for salmon net-pens based on the balance of benthic oxygen supply and demand. Marine Ecology Progress Series, 155: 147–157.

Findlay, R.H., Watling, L., and Mayer, L.M. 1995. Environmental impact of salmon net pen culture on marine benthic communities in Maine—a case-study. Estuaries, 18: 145–179.

Fischer, J., Haedrich, R.L., and Sinclair, P.R. 1997. Inter ecosystem impacts of forage fish fisheries. *In* Role of forage fishes in marine ecosystems, pp. 311–321. Alaska Sea Grant College Program Report No. 97-01, Anchorage.

Frid, C.L.J., and Dobson, M. 2002. Ecology of aquatic management: resources, pollution and sustainability. Pearson Education, Harlow, UK. 274 pp.

Hall, S.J. 1994. Physical disturbance and marine benthic communities: life in unconsolidated sediments. Oceanography and Marine Biology, 32: 179–239.

Hall, S.J. 1999. The effects of fishing on marine ecosystems and communities. Blackwell Science, Oxford. 296 pp.

Hanley, J.R., and Couriel, D. 1992. Coastal management issues in the Northern-Territory—an assessment of current and future problems. Marine Pollution Bulletin, 25: 134–142.

Harris, M.P., and Bailey, R.S. 1992. Mortality-rates of puffin *Fratercula arctica* and guillemot *Uria aalge* and fish abundance in the North Sea. Biological Conservation, 60: 39–46.

Harwood, J., Hall, A.J., McConnell, B., Pomeroy, P., Duck, C., Fedak, M., Matthiopoulos, J., Walton, M.J., Greenstreet, S.P.R., Emmerson, H.M.F., Armstrong, E., Gibb, I., Doyle, C.J., Wanless, S., Rindorf, A., Pedersen, S.A., Jensen, H., Davis, J., and Ligdas, C.N. 2000. Effects of large-scale industrial fisheries on non-target species (ELIFONTS), Report of EU project No. 95/078. Sea Mammal Research Unit, St. Andrews, UK.

Heino, M., and Godø, O.L. 2002. Blue whiting – a key species in the mid-water ecosystems of the north-eastern Atlantic. ICES CM 2002/L:28

Hill, L., and Borges, M.F. 2000. A comparison of the seasonal abundance of hake (*Merluccius merluccius*) and its main prey species off the Portuguese coast. ICES CM 2000/Q:13.

Hill, L., Macara, H., and Borges, M.F. 1999. Changes in the diet of hake (*Merluccius merluccius*) in Portuguese coastal waters since 1983. Poster presented at the ICES/SCOR Symposium on Ecosystem Effects of Fishing, Montpellier, France.

ICES. 1996. Report of the Working Group on the Assessment of the Demersal Stocks in the North Sea and Skagerrak. ICES CM 1996/Assess:6.

ICES. 1997a. Report of the Multispecies Assessment Working Group. ICES CM 1997/Assess:16.

ICES. 1997b. Database of the stomach sampling project 1991. ICES Cooperative Research Report No. 219.

ICES. 1999. Report of the Study Group on Effects of Sandeel Fishing. ICES CM 1999/ACFM:19. 12 pp.

ICES. 2000. Report of the ICES Advisory Committee on Fishery Management, 1999. ICES Cooperative Research Report No. 236.

ICES. 2001a. Report of the ICES Advisory Committee on Fishery Management, 2001. ICES Cooperative Research Report No. 246.

ICES. 2001b. Report of the Herring Assessment Working Group for the Area South of 62°N. ICES CM 2001/ACFM:12.

ICES. 2002a. Report of the ICES Advisory Committee on Ecosystems, 2002. ICES Cooperative Research Report, 254: 49–53.

ICES. 2002b. Report of the ICES Advisory Committee on Fishery Management, 2002. ICES Cooperative Research Report No. 255.

ICES. 2002c. Report of the Northern Pelagic and Blue Whiting Fisheries Working Group. ICES CM 2002/ACFM:19.

ICES. 2003a. Report of the Working Group on the Assessment of the Demersal Stocks in the North Sea and Skagerrak. ICES CM 2003/ACFM:02.

ICES. 2003b. Report of the Northern Pelagic and Blue Whiting Fisheries Working Group. ICES CM 2002/ACFM:19.

ICES. 2003c. Report of the Baltic Fisheries Assessment Working Group. ICES CM 2003/ACFM:21.

Iwama, G.K. 1991. Interactions between aquaculture and the environment. Critical Reviews in Environmental Control, 21: 177–216.

Jensen, H., Tasker, M.L., Coull, K., and Emslie, D. 1994. A comparison of distribution of seabirds and preyfish stocks in the North Sea and adjacent areas. JNCC Report 207/Final report to EC DGXIV PEN 92/3501. 116 pp.

Kaiser, M.J., Edwards, D.B., Armstrong, P.J., Radford, K., Lough, N.E.L., Flatt, R.P., and Jones, H.D. 1998. Changes in megafaunal benthic communities in different habitats after trawling disturbance. ICES Journal of Marine Science, 55: 353–361.

Macer, C.T. 1966. Sand eels (Ammodytidae) in the south-western North Sea; their biology and fishery. Fisheries Investigations, London Series 2, 24(6): 1–55.

McIntyre, A.D. 1995. Human impact on the oceans - the 1990s and beyond. Marine Pollution Bulletin, 31: 147–151.

Monaghan, P., Uttley, J.D., and Burns, M.D. 1992. Effect of changes in food availability on reproductive effort in Arctic terns *Sterna paradisaea*. Ardea, 80: 71–81.

Newton, A., Coull, K., Peach, K., Coggan, A., Robb, A., Blasdale, T., Breen, M., Burns, F., Davis, S., and Bullough, L. 2002. Report on biological information gathered from Scottish fishing vessels. Industry/Science Partnership, Aberdeen.

Naylor, R., Goldburg, R., Primavera, J., Kautsky, N., Beveridge, M., Clay, J., Folke, C., Lubchenco, R., Mooney H., and Troell, M. 2000. Effect of aquaculture on world fish supplies. Nature, 405: 1017–1024.

OSPAR. 2000. Quality status report 2000. OSPAR Commission, London. 107 pp.

Pearson, T.H., and Black, K.D. 2001. The environmental impact of marine fish cage culture. *In* Environmental impacts of aquaculture, pp. 1–31. Ed. by K.D. Black. Sheffield Academic Press, Sheffield.

Phillips, R.A., Caldow, R.W.G., and Furness, R.W. 1996. The influence of food availability on the breeding effort and reproductive success of Arctic skuas *Stercorarius parasiticus*. Ibis, 138: 410–419.

Pope, J.G., and Macer, C.T. 1996. An evaluation of the structure of North Sea cod, haddock and whiting since 1920, together with a consideration of the impacts of fisheries and predation effects on their biomass and recruitment. ICES Journal of Marine Science, 53: 1157–1169.

Procter, R., Wright, P.J., and Everitt, A. 1998. Modelling the transport of larval sandeels on the north-west European shelf. Fisheries Oceanography, 7: 347–354.

Schwinghamer, P., Gordon, D.C., Rowell, T.W., Prena, J., McKeown, D.L., Sonnichsen, D., and Guigne, J.Y. 1998. Effects of experimental otter trawling on surficial sediment properties of a sandy bottom ecosystem on the Grand Banks of Newfoundland. Conservation Biology, 12: 1215–1222.

Silva, A., Azevedo, M., Cabral, H., Machado, P., Murta, A., and Silva, M.A. 1997. Blue whiting (*Micromesistius poutassou*) as a forage fish in Portuguese waters. International Symposium on the Role of Forage Fishes in Marine Ecosystems, Anchorage, Alaska (USA), 13–16 Nov 1996. Lowell Wakefield Fisheries Symposium Series, 14: 127–146.

Soley, N., Neiland, A., and Nowell, D. 1994. An economic approach to pollution control in aquaculture. Marine Pollution Bulletin, 28: 170–177.

Southward, A.J., Boalch, G.T., and Maddock, L. 1988. Fluctuations in the herring and pilchard fisheries of Devon and Cornwall linked to changes in climate since the 16th century. Journal of the Marine Biological Association of the United Kingdom, 68: 423–445.

Sparholt, H., Larsen, L.I., and Nielsen J.R. 2002. Non-predation natural mortality of Norway pout (*Trisopterus esmarkii*) in the North Sea. ICES Journal of Marine Science, 59: 1276–1284.

Stratoudakis, Y., Fryer, R.J., Cook, R.M., Pierce, G.J., and Coull, K.A. 2001. Fish bycatch and discarding in Nephrops trawlers in the Firth of Clyde (west of Scotland). Aquatic Living Resources, 14: 283–291.

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.

Tasker, M.L., and Furness, R.W. 1996. Estimation of food consumption by seabirds in the North Sea. ICES Cooperative Research Report, 216: 6–42.

UBA. 1996. Einflüsse der Fisherei und Aquakultur auf die marine Umwelt. UBA Texte 46-96 Umweltbundesamt, Berlin. 136 pp.

Wanless, S., Harris, M.P., and Greenstreet, S.P.R. 1998. Summer sandeel consumption by seabirds breeding in the Firth of Forth, south-east Scotland. ICES Journal of Marine Science, 55: 1141–1151

Wright, P.J. 1996. Is there a conflict between sandeel fisheries and seabirds? A case study at Shetland. *In* Aquatic predators and their prey, pp. 154–165. Ed. by S.P.R. Greenstreet and M.L. Tasker. Fishing News Books, Oxford.

Wright, P., and Bailey, M.C. 1993. Biology of sandeels in the vicinity of seabird colonies at Shetland. Fisheries Research Services Report, 15/93. Marine Laboratory, Aberdeen. 64 pp.

Wright, P.J., and Begg, G.S. 1997. A spatial comparison of common guillemots and sandeels in Scottish waters. ICES Journal of Marine Science, 54: 578–592.

Wright, P.J., Verspoor, E., Anderson, C., Donald, L., Kennedy, F., Mitchell, A., Munk, P., Pedersen, S.A., Jensen, H., Gislason, H., and Lewy, P. 1998. Population structure in the lesser sandeel (*Ammodytes marinus*) and its implications for fishery-predator interactions. Final Report to DG XIV 94/C 144/04 Study Proposal 94/071. Marine Laboratory, Aberdeen. 59 pp.

Zilanov, V.K. 1968. Some data on the biology of *Micromesistius poutassou* (Risso) in the North-east Atlantic. Journal du Conseil International pour l'Exploration de la Mer, Rapports et Procès-Verbaux des Réunions, 158: 116–122.

12 IMPACT OF CURRENT FISHING PRACTICES ON NON-TARGET SPECIES

Request

The European Commission, Directorate General for Fisheries, has expressed (in a letter of 20 September 2001) its immediate interest in an *"Evaluation of the impact of current fishing practices on 'non-target species' ... and suggestions for appropriate mitigating measures"*.

Source of information

The 2003 Report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO) (ICES CM 2002/ ACE:03).

Scientific background

12.1 Introduction

In 2002, ACE described some general problems related to the provision of advice on non-target species (ICES, 2002). The lack of priority in research programmes aimed at analysing existing data or collecting new data largely prohibits the provision of management advice, even though various non-target fish species may be presently at risk. Only for elasmobranchs is progress being made and relevant information is expected in 2004. For all other species, the situation has remained virtually unchanged from last year and no relevant new information on the impact of current fishing practices on non-target species or appropriate mitigation measures can be provided.

In the past, it has proved impossible to collate the information on non-target species from recent EU discard sampling programmes, because these data are treated by some countries as confidential. However, the EC Common Fisheries Policy requires reporting of the by-catch of species listed in Annex II of Council Regulation (EC) No. 1543/2000 "Community framework for the collection and management of the data needed to conduct the Common Fisheries Policy". At present, the species listed are all target species, but the Annex II list could be extended by Council decision. The Århus Convention on Access to Information, Public Participation in Decision-making and Access to Justice in Environmental Matters (entered into force on 30 October 2001) guarantees access to environmental data and should ensure that data collected by other European nations, i.e., those which are not members of the EU, are also made available. Thus, although in principle all information can and should be made available, in practice there are major stumbling blocks.

The other important input for an evaluation of the potential effects of fishing on non-target species is information on trends in (preferably absolute) biomass estimates from surveys. While a large amount of survey data is available for this purpose, a major research effort would be required to make a comprehensive evaluation of all these data. It seems unlikely that this can be achieved unless specific funds are made available.

12.2 Conclusion

To further evaluate the impacts of fisheries on non-target species and potential mitigation measures, two important issues must be addressed:

1) Information on the spatio-temporal distribution of non-target species in discards by major fleet must be made available;
2) Survey information on spatio-temporal variations in the abundance of non-target species must be analysed, preferably in terms of absolute biomasses.

Further progress depends entirely on comprehensive information being made available to ICES working groups.

Reference

ICES. 2002. Report of the ICES Advisory Committee on Ecosystems, 2002. ICES Cooperative Research Report, 254: 47–49.

13 PRESERVATION OF GENETIC DIVERSITY OF EXPLOITED STOCKS

Request

The European Commission, Directorate General for Fisheries, has expressed its immediate interest in the "*development of advisory forms appropriate to the preservation of genetic diversity from detrimental impacts of fishing*".

Source of information

The 2003 Report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO) (ICES CM 2003/ ACE:03).

The 2003 Report of the Working Group on the Application of Genetics in Fisheries and Mariculture (WGAGFM) (ICES CM 2003/F:01).

Summary

Four general measures have previously been suggested by ICES to mitigate against the loss of genetic diversity, recommending that alternative management options be evaluated on a case-specific basis. These were suggested as "common sense" approaches for managers to follow until the scientific community could recommend a more rigorous framework. Considerations on the preservation of the genetic diversity within and between populations and population structure were provided with corresponding examples of management objectives. The 2003 review of the management objectives leads to the conclusion that there would be many different management objectives for the different threats to genetic diversity and the species affected. A prioritization scheme is proposed to assist with the decision on which population to protect. Marine organisms are grouped into five categories based on life history characteristics (fecundity, size, and life duration) and sedentary nature (motility, dispersal range). For each of these groups, the threats are defined and generic genetic concerns are identified.

Recommendations

ICES recommends that:

1) an additional limit reference point for the management objective "maintain number of populations" be considered as it could help maintain all evolutionary units above the level of population;
2) the geographical area of concern for the preservation of the genetic diversity of exploited stocks and by-catch stocks be defined;
3) lists be produced of 1) species for which we have reason to be concerned about loss of genetic variation; and 2) species for which we have good genetic information from which to advance management advice.

Management objectives and associated reference points, where possible, will be determined for the species on these lists in order to complete the advisory forms appropriate to the preservation of the genetic diversity of the stocks.

Scientific background

13.1 Introduction

In its 2002 report, ACE (ICES, 2002) endorsed a three-phase approach to the development of this advisory form: 1) identification of management objectives; 2) definition of acceptable risk and/or identification of appropriate reference points (when possible); and 3) development of a monitoring programme. ACE further identified three considerations for defining management objectives for maintaining intra-specific genetic diversity. These were the genetic diversity within and between populations and the population structure, that is, the preservation of the paths of gene flow (cf. Smedbol *et al.*, 2002). For each of these considerations, example management objectives were provided (Table 13.1.1).

Table 13.1.1. Examples of management objectives to address generic concerns related to the loss of genetic diversity in marine species (ICES, 2002).

Consideration	Example management objective
1. Genetic diversity among populations	1. Maintain number of populations
2. Population structure and relative abundance	2. Maintain relative size of populations
3. Within-population genetic diversity	3.1 Maintain large abundance of individual populations
	3.2 Minimize fisheries-induced selection

ACE suggested four general measures to mitigate against the loss of genetic diversity (ICES, 2002):

1) Fishing mortality should be kept sufficiently low to maintain large populations.
2) From a genetic perspective, the harvest should be widely distributed geographically and among all the recruited populations, so that the risk of local depletion and fragmentation of a population and selective removal or modification of particular traits is kept low.
3) Genetic considerations would usually favour an overall reduction of fishing effort over alternative management approaches that result in fisheries becoming even more selective on only parts of a population, either spatially or by some life characteristics.
4) The alternative management options have to be evaluated on a case-specific basis. For example, in establishing closed areas to protect a stock from overfishing, it often could be concluded that the benefits of protecting at least a part of a population exposed to fishing may outweigh the risks of reducing genetic diversity in the part of the population still exposed to fishing.

These were suggested as “common sense” approaches for managers to follow until the scientific community could recommend a more rigorous framework. In 2003, the Working Group on Ecosystem Effects of Fishing Activities (WGECO) and the Working Group on the Application of Genetics in Fisheries and Mariculture (WGAGFM) considered these common sense approaches and deliberated further on the establishment of management objectives and reference points within the ICES context.

13.1.1 Why preserve genetic diversity?

Genetic diversity is the product of thousands of years of evolution, yet irreplaceable losses can occur very quickly (cf. Nielsen and Kenchington, 2001; Kenchington, 2003). This diversity is important for the long-term ability of a species to adapt to extrinsic factors such as pollution or climate change, and loss of populations (extirpation) most likely equates with a loss of adaptive variation. Yet, management units are often discordant with population structure. For example, in the blue whiting (*Micromesistius poutassou*), the main oceanic distribution is considered to represent a single stock and is managed accordingly. Population genetic studies have, however, indicated that partially separated stocks exist in the Mediterranean and in the Northeast Atlantic (Mork and Giæver, 1993; Giæver and Stien, 1998). Similarly, Ruzzante *et al.* (2001) reported on the decadal stability of the genetic differentiation of five cod (*Gadus morhua*) spawning banks off Newfoundland and Labrador, Canada. This genetic structure persisted through the recent population collapse, with only some suggestion of post-collapse mixing between two of the spawning banks. This information is critical to recovery management as it indicates that population re-growth will be the mechanism for rebuilding the stocks, as opposed to migration from other areas. Pragmatically, genetic diversity is also very important for aquaculture, providing the raw material for selective breeding programmes and revitalization of inbred broodstock.

13.2 Review of management objectives

The review of the management objectives leads to the consensus that there would be many different management objectives for the different threats to genetic diversity and the species affected. The example management objectives listed in Table 13.1.1 are

Table 13.2.1.1. Categories of marine organisms which have differing threats to genetic diversity (drawn from Nielsen and Kenchington, 2001).

Classification	Defining characteristics	Examples
Classic marine species	Large population size; high fecundity; pelagic larvae; wide distributions	Mackerel, herring, cod
Benthic/sessile invertebrates	Limited adult mobility; broadcast spawning	Scallops, mussels, coral
Apex species	Slow growth; long-lived; low reproductive potential; large size and/or late age at maturity; restricted dispersal	Sharks, rays, marlin, whales
Localized species	Restricted range; island habitats; or broad range with limited dispersal	Coral, whelks
Hermaphrodite species	Sex change (protoandrous or protogynous)	Groupers, snappers, shrimp

extended modestly to include an objective for the preservation of genetic diversity among populations: *Maximize the amount of diversity maintained when prioritizing populations*. This objective is a development of the "common sense" ACE (ICES, 2002) recommendation (4) summarized above. Managers should consult the prioritization scheme proposed by Nielsen and Kenchington (2001) to assist with the decisions on which populations to protect.

ACE wishes to draw attention to the importance of the third recommendation. To rebuild overfished stocks, much effort is now put into the development of technical measures to make fishing gears more size-selective. In doing this, potentially negative genetic effects must be accounted for. If these problems are disregarded, the fishery may inflict serious damage on genotypes with fast growth, which is in direct conflict with the long-term interest even of the industry.

13.2.1 The application of management objectives to different types of organisms

Marine organisms have a wide range of intraspecific genetic complexity, and biological and ecological characteristics. Nielsen and Kenchington (2001) grouped marine organisms into five categories based on life-history characteristics and population dynamics, which are important to the conservation of genetic diversity (Table 13.2.1.1). These categories are not mutually exclusive, however, most organisms reviewed fall into only one classification. There is some degree of overlap between classic marine organisms and benthic/sessile invertebrates; the latter classification is distinguished from the former because of their sedentary nature. They are not able to relocate in response to habitat disturbance or degradation. For dioecious broadcast spawners (organisms, having separate sexes, that release gametes directly into the sea for external fertilization) who are also sedentary, small-scale spatial structure (nearest-neighbour distances) becomes critical to spawning success (e.g., scallops).

Nielsen and Kenchington (2001) provided a detailed description of these classifications and also put the threats to genetic diversity into context for each group. One threat which is not intuitive is the threat to inter-population genetic diversity in classic marine organisms. Species within this group have generally been regarded as "safe" in a classical conservation genetic context. Beverton (1990) reviewed the well-documented population crashes for ten species of small pelagic marine fish. He demonstrated that even in the case of Icelandic spring-spawning herring, which had the worst population crash, the lowest census size in the time series was estimated at more than one million individuals. However, the use of new genetic markers has challenged this conventional wisdom in some cases by identifying substantive population structuring (cf. Ruzzante *et al.*, 1999), although the within-population genetic variance remains high.

13.2.2 The primary genetic concerns for different types of marine organisms

The life histories and ecology of different types of marine organisms result in different population structures. In turn, the threats to genetic diversity are different for each group. Of course, these are generalizations and ACE emphasizes that case-specific evaluations must be made and endorses the prioritization scheme put forward by Nielsen and Kenchington (2001) to assist in decision-making. Clearly, factors such as overall species abundance are critical in determining the relevant genetic concerns and options. However, Table 13.2.2.1 identifies the generic concerns that are likely to be the most important to the management of genetic diversity.

13.3 Reference points

A limit reference point for individual population size was established by ACE in its 2002 report. This was based on theoretical calculations of successful breeding population sizes required for long-term population viability (Lynch and Lande, 1998). WGECO was able to suggest an additional limit reference point for the management objective: *Maintain number of populations*. Here, the target would be to *Maintain all populations*, but a limit reference point could be to *Maintain all Evolutionary Significant Units* (ESUs *sensu* Waples, 1985). The concept of ESUs is drawn from the conservation biology literature and is a means of preserving evolutionary units above the level of population.

Table 13.2.2.1. Generic *prima fascie* genetic concerns for divergent groups of marine organisms based on a review paper by Nielsen and Kenchington (2001). Entries are in order of priority.

Classic marine species	Benthic/sessile invertebrates	Apex species	Localized species	Hermaphrodite species
Inter-population variation Directional selection within populations	Inter-population variation Preservation of small-scale spatial structure	Population size Inter-population variation	Population size Inter-population variation	Population size Inter-population variation

13.4 References

Beverton, R.H.J. 1990. Small marine pelagic fish and the threat of fishing; are they endangered? Journal of Fish Biology, 37(suppl. A): 5–16.

Giæver, M., and Stien, J. 1998. Population genetic substructure in blue whiting based on allozyme data. Journal of Fish Biology, 52(4): 782–795.

ICES. 2002. Report of the ICES Advisory Committee on Ecosystems, 2002. ICES Cooperative Research Report, 254: 54–59.

Kenchington, E. 2003. The effects of fishing on species and genetic diversity. *In* Responsible fisheries in the marine ecosystem, Chapter 14. Ed. by M. Sinclair and G. Valdimarson. CAB International.

Lynch, M., and Lande, R. 1998. The critical effective size for a genetically secure population. Animal Conservation, 1: 70–72.

Mork, J., and Giæver, M. 1993. The genetic population structure of the blue whiting (*Micromesistius poutassou*). ICES CM 1993/H:5.

Nielsen, E.E., and Kenchington, E. 2001. Prioritising marine fish and shellfish populations for conservation: A useful concept? Fish and Fisheries, 2: 328–343.

Ruzzante, D.E., Taggart, C.T., and Cook, D. 1999. A review of the evidence for genetic structure of cod (*Gadus morhua*) populations in the Northwest Atlantic and population affinities of larval cod off Newfoundland and the Gulf of St. Lawrence. Fisheries Research, 43: 79–97.

Ruzzante, D.E., Taggart, C.T., Doyle, R.W., and Cook, D. 2001. Stability in the historical pattern of genetic structure of Newfoundland cod (*Gadus morhua*) despite the catastrophic decline in population size from 1964 to 1994. Conservation Genetics, 2: 257–269.

Smedbol, R.K., McPherson, A.A., Kenchington, E., and Hansen, M.M. 2002. Metapopulations in the marine fish literature: the use and misuse. Fish and Fisheries, 3: 20–25.

Waples, R.S. 1985. Evolutionary significant units and the conservation of biological diversity under the endangered species act. American Fisheries Society Symposium, 17: 8–27.

14 CONSIDERATION OF ECOLOGICAL DEPENDENCE IN FISHERIES MANAGEMENT ADVICE

Request

The European Commission, Directorate General for Fisheries, has expressed (in a letter of 20 September 2001) its immediate interest in a "*Consideration of ecological dependence in management advice, firstly addressing the groups of species with the ecological linkages that are known with high reliability to have strong ecological linkages*".

Source of information

The 2003 Report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO) (ICES CM 2002/ ACE:03).

Summary

Ecological dependence may need to be considered in the management advice for more fish stocks, but there is no consistent approach for assessing when it needs to be considered. Managers and assessment scientists would benefit from a clear set of guidelines (but not rigid rules) for assessing the strength of ecological dependence, and these guidelines would be used to identify when reference points should be adjusted to account for ecological dependence.

Stocks for which ecological dependence is considered in management advice have not, to date, been identified on a systematic basis. Rather, management strategies that take account of ecological dependence have been adopted because various research techniques have clearly shown that ecological dependence has a significant effect on the dynamics of the target stock, their prey or predators. The stocks for which ecological dependence has already been considered in management advice are: Barents Sea capelin, sandeel in the Shetland area, and sandeel in Sub-area IV. There are also a number of stocks for which ecological dependence may need to be considered in management advice. These are typically "forage fish" stocks for which quantitative assessments may or may not be available and which, on the basis of existing observations on the distribution and abundance of associated predators and (in some cases) their diets, may have ecologically dependent predators. These stocks are: capelin in the Iceland-East Greenland-Jan Meyen area, sandeel in Division IIIa, Norway pout in Sub-area IV and Division IIIa, sandeel in Sub-area IV, Norway pout in Division VIa, and sandeel in Division VIa.

ICES is not in a position to develop a full set of guidelines for assessing ecological dependence. The current approaches for assessing ecological dependence cannot be widely applied and further fundamental research is needed to develop an appropriate method for assessing and ranking the strength of ecological dependence. Classic approaches based on correlating the abundance of predator and prey species, or looking at how the energy content and growth of predators changes with prey abundance were reviewed, but such approaches do not necessarily provide a continuum against which the likely extent of ecological dependence can be judged, and the research resources required to apply these methods in many ecosystems are likely to be prohibitive. An example of one potential approach for ranking the likely strength of ecological dependence in North Atlantic ecosystems is provided here.

Recommendations and advice

ICES advises that:

1) The development of a set of guidelines for assessing ecological dependence is highly desirable.
2) The data required to assess ecological dependence are lacking at this time. Consideration should, therefore, be given to the establishment of an appropriate programme of research to fill this information gap.
3) In the absence of a standard set of guidelines, the present approach, with individual stock assessment working groups making the decision as to whether ecological dependence is important, should continue.

Scientific background

14.1 Ecological dependence

Following ICES (2002b), WGECO focused on the effect of exploiting forage fishes on their predators (as this was the focus of the EC request) even though this is only one of many potential ecological interactions that occur in marine ecosystems.

ICES (2002a) identified circumstances where knowledge of ecological dependence currently had a low, modest or significant impact on the advice and where ecological dependence was already considered in management advice. The stocks for which ecological dependence was already considered in management advice are: Barents Sea capelin, sandeel in the Shetland area, and sandeel in Sub-area IV, and those stocks for which ecological dependence has been proven and quantified and for which Multispecies Virtual Population Analysis (MSVPA) is used in stock assessment.

For example, capelin in the Norwegian-Barents Sea ecosystem is managed with a target escapement strategy. The harvest control rule allows (with 95% probability) the spawning stock biomass (SSB) to be above B_{lim}, taking account of expected predation by cod. ACFM has also noted that the negative influence of herring on capelin recruitment should be included in the B_{lim} rule if such a relationship can be described quantitatively. ACFM also note that adjustments to the harvest control rule should be investigated further to take the uncertainty in the predicted amount of spawners and the role of capelin as a prey item into account (ICES, 2001).

Stocks for which ecological dependence is considered in management advice have not, to date, been identified on a systematic basis. Rather, management strategies that take account of ecological dependence have been adopted because various research techniques have clearly shown that ecological dependence has a significant effect on the dynamics of the target stock, their prey or predators.

Demonstrating that an ecological link exists does not clarify how advice should take account of the link. The effect of ecological dependence on advice can legitimately vary from minor to dominant, and some guidelines are needed for ensuring that ecological dependence receives the proper weight in each case. At present, these guidelines are not explicit and are generally based on the opinions of those conducting specific stock assessments, as informed by a range of scientific research.

The absence of a systematic basis for identifying ecological dependence is not necessarily disadvantageous, since ecological dependence may reasonably be identified using many research methods including diet and bioenergetic studies, analyses of abundance and recruitment trends, models of predator-prey interactions, and behavioural studies (e.g., Vader *et al.*, 1990; Nakken, 1994; Stillman *et al.*, 1996, 2001). However, it is recognized that the application of assessments that account for ecological dependence is not consistent and it would be helpful to have some relatively straightforward approaches for identifying when ecological dependence may be a concern.

14.2 When are considerations of ecological dependence required?

ICES (2002a) previously identified a number of stocks for which ecological dependence may need to be considered in management advice. These are typically "forage fish" stocks for which quantitative assessments may or may not be available and for which, on the basis of existing observations on the distribution and abundance of associated predators and (in some cases) their diets, may have ecologically dependent predators. The stocks are:

Capelin in the Iceland-East Greenland-Jan Meyen area;

Sandeel in Division IIIa;

Norway pout in Sub-area IV and Division IIIa;

Sandeel in Sub-area IV;

Norway pout in Division VIa;

Sandeel in Division VIa.

However, it should be emphasized that demonstrations of strong ecological dependence are attractive to scientists, referees, journals, and the fishing press. Studies that fail to show strong ecological dependence are often regarded with less interest, even if they are well conducted and presented. Therefore, the evidence for the effects of strong ecological interactions on some stocks should not be taken as evidence that they are necessarily a concern to managers of all stocks. For this reason, a simple procedure to rank the likely strengths of ecological dependence between species or small groups of species in a range of ecosystems would be very useful.

The probability of quantifying the strength of a link will depend on the sources of available data on abundance, diet and functional response, and the power of the analytical and sampling procedures. Existing studies, such as the MSVPA analyses, suggest that estimating the strength of ecological links will not be realistic for most interactions that are identified.

A complicating factor is the transience of most predator-prey relationships in space and time and the influence of environmental factors on these relationships. Strong interactions over very short periods may be difficult to detect but will have a key influence on predator and prey dynamics. Moreover, even if strong interactions are accounted for, many other interactions can still complicate predictions. For example, recent analyses have shown the importance of large numbers of weak and largely unpredictable interactions in governing population dynamics (McCann *et al.*, 1998).

14.3 Approaches for assessing ecological dependence

ICES (2002a) concluded that managers and assessment scientists would benefit from a clear set of guidelines (but not rigid rules) for assessing the strength of ecological dependence. These guidelines are required to identify when reference points should be adjusted to account for ecological dependence. ICES was not in a position to develop a full set of guidelines at that stage, and was unwilling to offer a partial and untested set, for fear that they might make practice worse rather than better.

Previously, ICES had recommended that, to take ecological dependence into account when formulating management advice, the following stages could be followed:

1) to identify existing and potential links: principally using diet analysis and a range of ecosystem models;
2) to determine the strength of these links.

14.3.1 Classic approaches

Classic approaches based on correlating the abundance of predator and prey species, or looking at how energy content and growth of predators change with prey abundance are not covered here, as they were reviewed by ICES (2002b). Such approaches are easily applicable, but do not necessarily provide a continuum against which the likely extent of ecological dependence can be judged. Moreover, the research resources required to apply these methods in many ecosystems are likely to exceed those available. A procedure that provides a ranking of the relative probability of observing strong ecological dependence in a range of ecosystems would help to provide a more objective basis for making such decisions.

14.3.2 New approaches

There are many possible ways of investigating the strength of ecological dependence. In 2003, WGECO presented one example of a new method that may have relatively wide application in North Atlantic ecosystems where the principal interest is in assessing the strength of ecological interactions among fish species. The method is applicable to size-based food webs and, in particular, to the part of the size spectrum in which there is a relatively constant body mass ratio between predators and their prey. For this reason, the method is not applicable for interactions involving large zooplankton-feeding species such as basking sharks that "feed down the food web" on small prey. It is also unlikely to be appropriate for dealing with issues of ecological dependence that relate to marine mammals or birds, which have become an important issue in many ecosystems.

It was emphasized that all possible methods for usefully categorizing the expected strength of ecological dependence between fishes in a range of different ecosystems will require new data. This conclusion is based on existing knowledge of available data on the structure of marine food webs and the data requirements of methods such as MSVPA that have previously been applied to some ecosystems.

While perspectives thus exist to categorize ecological dependence, it would be premature to provide further advice at this stage.

14.4 References

ICES. 2001. Report of the ICES Advisory Committee on Fishery Management, 2000. ICES Cooperative Research Report, 246 (Parts 1–3).

ICES. 2002a. Report of the ICES Advisory Committee on Ecosystems, 2002. ICES Cooperative Research Report, 254: 49–53.

ICES. 2002b. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 2002/ACE:03.

McCann, K., Hastings, A., and Huxel, G.R. 1998. Weak trophic interactions and the balance of nature. Nature, 395: 794–798.

Nakken, O. 1994. Causes of trends and fluctuations in the Arctic-Norwegian cod stock. ICES Marine Science Symposia, 198: 212–228.

Vader, W., Barrett, R.T., Erikstag, K.E., and Strann, K.-B. 1990. Differential responses of common and thick-billed murres to a crash in the capelin stock in the southern Barents Sea. Studies in Avian Biology, 14: 175–180.

Stillman, R.A., Goss-Custard, J.D., McGrorty, S., West, A.D., dit-Durell, S.E.A., Clarke, R.T., Caldow, R.W.G., Norris, K.J., Johnstone, I.G., Ens, B.J., Bunskoeke, E.J., Merwe, A.v.-d., van der Meer, J., Triplet, P., Odoni, N., Swinfen, R., and Cayford, J.T. 1996. Models of shellfish populations and shorebirds. Report to Commission of the European Communities by the Institute of Terrestrial Ecology, Furzebrook.

Stillman, R.A., Goss-Custard, J.D., West, D.A., Durell, S.E.A., McGrorty, S., Caldow, R.W.G., Norris, K.J., Johnstone, I.G., Ens, B.J., van der Meer, J., and Triplet, P. 2001. Predicting shorebird mortality and population size under different regimes of shellfishery management. Journal of Applied Ecology, 58: 857–868.

15 FRAMEWORK FOR THE PROVISION OF INTEGRATED ADVICE

Request

This is an ICES issue, to continue development of a framework for the provision of integrated ecosystem advice within ICES, and consider how this could be operationalized in the near future.

Source of information

The 2001 and 2003 Reports of the Working Group on Ecosystem Effects of Fishing Activities (WGECO) (ICES CM 2001/ACME:08; ICES CM 2003/ACE:05).

The ICES Strategic Plan and Integrated Action Plan.

Report of the EC Stakeholders Conference "Towards a Strategy to Protect the Marine Environment" (Køge, Denmark, 4–6 December 2002).

The 2003 Report of the Study Group on ACFM, ACE, and ACME, and Working Group Protocols (SGAWWP) (ICES CM 2003/MCAP:02).

The 2003 Report of the Study Group on Precautionary Reference Points for Advice on Fishery Management (SGPRP) (ICES CM 2003/ACFM:15).

The 2003 Report of the Planning Group on the Implementation of the Baltic Sea Regional Project (PGIBSRP) (ICES CM 2003/H:05).

The 2003 Report of the Regional Ecosystem Study Group for the North Sea (REGNS) (ICES CM 2003/ ACE:04).

Report of a Joint Workshop (EEA-EC DG Fisheries-DG Environment) on "Tools for measuring (integrated) Fisheries Policy aiming at a sustainable ecosystem", Brussels, 28–29 October 2002.

Summary

This is a discussion document, and presents ideas that ICES needs to consider internally and discuss with partners, and then take actions to move forward. It is not the final word of ACE on the provision of ecosystem advice, but is intended to provoke discussions and actions that increase the ability of ICES to serve that role.

Recommendations and advice

These are recommendations to ICES. Some will require work with partners in ICES science and clients for ICES advice, but in the first instance the recommended actions are directed at ICES, not at its partners and clients.

ICES should:

1) Continue dialogue with client Commissions and other agencies and organizations interested in more integrated ecosystem-based management, and the scientific advice needed to support such management. Good dialogue will increase the likelihood that thinking and practice on this topic will progress in compatible ways within the scientific and management communities, and between ICES and its clients.

2) Continue to respond with timely and high quality scientific advice to OSPAR and other clients with regard to making the EcoQ/EcoQ element/EcoQObjective framework fully operational. The goal here is to support organizations that are trying to take a more ecosystem-based approach to their tasks, through the provision of sound, concrete, natural scientific advice on specific practices that can be implemented in the short or medium term, and that will yield concrete benefits. Such ICES activities will help organizations actually change management practices and see the positive effects of a more integrated ecosystem approach. The framework for proceeding with this has largely been developed and tested by WGECO, although much remains to be done in putting specific considerations within that framework.

3) ACFM, ACE, and the Resource Management Committee (RMC) should review the contributions of the Study Group on Incorporation of Process Information into Stock Recruitment Models (SGPRISM) and the Study Group on Growth, Maturity and Condition in Stock Projections (SGGROMAT), with regard to the effectiveness of their recommendations for placing fisheries advice into a larger ecological context. If these concrete proposals for assessment practice are found to be constructive steps ahead, ACFM, ACE, and RMC should:
   - Set out a work plan and timetable to enable the accepted products of these Study Groups to become routine practice in fish stock assessment;
   - Identify and rank in priority opportunities for other focused and topic-specific Study Groups to attack similar individual components of current practices in fisheries (or other topic) assessment and the provision of advice.

4) The Living Resources, Oceanography, Resource Management, and Marine Habitat Committees should have some time at the 2003 or 2004 Annual Science Conference (ASC) to discuss the possibilities for more integrated monitoring. Several documents, particularly those associated with EuroGOOS, already exist, and could form a basis for such a discussion. The goal would be to

maximize opportunities for building on existing activities, while filling important gaps in the necessary monitoring for integrated assessments.

5) An expert group should be appointed to address the specific question of what comprises "serious or irreversible harm" on the scale of management in an integrated ecosystem context. The Precautionary Approach (PA) framework within ICES fisheries advice has been challenging to develop. Consistency and acceptance among clients has required that ICES (and fisheries organizations more generally) pay close attention to the nearly universal reference in documents citing the PA or Precautionary Principle (PP) that the application of precaution is only warranted in cases where what is at risk is "serious or irreversible harm", not just to avoid doing something that a group of specialists does not like. The goal of this activity would be for ICES and its clients to commence development of an operational interpretation of "precaution" (as distinct from simply good risk management), to allow progress on this topic as well.

6) ACE, ACME, ACFM, the Management Committee on the Advisory Process (MCAP), and RMC should create terms of reference and identify appropriate membership for a Workshop or Study Group to consider and, to the extent possible, to investigate with practical examples, the implications of moving to a more integrated ecosystem approach on the consistency and timeliness of ICES advice on fisheries and other issues. Many ICES reports assume that moving to more integrated regional assessments will improve the basis for ICES advice. However to this point, no one has analysed whether these two initiatives (improved timeliness and consistency of advice and advice based on regional integrated assessments) would naturally move ICES in the same direction and hence be relatively easy to achieve simultaneously, or if they might necessarily be incompatible, and hence require even more effort to achieve together. The goal here is to build a better understanding within ICES, and with ICES clients, with regard to what changes are to be expected in ICES advice as ICES moves in this direction. The template for ICES advice on fisheries management (and other topics) could also be reviewed to ensure that the advisory templates can accommodate the types of information that ICES might be providing in a more integrated ecosystem context.

7) The Consultative Committee and MCAP should develop terms of reference for a subsidiary group to consider the minimum participation in numbers, geographical distribution, and disciplines, in order for a Regional Assessment group to proceed effectively. Proposals for integrated assessments which are "designed to integrate existing effort, not duplicate it or create unnecessary new effort" are welcome, but even the necessary new effort may be substantial, and should be estimated to ensure that the work will be able to be conducted thoroughly and effectively.

8) In situations where ICES enters into a dialogue with clients about specific non-standard advisory requests, ICES should be alert for well-chosen opportunities to include in the Memorandum of Understanding or formal request negotiated with the client, an acknowledgement that the advice to be provided will give appropriate attention to the broader ecosystem context. The goal is to provide some of the "living examples" of how scientific advice in a more integrated ecosystem context is the product managers and other clients should always be seeking. Recovery planning of collapsed fish stocks and evaluating the ecosystem impacts of aquaculture are two examples of good candidates for such dialogue.

9) Notwithstanding 7), above, ICES should be vigilant not to undertake to provide "expert advice" on issues of governance and social sciences, where ICES does not have such expertise. The goal is not to deny that governance and social sciences are central to a more integrated ecosystem approach, but rather to ensure that clients know that, with current expertise and operational characteristics, ICES is not able to draw on any special expertise in these fields.

10) ACE recommends that:

A Theme Session covering "Integrated Assessments" should be held at the 2005 Annual Science Conference. This would be timely in respect of the OSPAR intermediate quality status assessment in 2005 and other specific OSPAR thematic assessment needs.

Scientific background

This is a discussion document that presents ideas that ICES needs to consider internally and discuss with partners, and then take actions to move forward. It is not the final word of ACE on the provision of ecosystem advice, but is intended to provoke actions that increase the ability of ICES to serve that role.

This is a self-generated term of reference, and has been addressed by ACE during each of its meetings. Many national and international jurisdictions with management responsibilities for aspects of marine environments or their uses also have sponsored meetings or initiatives to advance the ability to provide integrated management advice. A number of these initiatives were reviewed in ICES (2002). Since that review, meetings have continued to address the issue; within the ICES area, most notably the Stakeholders Conference "Towards a Strategy to Protect the Marine Environment" at Køge, and initial meetings of ICES regional ecosystem study groups for the North Sea (REGNS) and the Baltic Sea (PGIBSRP).

Each of these initiatives builds on the text of preceding reports, but the outputs of these initiatives are still generally characterized by generalities and conceptual terms. As recently as December 2002, the Stakeholders Conference in Køge concluded that there is STILL a "need to complete the task of interpreting, in clear and unambiguous terms, what the 'the concept of an integrated ecosystem approach' means, and how it is applied in practice". It is time to move from conceptual language to very specific points. To move ICES forward, this section undertakes four tasks:

1) From the WGECO report, it identifies commonalities among the many discussions in the literature and meeting reports, with regard to key features of "integrated ecosystem advice".
2) It considers which of those common features in (1) lie within the expertise and purview of ICES and its scientific and advisory competences. Where components commonly held to be part of an integrated ecosystem approach are or could be within ICES competence, they are reviewed critically to identify specific enabling activities. Where they are not within ICES competence, suggestions are offered for where actions to address the needed components would be most appropriate.
3) It consolidates and reviews the conclusions of two new groups created in 2002 to initiate coordinated ecosystem studies and, eventually, assessments in the North Sea (REGNS) and the Baltic Sea (PGIBSRP).
4) It offers specific activities that ICES could undertake in the short term, to promote progress towards providing the scientific and advisory basis for more integrated ecosystem approaches to management of the uses of marine ecosystems.

15.1 Working Group on Ecosystem Effects of Fishing Activities (and its supporting documents)

15.1.1 Common features of discussions of integrated ecosystem advice and management

The reports and initiatives nationally, regionally, and globally that were reviewed by WGECO in 2003 and earlier years, and by ACE in 2002, bring out a number of features of an integrated ecosystem approach that are common across these iniatives. These include:

Inclusive, participatory governance and decision-making, with an informed citizenry is featured in nearly every discussion of integrated ecosystem approaches. Past treatments of the advisory framework by ICES also acknowledge this as an important feature of ecosystem approaches.

It is **human activities that are managed, and the not the ecosystem**. Many decisions are perceived as risk-risk choices among competing uses, not just balancing the intensity of use with protection of the environment. Not only are human activities the ecosystem properties that are managed, but of the impediments and components of the way forward listed in the Køge Stakeholders Conference report, eight of thirteen cannot be addressed without society making value-based choices among competing potential human activities.

Almost every initiative and document gives a **prominent role for the social sciences** in identifying goals, developing management approaches, and evaluating the consequences of management actions. Over half the points in the Køge Stakeholders Conference report that were analysed by WGECO required a moderate or high degree of social science input in order for any meaningful progress to be made.

Specification of higher-order management objectives is required, although these are usually highly conceptual and additional work is needed to make them operational. These characterize nearly every initiative reviewed in ICES (2002), and form the core of the approach endorsed at the Køge Stakeholders Conference.

Indicator-based approaches, often with explicit operational objectives and reference points, are the basis for operationalizing the conceptual objectives. This is particularly prominent in the Bergen Declaration from the Fifth North Sea Conference and associated documents, the approach adopted by the Monitoring and Assessment Group (MONAS) for HELCOM, and the Køge Stakeholders Conference report.

Most proposals stress a **reliance on the Precautionary Approach (PA)** in advice and decision-making.

Advice on single resource uses needs to include consideration of the status of not just the resource being used, but other **ecosystem components interacting with or influencing the resource, and other human activities that affect the resource or interact with the resource use.**

This is most often specified for fisheries, where it is argued that assessments should consider more environmental influences on stock status and dynamics, and advice should be more fleet-based and consider the ecosystem effects of the entire fishery.

Monitoring covers many ecosystem components and is conducted in integrated programmes. Many of the international organizations around the North Sea and Baltic Sea (OSPAR, IBFSC, HELCOM) or more regionally and globally (IOC, SCOR, and other sponsors of GOOS, GLOBEC, etc.) feature this point.

Regional assessments that integrate all major ecosystem components and human activities in the regional seas are conducted and reported periodically. This is given prominence in the Bergen Declaration, many of the Baltic initiatives, and in past treatments of the topic by ICES.

Management that is integrated and adaptive, rather than piecemeal and rigid, is required. This need is acknowledged in the Bergen Declaration and the Køge Stakeholders Conference report. Various organizations and jurisdictions are undertaking discussions both officially and informally with regard to coordinating their management approaches more effectively.

Analysis of common themes from an ICES perspective

Inclusive, participatory governance and decision-making. ICES is not a body with expertise in governance structures and processes. As a scientific advisory body, ICES is positioned to inform governance and decision-making bodies, whatever their nature. However, its disciplines of strength give ICES no particular legitimacy in advising on one form of governance over another. Rather, through maintaining credibility, relevance, and timeliness of scientific information and advice, ICES should be the primary source that all groups come to for their scientific advice. Therefore, the ICES role in discussions of governance structures is simply to keep the lines of communication open, and ensure that the evolving governance systems appreciate the need for scientific support, and look to ICES as the best source for that support.

It is human activities that are managed, and not the ecosystem. The ICES strength has traditionally been in understanding the structure and function of marine ecosystems, and the impacts of human activities on those ecosystems and their components. Science to understand the management of human activities has played a much smaller role in ICES science programmes. Examples of such initiatives exist, largely in subsidiary groups reporting to the Resource Management Committee. ICES has not developed strong expertise in the area of actually advising on how human activities should be managed, despite ICES expertise in advising on ecological and environmental aspects of management.

Prominent role for the social sciences. Despite several theme sessions at Annual Science Conferences in the 1990s on the opportunities for and values of linkages with the social sciences, aside from the Study Group on Fisheries Systems ICES has made almost no such linkages. ICES lacks expertise in this area, and the ICES Strategic Plan and Action Plan do not include provisions to develop such expertise in the medium term. The Social and Economic Module of the Baltic Sea Regional Project is structured around these considerations, but the linkage of this module to ICES is less clear than the linkage of the modules on Productivity, Pollution and Ecosystem Health, and Fish and Fisheries.

The current limited capacity in social sciences puts important boundaries on the ICES role in the provision of advice to support integrated ecosystem approaches to management. Of the fourteen impediments and challenges to progress on implementing an ecosystem approach that were identified at the Køge Stakeholders Conference, half of them required high or moderate involvement of social scientists in order to address the impediment or challenge. The ICES role in the provision of information and advice on progressing towards a more integrated ecosystem approach to the management of human activities in marine ecosystems will be primarily in diagnosing what needs to be done to improve ecosystem status and sustainability of uses, and developing and evaluating tools to do that task better. ICES will have little expert role in advising on how to manage human activities to deliver the needed improvements.

Specification of higher-order management objectives. ICES has significant technical expertise in advising on conceptual, higher-order management objectives, and has served this function many times in the past, whether in response to external requests for advice or self-motivated. Where needed, ICES can continue to provide support for this activity with existing expertise and structures, although requests for such support are unlikely to be common. Most organizations seem to have little difficulty in setting higher-level ecological, social, and economic objectives.

Indicator-based approaches. The commitments to indicator-based approaches in the Bergen Declaration and the Køge Stakeholders Conference report are very important. The approach buys wholly into the framework proposed in the past by ICES (2002) for an objective/indicator/reference point approach to more integrated ecosystem management of human activities. The adoption of this approach means that progress can be incremental, building on current practice and experience in each scientific discipline. It also means that the progress towards a more integrated ecosystem approach in all cases will be science based, using as much or as little scientific capacity as exists. Key science roles will include testing the information content and reliability of indicators for conceptual objectives, refining conceptual objectives into operational objectives, and identifying suitable positions for limits on the indicators. Section 6 of this report provides concrete examples of exactly how science will contribute to continued progress towards a more integrated ecosystem approach to management.

ICES has important strengths in providing the scientific support for setting operational ecosystem objectives, selecting appropriate indicators, and estimating appropriate conservation reference points. The explicit mention in the Bergen Declaration of ICES as the source of such support, and the OSPAR request for advice addressed in Section 6 of this report, illustrate both our stature and our ability to serve this function professionally. The reports of WGECO in 1999 through the present all address directly the issue of what should comprise operational objectives for integrated ecosystem approaches to fisheries, and the reports of the Working

Group on Seabird Ecology (WGSE), the Working Group on Marine Mammal Ecology (WGMME), and other groups do similar but more case-specific jobs for other activities. ICES is also serving a very important function by identifying the limitations on uses that can be made of individual indicators, and on indicator-based approaches in general.

ICES is doing the right things here, and generally doing them in the right way. The ICES future is likely to include:

1) assessing and reporting the status against indicators for many ecosystem components on a regular basis (as ACFM does now for the spawning stock biomass (SSB) and fishing mortality (F) of many harvested fish stocks);
2) identifying new indicators and reference points where they are needed; and
3) advising on the cautions and limitations that are appropriate when interpreting or making decisions based on indicators.

Nonetheless, although the framework for selecting and evaluating indicators is developed moderately well, ICES is still in the testing phase for the framework. ICES is learning a lot from the testing, and finding many ways that the framework needs to be adapted and refined. It is far from mature.

ICES should not be complacent here, however. There have been some challenges in reaching this position, including difficulties with clients in framing scientifically clear and tractable requests for advice on appropriate indicators and objectives. Also, ICES has no monopoly on scientific expertise in this area. ICES clearly has the expertise and competence to provide leadership and the necessary support to managers in the tasks of identifying appropriate indicators and reference points, evaluating status on them, and reporting status relative to management reference points. However, ICES must continually ensure that it keeps the necessary scientific rigour and diversity of expertise in its contributions to demonstrate that it is, in fact, the most authoritative science voice on these issues.

Reliance on the Precautionary Approach. ICES has provided leadership in the application of the precautionary approach in advice on fisheries, using limit and precautionary reference points for biomass and fishing mortality as the basis for quantitative harvest advice for most fish stocks. However, even in these single-stock, two-indicator applications, it has proved challenging to keep the application consistent across stocks and current with changing assessments (ICES, 2003). Ways to set comparable reference points for indicators of ecosystem status have been discussed in several working groups. However, the challenge of developing a consistent framework for the application of precaution in more integrated ecosystem advice has not yet been tackled, by ICES or by other groups. It will not be simple to define objectively what comprises "serious or irreversible harm" for many of the more integrative ecosystem properties, and even more difficult to apply a consistent standard for such harm across many different attributes. There is an opportunity for ICES to display global leadership on this topic, but based on the experience with fisheries advice, the task will require a significant amount of time and effort.

Advice on single resource uses needs to include additional ecosystem components, and other human activities. The goal features prominently in the ICES Strategic Plan and supporting Action Plan, and in the Report of the Study Group on ACFM, ACE, and ACME and Working Group Protocols (SGAWWP). ICES has already demonstrated strengths in assessing the impacts of fishing on marine ecosystems (the work of WGECO), and a number of Study Groups as well as the GLOBEC Office are doing important work on assessing the impact of the environment on fish populations. ACE is building capacity and acquiring experience at providing somewhat more integrated advice.

What is absent is a clear roadmap for how to move forward in a concerted manner, rather than simply having study groups proliferate and WGECO's terms of reference become longer every year. The SGAWWP report includes a prominent role for "Integrating Assessment Groups" and acknowledges explicitly the desire to place fisheries advice in an ecosystem context, as well as taking an ecosystem approach in ICES advice generally. However, the report is vague about how this will be done. It notes that even among the Study Group participants "there was a range of views" on the best organization and activities of the ICES groups inputting to the final advisory step.

There are some specific steps available to ICES in the short term that would make at least fisheries advice more inclusive of additional influences on stock dynamics. For example, tools developed in SGPRISM and SGGROMAT could be made part of standard assessment practice. However, such steps are small compared to the expectations arising from the language in the commitments to integrated ecosystem advice.

Put simply, it is clear that ICES is far from having consolidated its own thinking on how to make advice more integrated at the level of either Working Group activities or Advisory Committees. Without specific guidance on what a single advisory committee structure and regional assessment groups would do differently from at present, it would be premature to conclude that such structural changes would necessarily make advice more ecosystem-based. After all, regional assessment groups have been proposed as an alternative to the current ACFM structure, simply to address some fisheries issues better.

There is another issue that ICES must address with regard to the provision of more integrated advice. Although clients of ICES advice have all subscribed to the concept of more integrated, ecosystem advice, they are also calling for advice of ever greater consistency and on faster schedules. Although a formal analysis has not been undertaken by ICES, it seems likely that moving to a more integrated ecosystem approach would be more likely to increase inconsistencies and require more time for generating advice. The parameterized functional models and supporting data that are necessary for assessments to include the effects of environmental covariates and other human activities on stock dynamics will be even less consistently available than are catch and research survey data. Moreover, stocks are likely to react in more diverse ways to these factors than they do to directed harvesting, and methods to include such factors in analyses or interpretations are less consistently codified. All these factors are likely to make integrated ecosystem advice on fisheries less consistent rather than more consistent, even if ICES considers such advice to have a more complete scientific basis. Expertise from more groups will also be needed for input on the path to the provision of the final advice, so more time (and more careful scheduling), rather than less time, will be needed to provide the advice as well.

ICES clearly has in-house work to do to consolidate its own views of the detailed process by which integrated ecosystem advice can be provided, and to sell those views to its own Council. ICES also must undertake serious discussions with clients on nuts-and-bolts implications of providing integrated ecosystem advice, not just higher-level conceptual discussions of whether or not the idea of more integrated ecosystem advice is a good one. Both ICES and its clients need to understand the costs as well as the benefits of moving in this direction, and engage in focused dialogue on specifics.

Monitoring covers many ecosystem components and is conducted in integrated programmes. ICES has a long history of coordinating monitoring in the ICES area. Several subsidiary groups of the Living Resources Committee coordinate fish surveys throughout the Northeast Atlantic, and subsidiary groups of the Marine Habitat and Oceanography Committees identify appropriate monitoring standards and in some cases establish survey protocols for monitoring lower trophic levels, and physical and chemical ecosystem properties. ICES has had a central role in recent initiatives to expand and coordinate monitoring programmes, particularly EuoGOOS and the Baltic Sea Regional Project. Other organizations, such as OSPAR and HELCOM, routinely consult ICES for advice on monitoring needs and standards. Monitoring features clearly in the ICES Action Plan for the near future. It seems that ICES is doing the right things in the right way in this area, and such activities should remain a priority. There are many needs and opportunities for more monitoring programmes on more ecosystem components and in new places. ICES can expect to continue to play a major role in diagnosing the needs for scientific monitoring programmes, planning the activities that would comprise the additional programmes, and coordinating and reviewing progress on biological, physical, and chemical monitoring programmes once in place.

Regional assessments that integrate all major ecosystem components and human activities. Regional assessments have been undertaken by other organizations, and these groups have come to ICES for the primary scientific input on the physical and biological ecosystems. However, in all policy-scale documents the expectation is that "integrated" means that the social and economic aspects of uses of regional marine ecosystems will be part of the assessment.

As noted earlier, ICES does not have expertise in social or economic sciences, so by itself, ICES can never undertake fully integrated assessments of the scope expected by our main clients. As ICES considers creating regional ecosystem assessment groups internally, it needs to consider seriously how the full integration will be achieved. To this point, external clients have been willing to coordinate production of the overall regional ecosystem assessments, with ICES receiving cost reimbursement for the work that if contributes to these initiatives. Even if ICES continues to serve only this role, it faces major challenges.

First of all, if ICES attempts to undertake regional assessments using only its existing strengths, we would produce a product that would not be fully useful to managers. Even if ICES did its job well, clients might perceive such "natural science" assessments as only making their jobs more difficult. Such limited regional assessments could report poor biological conditions, but would not help make managers more effective at doing something about the degrading ecosystems. Effective management actions for healthier environments depend on good knowledge of and planning for the social and economic consequences of management measures, and not just sound scientific evidence that actions are needed. To produce regional assessments of full value to clients, ICES needs to have strong linkages to experts who can review the human practices that are associated with the declines, identify the key social and economic drivers, and evaluate the sustainability of options. This is much more ambitious than just summarizing where and how large the fish harvests were.

Second, even if there were reasons to prepare regional assessments addressing only the biology, physics, and chemistry, they would be very demanding on ICES resources. Attempts to set up regional groups for integrated regional assessments have had trouble attracting an adequate number of participants and/or the right diversity of experts. If ICES attempts integrated regional assessments, and cannot summon enough expertise to do a first-rate job, ICES may be hurting its own credibility. These are issues that all of ICES needs to consider carefully.

Management that is integrated and adaptive. Adaptive management generally means monitoring the consequences of a management alternative that is selected for implementation, evaluating monitoring results against performance standards, and modifying the management approach based on the feedback of its performance. Many current ICES programmes and strengths position it to play an important advisory role in adaptive management. Clients already commonly ask to be advised on the risks and consequences of management alternatives, and not just a single “preferred choice”. ICES is also well prepared to identify scientifically sound performance properties to monitor, and performance standards that would indicate whether the management strategy was succeeding or failing to make progress towards its ecological and environmental goals. Where failures occur, ICES could also advise on the nature of the adaptations to management that would be necessary to correct the failures.

Moving to integrated management would pose more challenges to ICES. Integrated management needs more integrated assessments and a different type of scientific advice. Although ACE and MCAP both exist in large part to make sure that ICES advice is consistent and well integrated, the integration, to the extent that it occurs, is at a very late stage in the process. All the scientific basis for the advice has already been prepared by the subsidiary groups. Moreover, both the time available for ACE work, and the nature of the work of all the advisory committees, means that very little true integration can be done; material from different sources can be put next to each other, but that is not the integrated advice required for truly integrated management. If ICES is to provide scientific advisory support for integrated management, more fundamental changes in approach would be needed at the level of the working and study groups. In more limited tasks, an approach of sequential working or study group meetings to each add value and integration to the products of the preceding ones is proving taxing on ICES science capacity, and the consequences of a group early in the sequence failing to fully discharge its terms of reference amplify through the succeeding groups. This gives cause for caution about the ease with which the existing ICES approach to providing the scientific support for management can move through small evolutionary steps to providing support for integrated management.

If there is a silver lining for ICES in the prospect of moving to integrated management, it is that moving from the current management approaches to integrated management will require the management agencies to undergo some major changes themselves. If there is a true move to integrated management, managers will need a different kind of scientific support, and not everything they are currently obtaining for single species: single input management, plus something more integrated added on top. If ICES were freed from having to support all the advisory needs of the current “un-integrated” management systems, it would have more scope to rearrange its entire approach to supporting the advisory processes. That might make it somewhat more feasible to restructure in ways that could provide the scientific products needed for integrated management more effectively.

15.2 Regional Ecosystem Study Group for the North Sea

15.2.1 General considerations

At its first meeting in 2003, the Regional Ecosystem Study Group for the North Sea (REGNS) considered a variety of national initiatives around the North Sea that were intended to advance a more integrated ecosystem approach to management, and noted the following:

1) There is wide acceptance that the science of ecosystems is under development and in many cases questions relating to ecosystem function and response are unlikely to be answered completely for many years. The challenge facing Member Countries is therefore to make better use of present scientific knowledge to establish the operational scientific tools (models) to support the thematic assessment and management needs.

2) For most Member Countries, existing monitoring programmes demonstrate little integration between the scientific output from R&D programmes and the types of monitoring being undertaken. This is because the present programmes largely reflect compliance against traditional sectoral policy drivers dealing with fisheries, chemical contamination, ocean climate, and nature conservation. However, the ecosystem approach cuts across all these sectors. Without coordination of the respective national and international sectoral monitoring programmes, excessive duplication of effort may result. When one set of monitoring results provides inputs to evaluating the progress of many sectors towards their objectives, some measure of control and coordination is required to ensure equitability of activities between sectors.

3) Currently, the present system of assessment and control of monitoring is very much sectoral based. The need for adaptive management requires not only the monitoring to be integrated, for example, nutrients monitoring should be integrated with operational modelling of ocean processes and the measurement of eutrophication effects, but it also requires the regulatory advice to respond (pro-actively) to any changing pressures and environmental conditions which may give rise to adverse effects.

4) The feedback from the assessments to regulate the inputs and pressures on a time-scale commensurate with mitigating for any effects is essential. The mechanisms by which such feedback can be applied are subject to discussion and agreement, but they will ultimately depend on the type of activity, the location, and the resources available to the relevant competent authority.

The ministers at the Fifth North Sea Conference agreed to implement an ecosystem approach to the management of the North Sea. The ministers invited ICES and GLOBEC to consider the priority science issues and contribute to their development.

In brief, the scientific issues are:

Short-term (3–5 years) Priority Issues:

- Operational fisheries oceanography;
- Habitat mapping (first generation);
- Spawning areas of fish populations;
- Experimental studies of trawling closure areas;
- Threatened and declining species and habitats;
- EcoQOs and indicators.

Longer-term (> 5 years) Priority Issues:

- Role of benthic species richness:
- Ecological transfer efficiencies:
- Population dynamics of key species:
- Food web and life history interactions;
- Transport and effects of contaminants.

The North Sea Ecosystem Science Programme, recommended by the expert conference in Bergen in 2002, could serve as the framework for the implementation and coordination of regional ecosystem R&D. A Regional Programme for the North Sea could be co-sponsored by GLOBEC, whose North Atlantic office is housed in ICES. This research initiative should be open and inclusive, and care should be taken to avoid unnecessary duplication of research activities. ICES could use its machinery with Working Groups and Science Committees in assisting the planning and implementation of the research to address the priority science issues.

15.2.2 Integrated ecosystem assessments

ACE recognizes two types of assessment which underpin the ecosystem approach; these are i) general assessments; and ii) thematic- (or activity-) based assessments. Although thematic and general assessments are closely related, they have different purposes. Thematic assessments embody the adaptive management principle, which requires that activities are managed in a way that is responsive to the dynamics of the ecosystem. In considering general assessments, a combination of activities and their effects on the ecosystem need to be assessed. This requires a greater degree of understanding of ecosystem function and the cause-effect pathways which determine state and impact.

In both types of assessment, the ecosystem is considered as a set of biological compartments that are trophically linked and which interact with their environment, including human pressures. ICES Working Groups map onto the compartments (boxes) of the ecosystem reasonably well. However, the links between ecosystem compartments are not so well reflected in the ICES Working Group structure and this represents a gap in capability that must be addressed, if ICES is to be positioned to conduct integrated ecosystem assessments when these are requested by clients.

In order to move towards supporting both general and thematic assessments required by clients (OSPAR, HELCOM, EC, etc.), ACE recommends that the following three proposals be considered by ICES. These should not be seen in isolation of each other, but should be considered as complementary elements towards delivering a permanent mechanism to address integrated assessments by ICES Working Groups. The elements could be implemented in a step-wise manner. However, the outcome of the first step will determine the remaining steps; subsequent steps (if any) will be shaped entirely by the outcome of the Theme Session on Integrated Assessment at the ASC in 2005 and therefore subsequent steps should be viewed as conditional.

Step 1: 2005 ASC Theme Session on Integrated Assessments

Because the ICES Working Group structure does not lend itself to integrated assessments across the different components of the marine ecosystem or external drivers, or within individual regional ecosystems, a new process is required to facilitate holistic integration of advisory products within ICES. ACE is acutely aware of the heavy operational burden on some ICES Working Groups, especially the fish stock assessment Working Groups. Although the proposal outlined below will involve additional work for specific Working Group members, ACE wishes to stress that:

a) The proposal is designed to integrate existing effort, not duplicate it, or create unnecessary new effort;

b) ICES and its Member Countries will be required to undertake integrated assessments. Therefore, ICES needs to establish a process where value can be added to the exiting assessment work of Member Countries;

c) The proposal should take place over a two-year period in order to allow work planning.

ACE recommends that:

A Theme Session on Integrated Assessments should be included in the 2005 Annual Science Conference. This would be timely in respect of the OSPAR intermediate quality status assessment in 2005 and other specific OSPAR thematic assessment needs, as well as the needs of HELCOM.

Step 2: Regional Integrated Assessment Programmes

Taking advantage of the existing Working Group and Science Committee structure of ICES, Integrated Assessment Programmes could be established to meet specific customer needs. The programmes could be established either on a regional or a thematic basis depending on need, but in all cases would be led by a senior scientist from the ICES community. It is anticipated that programme meetings would be required and that they would be supported by appropriate levels of intersessional correspondence. The programme meetings should be chaired by the programme leaders and attended by the relevant Working Group Chairs. The meetings could be held when the relevant Working Groups have delivered products according to the specific terms of reference requested by the Integrated Assessment Programme objectives.

The advantage of this approach is that one individual is tasked with the responsibility of coordinating the inputs required from existing Working Groups. It represents no structural change to the existing working arrangements but rather adds an additional level of assessment which cuts across Working Group activities. It will significantly increase the workload of Working Group Chairs, and almost certainly will require Working Groups to address additional terms of reference as part of their tasks.

Step 3: Regional Integrated Assessment Groups

This step recognizes the need to formally acknowledge Step 2 by establishing permanent ICES integrated assessment groups. The groups should recognize the activities already in hand by Member Countries to coordinate national monitoring programmes to deliver both thematic and general assessments required by OSPAR (Joint Assessment and Monitoring Programme, JAMP), the EU (Water Framework Directive, WFD), and the Helsinki Commission. This effort in some cases is considerable, but is in general a common activity across Member Countries. ICES could play a value-added role in integrating the national assessments.

This could be undertaken at two levels:

i) for general assessments such as the periodic OSPAR Quality Status Reports (QSRs), Regional Ecosystem Groups (REGs) would be tasked with undertaking integrated assessments on a regional basis by drawing together the relevant national assessment products and, where available, thematic assessments. This would be on a time-scale commensurate with customer (QSR) reporting needs and the groups should be considered as permanent within ICES;

ii) for thematic assessments, which would address specific needs such as eutrophication and contamination, Thematic Ecosystem Groups (TEGs) would be established on an *ad hoc* basis and would consist of members drawn from existing ICES Working Groups that would enable the integration of the science needed to support the assessment needs. Common terms of reference for Working Groups which identify forthcoming thematic assessments would ensure that value is added to the work of TEGs when they are convened. These groups would, in general, be temporary.

15.2.3 Integrated monitoring

Most, if not all, the mandatory monitoring programmes undertaken at the national level by Member Countries can be placed into one of four sectors: i) fisheries; ii) contamination/pollution (including physical disturbance); iii) species and habitats; and iv) ocean climate and processes. Each Member Country has different mechanisms for managing the programmes within each sector, but in general the sectors and programmes have evolved in response to specific policy and legislative drivers laid down by international conventions and the European Commission, and require specific assessment products. Adopting an ecosystem approach will inevitably require the integration of sectoral-based monitoring programmes in order to avoid unnecessary duplication of effort.

ACE recognizes that the integration of monitoring programmes will need to occur at a number of institutional, national, and international levels, namely: i) harmonizing the monitoring effort between sectors to obtain the best use of resources; ii) recognizing the need to integrate within each sector; and iii) whilst i) and ii) enhance the thematic assessment needs, there is also the need to integrate the monitoring programmes with the R&D and current scientific understanding of ecosystems to deliver general integrated assessments.

An integrated monitoring programme should therefore have the following characteristics to allow it to be readily used for integrated assessment purposes:

1) complementary spatio-temporal scales;
2) metrics informed by R&D;
3) a consistent suite of base metrics to which a suite of additional, adaptable metrics can be added;
4) changes to methodology should not disrupt time series;
5) the time frame of resulting data availability should be similar for all metrics;
6) a framework for data quality assurance.

In addition to intersessional work to contrast existing national initiatives, within each Member Country considerable progress could be made towards identifying the potential for their integrated monitoring programmes. Such coordination and integration at the national level would serve as a template for integration at the international level. In this respect, the ICES-EuroGOOS Planning Group on the North Sea Pilot Project

(NORSEPP) provides an opportunity for Member Countries to coordinate their respective monitoring programmes within a regional context and offers this for international coordination and integration by NORSEPP. This might be most efficient if multiple assessment needs are met, such as oceanographic influences on fish stock assessments and their influences on nutrients, primary production, and regional eutrophication.

15.3 Planning Group for Implementation of the Baltic Sea Regional Project

ICES must develop a strategy for implementing the GEF-funded Baltic Sea Regional Project (BSRP) and ensuring effective ways for ICES to contribute to and to benefit from the Project. The ICES-coordinated component of the Baltic Sea Regional Project is intended to address priority issues related to three of the five project modules: the Productivity Module, the Pollution and Ecosystem Health Module, and the Fish and Fisheries Module. Details of these priorities are:

Productivity Module

- Assessment of productivity levels in the adversely affected coastal and offshore ecosystems of the Baltic Sea;
- Application of innovative technologies and buoy systems in environmental assessments;
- Identification of links between land-based nutrient inputs and long-term changes of both productivity and biodiversity in selected areas.

Pollution and Ecosystem Health Module

- Application of ecological quality criteria for the Baltic Sea;
- Implementation of the HELCOM Cooperative Monitoring in the Baltic Marine Environment (COMBINE) Programme;
- Eutrophication and biological effects;
- Chemical pollution and biological effects;
- Invasive species and biodiversity;
- Multiple marine ecological disturbances (MMED).

Fish and Fisheries Module

- Improvement of assessment and management scheme for main commercial fish stocks in the Baltic Sea;
- Improvement and implementation of assessment and management measures for sustainable exploitation of coastal fish resources;
- Implementation of IBSFC Salmon Action Plan;
- Evaluation of the impact of fisheries upon ecosystems of the Baltic Sea.

It will be very difficult to integrate these activities into existing ICES groups, because most appropriate groups already have overloaded agendas. Therefore, a key component of the strategy will be the establishment of new expert groups under the Baltic Committee whose activities will lead to the development of ecosystem-level assessments, advice, and management. These groups will have to have integrated and overlapping terms of reference. There will also need to be an integrating Study Group on modelling issues, including the identification of the data requirements for ecosystem modelling work in the Baltic area.

A number of issues have been identified as relevant to enhance the Baltic Sea multispecies/ecosystem management process. In particular, it was pointed out to ACE from the Baltic GEF Project that the following issues have been identified as basic objectives of the groups comprising the Baltic GEF project:

a) Improvement of the temporal and spatial coverage of physical oceanographic factors (coarse and fine scale) and assessment of the plankton community (pelagic fish growth and feeding);

b) Improved acoustic estimates of pelagic species abundance and spatial distribution;

c) Establishment of a GIS Data Centre and GIS Database;

d) Development of environmental-fisheries integrated models for management;

e) Development of ecosystem health indicators versus indices;

f) Coordination of joint abundance surveys including stomach sampling (e.g., from market sampling and survey sampling);

g) Objective to move from single-species assessment/ management to multispecies assessment/ management.

h) Workshops to develop management models and indicators for sustainable fisheries (both open sea and coastal);

i) Coordination of Baltic Sea multispecies issues;

j) Promoting the use of Baltic herring and sprat for human consumption (e.g., dioxin issues).

ACE noted that many of these goals of the Baltic GEF project have important ecosystem implications which should be considered by appropriate ICES expert groups and ACE. Particular activities to advance work on these issues include:

Fish and Fisheries Module: The Study Group on Multispecies Assessment in the Baltic needs additional information related to the incorporation of environmental variability and spatial heterogeneity in fish stock modelling in the Baltic, which should be the central remit of the proposed BSRP Group for the Fisheries Module.

Pollution and Ecosystem Health Module: The traditional approach in the assessment and management of the Baltic Sea is based mainly on the assessment of water and sediment quality. Thus, the structure and function of the whole Baltic ecosystem as well as ecosystem health are not covered sufficiently. At present, there are not even any appropriate scientific tools available to use in ecosystem health assessment. Consequently, when developing the concept of ecosystem health, the following issues should be included in the remit for the supporting Working and Study Groups:

- Identification of natural sub-systems in coastal areas;
- Monitoring the biological effects of eutrophication, contamination, and fisheries;
- Development of criteria for assessing ecosystem health;
- Establishment of Environmental Reference Systems (including reference values, historical reference points, and reference areas);
- Development of classification lists of endangered species for different Baltic sub-regions;
- Updating and continuing development of existing biotope/habitat classification;
- Evaluating biological diversity (including xenodiversity/invasive species);
- Providing the scientific support for implementing nature conservation areas (including management/protection plans);
- Studying the effects of pollution on the functioning and structure of the ecosystem;
- Multiple Marine Ecological Disturbances (MMED).

These may require new terms of reference for existing groups as well as new groups to work in support of this module.

The implementation of this concept will require the development of decision-maker-friendly tools including decision-maker-friendly assessments and advice to management, using the standard tools of objectives, indicators, and reference points that ICES already uses in its advice.

Productivity Module: An ecosystem-based approach to marine assessment and management requires quantification of productivity, which in turn requires the BSRP to assess existing data, data needs, and data collection strategies for Baltic Sea productivity. Currently, Baltic Sea productivity-related data are collected at several separate trophic levels (primary production, phytoplankton, mesozooplankton, macrozoobenthos, phytobenthos, and fish biomass), but the data are rarely interpreted in the context of trophic interactions. Indicators have to be developed that allow material and energy flows to be followed from producers to consumers, including also the abiotic resources necessary to primary producers. The stability of these flows with respect to external disturbances should also be assessable from the indicator system. It will be important that the group is aware of indicator developments in other parts of ICES and takes heed of the guidelines prepared by ACE.

So far, productivity data have been collected mainly based on organism size, habitat type, or taxonomic entity. These data should be analysed with respect to functional groups of organisms, fulfilling specific functions within trophic flows in the Baltic Sea. Gaps where the current data do not cover relevant pathways should be identified and adaptations to the data collection and analysis strategy should be recommended for implementation by the BSRP. The use of the modern instrumentation and techniques in collecting marine productivity data should be evaluated and the need for unified standards within the Baltic Sea region should be addressed, in particular in partnership with the Baltic Ocean Observing System (BOOS) whose main concern is in the field of operational monitoring systems.

Ecosystem Modelling Module: There is need for complete ecosystem models of the Baltic covering the food web from nutrients to zooplankton. In current models used to address eutrophication and harmful algal bloom (HAB) issues, top-down control is truncated and parameterized in terms of mortality, whereas fishery models ignore the bottom-up effects. A first step to link ecosystem models and fish aspects is addressed in larvae drift models, but much more work has to be done to develop models that link bottom-up and top-down controls. Any new modelling group should interact closely with the Study Group on Modelling of Physical-Biological Interactions (SGPBI) in order to benefit from the best available expertise, and to ensure complementary activities. It was also clear that the work of this group complements the modelling work of SGMAB, and close association with that group was also necessary.

References

ICES. 2002. Report of the ICES Advisory Committee on Ecosystems, 2002. ICES Cooperative Research Report, 254. 129 pp.

ICES. 2003. Report of the Study Group on Precautionary Reference Points for Advice on Fishery Management. ICES CM 2003/ACFM:15.

ANNEX 1: REVIEW OF EVIDENCE FOR JUSTIFICATION FOR THE PROPOSED OSPAR PRIORITY LIST OF THREATENED AND DECLINING SPECIES

1 Introduction

The OSPAR Biodiversity Committee has nominated a variety of marine species that are considered threatened or declining and ICES was asked in 2002 to contribute to the peer-review process for these nominations. Whilst the better-known species were evaluated by several ICES working groups in 2002, ICES was not able to formulate advice for seven fish species (ICES, 2002). Hence, the Working Group on Fish Ecology (WGFE) was requested to evaluate these proposals. The species considered by WGFE were:

- Sea lamprey (*Petromyzon marinus*);
- Sturgeon (*Acipenser sturio*);
- Allis shad (*Alosa alosa*);
- Houting (*Coregonus lavaretus oxyrhinchus*);
- Short-snouted seahorse (*Hippocampus hippocampus*);
- Long-snouted seahorse (*Hippocampus guttulatus*);
- Couch's goby (*Gobius couchi*).

The OSPAR Texel-Faial criteria for identifying species in need of protection are given in Table 1.1, and the criteria identified for the species covered here are summarized in Table 1.2. The OSPAR areas are: Region I—Arctic Waters, Region II—Greater North Sea, Region III—Celtic Seas, Region IV—Bay of Biscay and Iberian Waters, and Region V—Wider Atlantic. A late request to assess the proposals for loggerhead and leatherback turtles was also addressed.

Other relevant policy drivers: Several other conventions have addressed the conservation of the species nominated by OSPAR (Table 1.3). These include the following:

The Habitats Directive: Council Directive 92/43/EEC on the conservation of natural habitats and of wild flora and fauna. This requires measures to be taken to maintain or restore to favourable conservation status in their natural range, habitats and species of wild flora and fauna of Community interest and listed in Annexes to the Directive. The directive includes lists of 623 species for which Member States must consider the designation of Special Areas of Conservation (SACs).

Bern Convention: Convention on the Conservation of European Wildlife and Natural Habitats. The aims of this Convention are to conserve wild flora and fauna and their natural habitats, especially those species and habitats whose conservation requires the cooperation of several States, and to promote such cooperation. Particular emphasis is given to endangered and vulnerable species, including endangered and vulnerable migratory species.

CITES: The Convention on International Trade in Endangered Species of Wild Fauna and Flora is an international agreement among Governments. Its aim is to ensure that international trade in listed species of wild animals and plants does not threaten the survival of their populations.

Reference

ICES. 2002. Report of the ICES Advisory Committee on Ecosystems, 2002. ICES Cooperative Research Report, 254: 42–46.

Table 1.1. Texel-Faial criteria for identifying species in need of protection.

1.	**Global importance**: Global importance of the OSPAR area for a species. Importance on a global scale, of the OSPAR Area, for the species is when a high proportion of a species at any time of the life cycle occurs in the OSPAR Area.
2.	**Regional importance**: Importance within the OSPAR Area, of the regions for the species where a high proportion of the total population of a species within the OSPAR Area for any part of its life cycle is restricted to a small number of locations in the OSPAR Area.
3.	**Rarity**: A species is rare if the total population size is small. In case of a species that is sessile or of restricted mobility at any time of its life cycle, a species is rare if it occurs in a limited number of locations in the OSPAR Area, and in relatively low numbers. In case of a highly mobile species, the total population size will determine rarity.
4.	**Sensitivity**: A species is "very sensitive" when: a. it has very low resistance (that is, it is very easily adversely affected by human activity); and/or b. it has very low resilience (that is, after an adverse effect from human activity, recovery is likely to be achieved only over a very long period, or is likely not to be achieved at all). A species is "sensitive" when: a. it has low resistance (that is, it is easily adversely affected by human activity); and/or b. it has low resilience (that is, after an adverse effect from human activity, recovery is likely to be achieved only over a long period).
5.	**Keystone species**: a species which has a controlling influence on a community.
6.	**Decline**: means an observed or indicated significant decline in numbers, extent or quality (quality refers to life history parameters). The decline may be historic, recent or current. "Significant" need not be in a statistical sense.

Table 1.2. Potential threatened fish species, indicating the OSPAR regions that were suggested and the rationale supporting the original nomination.

Species	Area	Global importance	Local Importance	Rarity	Sensitivity	Keystone species	Decline
Sea lamprey (*Petromyzon marinus*)	I, II, III, IV			✓	✓		✓
Sturgeon (*Acipenser sturio*)	II, IV	✓			✓		✓
Allis shad (*Alosa alosa*)	II, III, IV			✓			✓
Houting (*Coregonus lavaretus oxyrhinchus*)	II			✓			✓
Short-snouted seahorse (*Hippocampus hippocampus*)	II, III, IV, V		✓		✓		✓
Long-snouted seahorse (*Hippocampus guttulatus*)	II, III, IV, V		✓		✓		✓
Couch's goby (*Gobius couchi*)	All	✓		✓	✓		✓

Table 1.3. Conservation categories for the nominated species.

Species	Habitats Directive	CITES	Bern Convention	IUCN
Common sturgeon (*Acipenser sturio*)	Annexes II and IV	Appendix I	Annex II	classified as Critically Endangered
Allis shad (*Alosa alosa*)	Annexes II and V		Annex III	
Houting (*Coregonus lavaretus oxyrhynchus*)	Annexes II and V		Annex III	
Couch's goby (*Gobius couchi*)				
Short-snouted seahorse (*Hippocampus hippocampus*)		Appendix II		classified as Vulnerable
Long-snouted seahorse (*Hippocampus guttulatus*)		Appendix II		classified as Vulnerable

2 Fish species

2.1 Sea lamprey (*Petromyzon marinus*)

Status and distribution

The sea lamprey is a native anadromous species occurring over much of the Atlantic coastal area of western and northern Europe (from northern Norway to the western Mediterranean) and eastern North America, and in estuaries and easily accessible rivers in these regions. Occasional specimens are taken in mid-water in the Atlantic Ocean (Lelek, 1973). They have been reported as far east as the Aegean Sea (Economidis *et al.*, 1999).

According to the FAO fishery statistics (Figure 2.1.1), the main fisheries for sea lamprey are in France, Portugal, Latvia, and Estonia (it is assumed that prior to 1988 the USSR reported the catch from Latvia and Estonia). The indications are that the catch peaked in 1989 with a total declared catch of 254 tonnes; after 1990 the annual catch declined to an average of 27 tonnes. However, these data must be viewed with

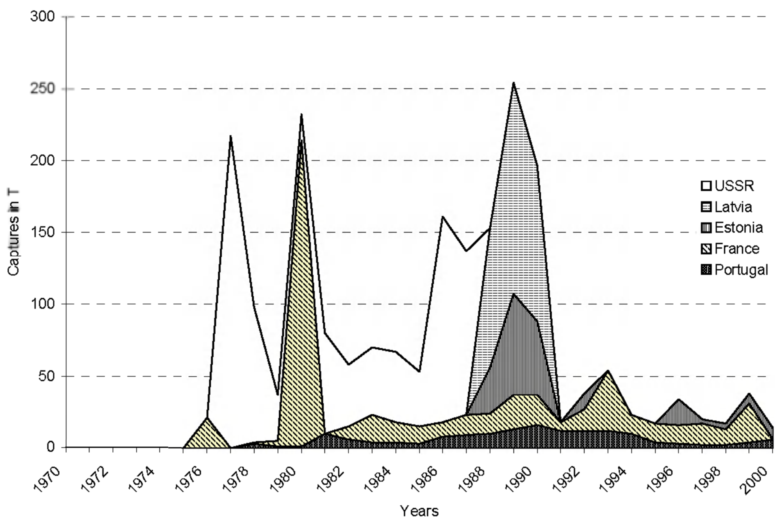


Figure 2.1.1. Reported total catch in tonnes of sea lamprey in the OSPAR maritime area (http://www.fishbase.org/report/FAO/FAOCatchList.cfm?scientific=Petromyzon%20marinus).

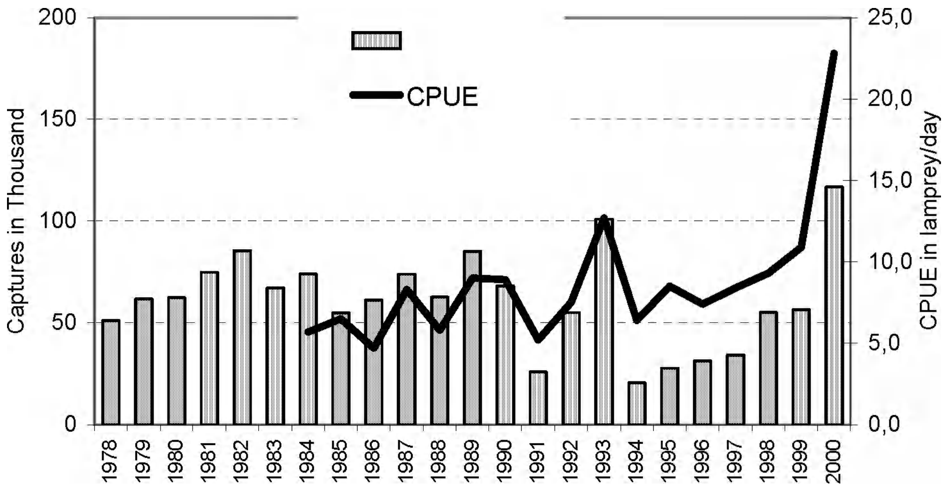


Figure 2.1.2. Catch and CPUE for sea lamprey in the Gironde system between 1978 and 2000. Captures in thousand of fish and CPUE in fish/trammel net/fisherman (From Girardin *et al.*, 2002).

caution and are likely to be a significant underestimation, as catches in the Gironde system alone have been between 40 tonnes and 155 tonnes per year (Girardin *et al.*, 2002), and the catch per unit of effort (CPUE) index of abundance suggests an increase in the size of the population (Figure 2.1.2).

In France, sea lamprey is also exploited commercially in the Loire and Adour Basin (Castelnaud, 2000). The species is present in most of the basins at least below the first obstacles. There is no evidence of any recent decrease in abundance and in other basins there are signs of recovery (e.g., Rhine system, T. Changeux, Conseil Supérieur de la Pêche, pers. comm.).

In Finland, reports of lamprey from coastal waters appear rare; most of the captures are taken off the southern coast and only a few in rivers (Tuunainen *et al.*, 1980). Though they appear relatively widespread in the rivers of Ireland (Kurz and Costello, 1999) and the UK (Brown *et al.*, 1997), they have been reported to be in decline. In the British Isles, lamprey is absent from northern rivers (i.e., it does not appear to occur north of the Great Glen of Scotland) and has become extinct in a number of southern rivers due to pollution and engineering barriers (Maitland, 1980; Maitland and Campbell, 1992). In the Severn, prior to the erection of navigation weirs in the 19th century, sea lampreys were considered abundant and supported a valuable fishery. Following the construction of these barriers, the species declined (Randell, 1882; Day, 1890) so that today it is only rarely caught (Henderson, pers. comm.). There are several landlocked populations in North America, but in Great Britain the only site where the species is known to feed in fresh water is Loch Lomond (Maitland *et al.*, 1994).

In Portugal, sea lamprey support commercial fisheries in the central and northern parts of the country, with the River Minho being the major fishery (P.R. Almeida, pers. comm.). There have been numerous reports of a

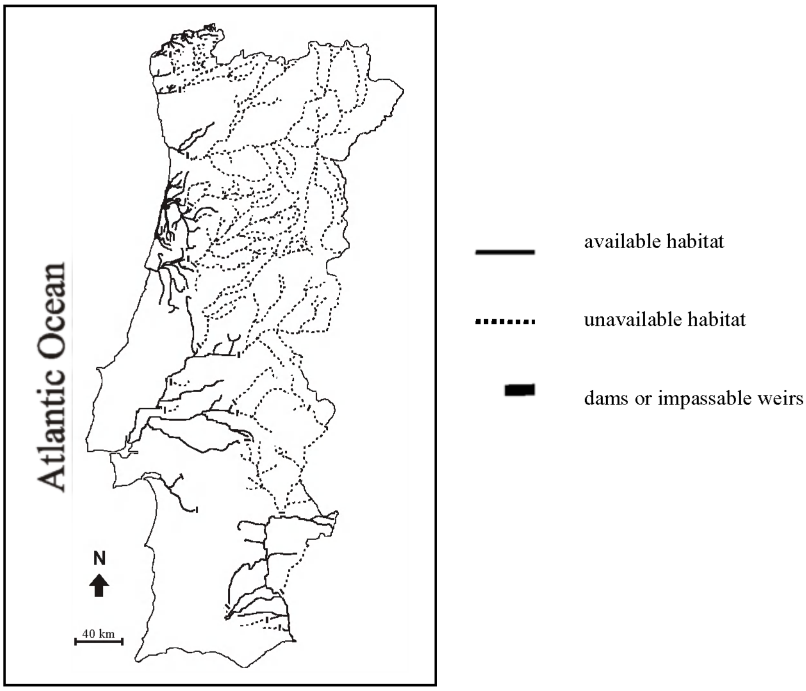


Figure 2.1.3. Habitat available to sea lamprey populations in Portuguese river basins where the species is known to occur (Almeida *et al.*, 2002a).

reduction in sea lamprey populations in Portuguese rivers (Guimarães, 1988; Almaça, 1990; Assis, 1990; Assis *et al.*, 1992; Ferreira and Oliveira, 1996; Almeida *et al.*, 2000, 2002a), so that they are now considered vulnerable (Vários, 1991).

Physical barriers on water courses, stream flow, water temperature, and stream bed composition can have a significant effect on the distribution of spawning sea lampreys (Haro and Kymar, 1997). In Portugal, one of the main reasons for their decline has been the construction of dams resulting in a reduction of available habitat (Figure 2.2.3) (Almeida *et al.*, 2002a). Alteration of the discharge pattern associated with dams has also been implicated, with high discharge causing a delay in the timing of the migration (Machado-Cruz *et al.*, 1990; Almeida *et al.*, in press; Quintella *et al.* (in press, cited by Almeida *et al.*, 2002b)), while at low flows the cues and stimulus to migrate may be reduced (Almeida *et al.*, 2000).

The ammocoete larvae are usually found in silty sands in running water, although they may occur in silt and gravel beds in large lakes (e.g., Loch Lomond). Given that a large proportion of the life cycle of lampreys is spent in burrows in silt beds, these beds must be considered as essential fish habitat for lamprey, as must spawning gravels. Certainly habitat connectivity is likely to be important and losses may be particularly high during dispersal from the nest to the ammocoete silt beds, at metamorphosis, and on their migration downstream (Swink, 1995). Almeida and Quintella (2002) have shown that sea lamprey ammocoetes use different habitat types depending on life stage (size), ranging from silt-sand for fish of 20–60 mm to coarse-grained sediment for those between 140 mm and 200 mm. As a result channelization, mainly through the removal of areas of riffle and associated spawning gravels, and the dredging of essential nursery silt beds may entirely eliminate lampreys from a river.

As lampreys can be regarded as almost entirely riverine and sedentary animals, they are susceptible to pollution and, as most polluting effluents are directed into running waters (and so to the sea), many rivers which became grossly polluted in the past lost their populations of lampreys. In addition to direct toxic effects, pollution can have a major impact on lampreys by smothering both spawning gravels and nursery silts. Eutrophication acts in a similar way to some other forms of pollution: the algal and bacterial production resulting from increased nutrients smothers both the spawning gravels (preventing spawning or killing eggs) and the nursery silts, creating anoxic conditions.

Various types of pollution, either alone or in combination with other factors, limit the distribution of sea lampreys (Morman *et al.*, 1980). Streams which are affected by domestic or industrial pollution or agriculture usually have no larvae, or only support small or discrete populations. Formation of methane in bottom habitats was considered to be the reason for the mortality and disappearance of sea lamprey larvae from areas where they were formerly abundant (Wilson, 1955). Spawning-run sea lamprey are known to be attracted to streams containing ammocoete populations (Moore and Schleen, 1980). This has been proved experimentally with chemical attractants which show that sexually immature sea lamprey migrants select water containing rinses from ammocoetes over other water (Teeter, 1980).

Both water abstraction and land drainage are likely to have similar negative effects on lamprey populations, leading to unstable habitats with variable water levels which flood and disturb both spawning gravels and nursery silts at some times but leave them exposed at other times. Certainly in rivers with intermittent flow (e.g., due to hydro schemes, etc.), larvae can live for some time in exposed beds but are often found dead in such situations. In general, flow intermittency is considered limiting to ammocoete populations (Morman *et al*., 1980). Low and unstable flows were considered by Morman (1987) to be two of the major limiting factors for the absence or scarcity of larvae in many streams, other factors of importance being pollution, sedimentation, and hard or unstable bottoms.

The larvae are eaten by eel, stickleback, and other fish as well as several different species of birds (e.g., herons). There are a number of records of birds and mammals attacking adult sea lamprey, especially at spawning time, but this is not considered a significant impact on the populations.

Ammocoetes feed on minute organisms filtered from the mud, and high mortality probably occurs at metamorphosis with the shift to parasitism (Swink, 1990; Walters *et al*., 1980). According to Walters *et al.* (1980), up to 80% mortality could happen at this time. The adults have been reported from a number of host species (see review by Kelly and King (2001)).

Relatively little is known about the precise habitats occupied by adult sea lamprey. Though adults are sometimes caught at sea, the precise conditions in which they occur have not been described. They are only rarely caught in trawls, suggesting that marine fishing is not a major threat. However, as they enter fresh water to spawn, they become vulnerable to exploitation and, in Portugal, Almeida *et al.* (2002a) mention that stocks are overfished and heavily poached. This does not seem to be the case in the Gironde system (Girardin *et al.,* 2002). However, the lack of homing suspected for this species (Bergstedt and Seelye, 1995) and the lack of information about the exchange between basins makes it difficult to assess to what degree the populations are self-sustaining.

Some sea lamprey are taken by power stations, but there is no evidence in the UK that the numbers concerned are detrimental to stocks, and such catches can be a valuable tool in monitoring (Henderson, pers. comm.). Adult sea lamprey, being long and thin fish, are vulnerable to passage through pumps.

Conclusion in relation to the Texel-Faial criteria for the identification of threatened species and habitats

The main threats to this species come from the continual loss of access, the degradation of spawning habitat, and poor water quality. Examples where threats such as these have been linked to human activities are the decline of *P. marinus* in the Dordogne (France) due to water pollution, the erection of dams and dredging of the channel (Ducasse and Leprince, 1980), and the blocking off of access by the fish to parts of the River Tagus. *P. marinus* is common in the Portuguese portion of the river Tagus, but it cannot move through to Spain because of dams lacking appropriate fish passes (Assis, 1990).

In the Dordogne, since the study of Ducasse and Leprince (1980), a number of fish passes have been installed and are known to be used by sea lamprey (Travade *et al.*, 1998). In addition, the cessation of the gravel extraction in the Dordogne and Garonne may explain the increase in the population in the Gironde, as indicated by the trend in CPUE (Figure 2.1.2).

Owing to its decline across Europe, the sea lamprey is now given some legal protection. It is listed in Annexes IIa and Va of the EU Habitats and Species Directive, Appendix III of the Bern Convention, and is listed under the UK Biodiversity Action Plan. There is no Red Data Book for fish in Great Britain, but Maitland (2000) considers this species to be Vulnerable. The Red Data Book for Ireland (Whilde, 1993), published before the IUCN (1994) revision of categories, lists the sea lamprey as Indeterminate.

Quantitative data indicating a decline in either the range or in the size of the population were considered lacking. The statistics from the FAO indicate a decline, as do qualitative statements in the literature. However, it is evident that the FAO statistics underestimate, at least in France, the true level of captures and thus interpretation of the data must be made with caution.

There is certainly much circumstantial evidence that human activity can have a detrimental effect on sea lamprey populations and in some cases there is strong historical evidence, for example, in the Severn, that the species was more abundant in the past. In the absence of quantitative data, it is recommended that further efforts, in particular a search of the grey literature to confirm the current status of this species, be undertaken.

In those rivers where a self-maintaining population still exists, the lack of data will make it difficult to detect changes as a result of management action. In those rivers where the population has become extinct, the effect of any intervention will be more easily quantified.

Most of the environmental problems affecting sea lamprey are in freshwater and estuarine environments, and there is no evidence that anthropogenic activities in fully marine environments are threatening sea lamprey populations.

References

Almaça, C. 1990. A lampreia e o esturjão na bacia do Douro. Observatório, 1: 377–382.

Almeida, P.R., and Quintella, B.R. 2002. Larval habitat of the sea lamprey (*Petromyzon marinus* L.) in the River Mondego (Portugal). *In* Freshwater fish conservation: options for the future, pp. 121–130. Ed. by M.J. Collares-Pereira, M.M. Coelho, and I.G. Cowx. Fishing News Books, Blackwell Science, Oxford.

Almeida, P.R., Quintella, B.R., Dias, N.M., and Andrade, N. 2002a. The anadromous sea lamprey in Portugal: Biology and conservation perspectives. *In* The biology of lampreys. Symposium Proceedings, International Congress on the Biology of Fish. American Fisheries Society, 21–25 July, Vancouver, Canada, pp. 49–58. Ed. by M. Moser, J. Bayer, and D. MacKinlay.

Almeida, P.R., Quintella, B.R., and Dias, N.M. In press. Movement of radio-tagged anadromous sea lamprey during the spawning migration in the River Mondego (Portugal). Hydrobiologia.

Almeida, P.R, Silva, H.T., and Quintella, B. 2000. The migratory behaviour of the sea lamprey, *Petromyzon marinus* L., observed by acoustic telemetry in River Mondego (Portugal). *In* Advances in fish telemetry, pp. 99–108. Ed. by A. Moore and I. Russel. CEFAS, Suffolk, UK.

Almeida, P.R., Silva, H.T., and Quintella, B.R. 2002b. The spawning migration of the sea lamprey (*Petromyzon marinus* L.), in the River Mondego (Portugal). *In* Global importance of local experience, pp. 381–386. Ed. by M.A. Pardal, J.C. Marques, and M.A. Graça. Aquatic ecology of the Mondego river basin. Imprensa da Universidade de Coimbra, Portugal.

Assis, C.A. 1990. Threats to the survival of anadromous fishes in the river Tagus, Portugal. Journal of Fish Biology, 37(Supplement A): 225–226.

Assis, C., Costa, J.L., Costa, M.J., Moreira, F., Almeida, P., and Gonçalves, J. 1992. Ameaças à sobrevivência dos peixes migradores do Tejo. Sugestões para a sua conservação. Publicações Avulsas do INIP, 17: 429–446.

Bergsted, R.A., and Seelye, J.G. 1995. Evidence for lack of homing by sea lampreys. Transactions of the American Fisheries Society, 124: 235–239.

Brown, A.E., Burn, A.J., Hopkins, J.J., and Way, S.R. (eds.). 1997. Habitats Directive: Selection of Special Areas of Conservation in the UK. Joint Nature Conservation Committee Report No. 270. Joint Nature Conservation Committee, Peterborough.

Castelnaud, G. 2000. Localisation de la pêche, effectifs de pêcheurs et production des espèces amphihalines dans les fleuves français. Bulletin français de pêche et pisciculture, 357/358: 439–460.

Day, F. 1890. Notes on the fish and fisheries of the Severn. Proceedings of the Cotswold Naturalists Field Club, 9: 202–219.

Ducasse, J., and Leprince, Y. 1980. Etude préliminaire de la biologie des lamproies dans les bassins de la Garonne et de la Dordogne. Mémoire de fin d'études ENITEF, Cemagref de Bordeaux, Div. A.L.A. 160 pp.

Economidis, P.S., Kallianiotis, A., and Psaltopoulou, H. 1999 Two records of sea lamprey from the north Aegean Sea. Journal of Fish Biology, 55: 1114–1118.

Ferreira, M.T., and Oliveira, J.M. 1996. Gestão da lampreia marinha *Petromyzon marinus* no rio Tejo. Anuário do Instituto Superior de Agronomia, 45: 401–439.

Girardin, M., Castelnaud, G., and Beaulaton, L. 2002. Surveillance halieutique de l'estuaire de la Gironde – Suivi des captures 2000 – Etude de la faune circulante 2001. Rapport pour DF CNPE du Blayais / Etude Cemagref, groupement de Bordeaux, Cestas, No. 74. 196 pp.

Guimarães, M.T. 1988. Medidas para a protecção e fomento da lampreia do mar (*Petromyzon marinus* L.) no rio Mondego. Actas do Colóquio Luso-Espanhol sobre Ecologia das Bacias Hidrográficas e Recursos Zoológicos: 195–203.

Haro, A., and Kynar, B. 1997. Video evaluation of passage efficiency of American shad and sea lamprey in a modified ice harbour fishway. North American Journal of Fisheries Management, 17.

Kelly, F.L., and King, J.J. 2001. A review of the ecology and distribution of three lamprey species, *Lampetra fluviatilis* (L.), *Lampetra planeri* (L.), and *Petromyzon marinus* (L.): a context for conservation and biodiversity considerations in Ireland. Biology and Environment: Proceedings of the Royal Irish Academy, 101B(3): 165–185.

Kurz, I., and Costello, M.J. 1999. An outline of the biology, distribution and conservation of lampreys in Ireland. Irish Wildlife Manuals, No. 5. 27 pp. Du´chas—The Heritage Service, Dublin.

Lelek, A. 1973. Occurrence of the Sea Lamprey in midwater off Europe. Copeia, 1: 136–137.

Machado-Cruz, J.M, Valente, A.C.N., and Alexandrino, P.J.B. 1990. Contribuição para a caracterização ecológica e económica da pesca de migradores a jusante da barragem de Belver, Rio Tejo. Actas 1º Congresso do Tejo. Que Tejo, que futuro?: 189–200.

Maitland, P.S. 1980. Review of the ecology of lampreys in northern Europe. Canadian Journal of Fisheries and Aquatic Sciences, 37: 1944–1952.

Maitland, P.S. 2000. Fish. *In* Local Biodiversity Action Plans—technical information on species: IV. Vertebrate animals, pp. 81–91. Ed. by S.D. Ward. Scottish Natural Heritage Review No 10.

Maitland, P.S., and Campbell, R.N. 1992. Freshwater fishes of the British Isles. HarperCollins, London.

Maitland, P.S., Morris, K.H., and East, K. 1994. The ecology of lampreys (Petromyzonidae) in the Loch Lomond area. Hydrobiologia, 290: 105–120.
Moore, H.H., and Schleen, L.P. 1980. Changes in spawning runs of sea Lamprey (*Petromyzon marinus*) in selected streams of Lake Superior after chemical control. Canadian Journal of Fisheries and Aquatic Sciences, 37: 1851–1860.
Morman, R.H. 1987. Relationship of density to growth and metamorphosis of caged larval sea lampreys, *Petromyzon marinus* Linnaeus, in Michigan streams. Journal of Fish Biology, 30: 173–181.
Morman, R.H., Cuddy, D.W., and Rugen, P.C. 1980. Factors influencing the distribution of Sea Lamprey (*Petromyzon marinus*) in the Great Lakes. Canadian Journal of Fisheries and Aquatic Sciences, 37: 1811–1826.
Quintella, B.R., Dias, N.M., and Almeida, P.R. In press. Efeito das variações de caudal induzidas por aproveitamentos hidroeléctricos, no comportamento migratório da lampreia-marinha (*Petromyzon marinus* L.). *In* Actas do 2º Congresso Nacional de Conservação da Natureza.
Randell, J. 1882. The Severn Valley.
Swink, W.D. 1990. Effect of lake trout size on survival after a single sea lamprey attack. Transactions of the American Fisheries Society, 119: 996–1002.
Swink, W.D. 1995. Growth and survival of newly parasitic sea lampreys at representative winter temperatures. Transactions of the American Fisheries Society, 124: 380–386.
Teeter, J. 1980. Pheromone communication in sea lampreys (*Petromyzon marinus*): implications for population management. Canadian Journal of Fisheries and Aquatic Sciences, 37: 2123–2132.
Torblaa, R.L., and Westman, R.W. 1980. Ecological impacts of lampricide treatments on sea lampreys (*P. marinus*) ammocoetes and metamorphosed individuals. Canadian Journal of Fisheries and Aquatic Sciences, 37: 1835–1850.
Travade, F., Larinier, M., Boyer-Bernard, S., and Dartiguelongue, J. 1998. Performance of four fish pass installations recently built on two rivers in south-west France. *In* Fish migration and fish bypasses, pp.146–170. Ed. by M. Jungwirth, S. Schmutz, and S. Weiss. Fishing News Books-Blackwell Science Ltd., Oxford.
Tuunainen, P., Ikonen, E., and Auvinen, H. 1980. Lampreys and lamprey fisheries in Finland. Canadian Journal of Fisheries and Aquatic Sciences, 37: 1953–1959.
Vários. 1991. Livro Vermelho dos Vertebrados de Portugal. vol. II – Peixes Dulciaquícolas e Migradores. SNPRCN, Lisboa. 55 pp.
Walters, C.J., Spangler, G., Christie, W.J., Manion, P.J., and Kitchell, J.F. 1980. A synthesis of knowns, unknowns and policy recommendations from the Sea Lamprey International Symposium. Canadian Journal of Fisheries and Aquatic Sciences, 37: 2202–2208.
Whilde, A. 1993. Threatened mammals, birds, amphibians and fish in Ireland: Irish Red Data Book 2: vertebrates. HMSO, Belfast.
Wilson, F.H. 1955. Lampreys in the Lake Champlain basin. American Midland Naturalist, 54: 163–172.

2.2 Common sturgeon (*Acipenser sturio*)

Status and distribution

The marine geographical distribution of *Acipenser sturio* in the OSPAR area is now restricted from the southern Bay of Biscay to the North Sea including the British Isles (Castelnaud, 1988; Rochard *et al.*, 1990, 1997; Lepage and Rochard, 1995) (Figure 2.2.1). The last recorded information about the former population of the Rioni river, a tributary of the Black Sea in Georgia, was more than twenty years ago (Ninua, 1976) and recent efforts of German and Georgian scientists (J. Gessner, Society to Save the Sturgeon, Berlin, pers. comm.) did not succeed in catching any common sturgeon in this region. According to the criteria of Grogan and Boreman (1998), *A. sturio* is now considered extinct in the Rioni. The status of *A. sturio* in the Danube is uncertain.

The common sturgeon reproduces only in the Garonne and Dordogne Rivers in southwest France (Castelnaud *et al.*, 1991; Rochard, 1992; Williot *et al.*, 1997) and juveniles stay several years in the Gironde estuary in feeding zones, and this region is of particular importance for the species (Rochard *et al.*, 2001).

Sturgeon were common and widespread in Northeastern European waters in the 8–11th centuries, however they underwent a major decline in abundance and distribution in the 13–14th centuries due to exploitation and damming of rivers. Their abundance increased after storms destroyed many dams, and remained relatively high until the 18th century, when exploitation and the renewed damming of rivers initiated further widespread declines (Hoffmann, 1996). The common sturgeon was historically present in most large western European rivers (Magnin, 1959; Hoffmann, 1996) from the Black Sea (Ninua, 1976) to the Baltic Sea, including the Iberian peninsula (Classen, 1944; Almaça, 1988; Almaça and Elvira, 2000) and the British isles. Populations of the Rioni, Rhone, Ebro, Guadalquivir, Guadiana, Gironde, Rhine, and Elba were probably the most important.

However:

- considering the results of Ludwig *et al.* (2002), it is not certain that sturgeon from the Barents Sea were *A. sturio*; they could also have been *Acipenser oxyrhinchus*;
- the historical presence of *A. sturio* in the Danube is also questionable (I. Navodaru, Danube Delta Institute, pers. comm.);
- there is a strong controversy about the presence of *A. sturio* in sympatry with the Adriatic sturgeon (*A. naccarii*) in the Mediterranean part of the Iberian peninsula (Elvira *et al.*, 1991a, 1991b; Garrido Ramos *et al.*, 1997).

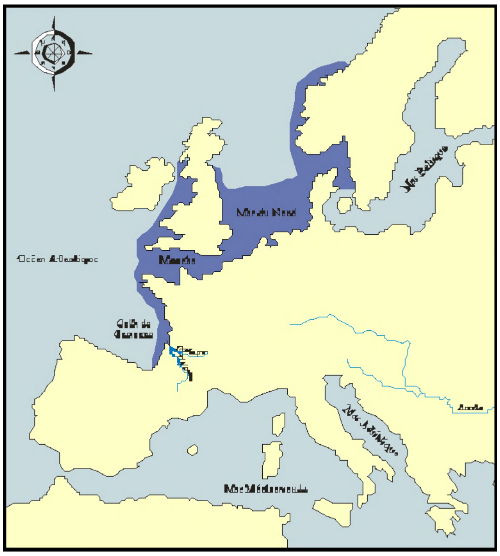


Figure 2.2.1. Present distribution area of *A. sturio* (from Elie, 1997). The shaded area represents the area where the species is still encountered. It does not imply that the distribution within this zone is homogeneous in abundance.

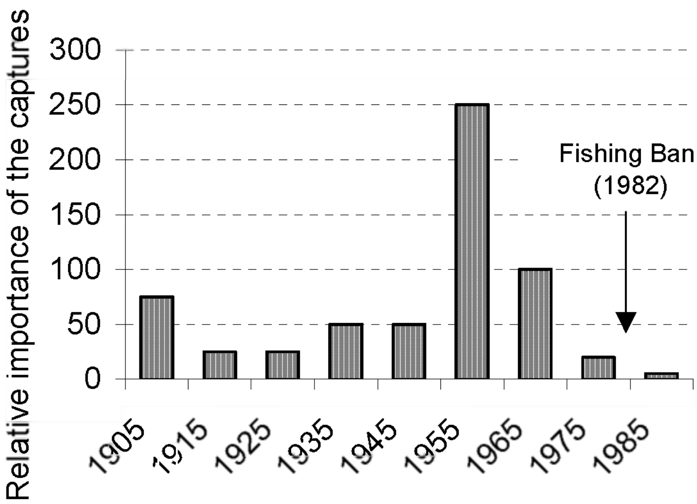


Figure 2.2.2. Relative importance of the captures of common sturgeon in the Gironde basin (from Rochard, 2002b).

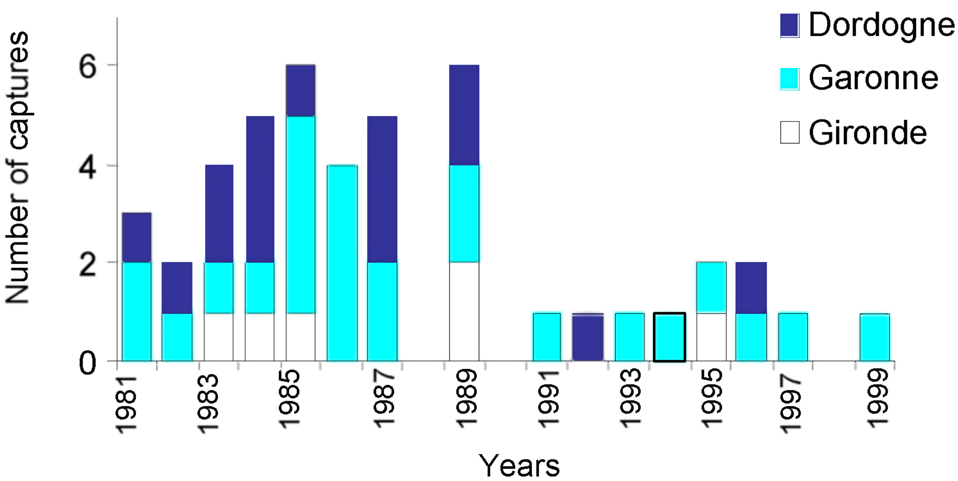


Figure 2.2.3. Number of incidental captures of adult common sturgeon in the Gironde system (from Rochard, 2002b).

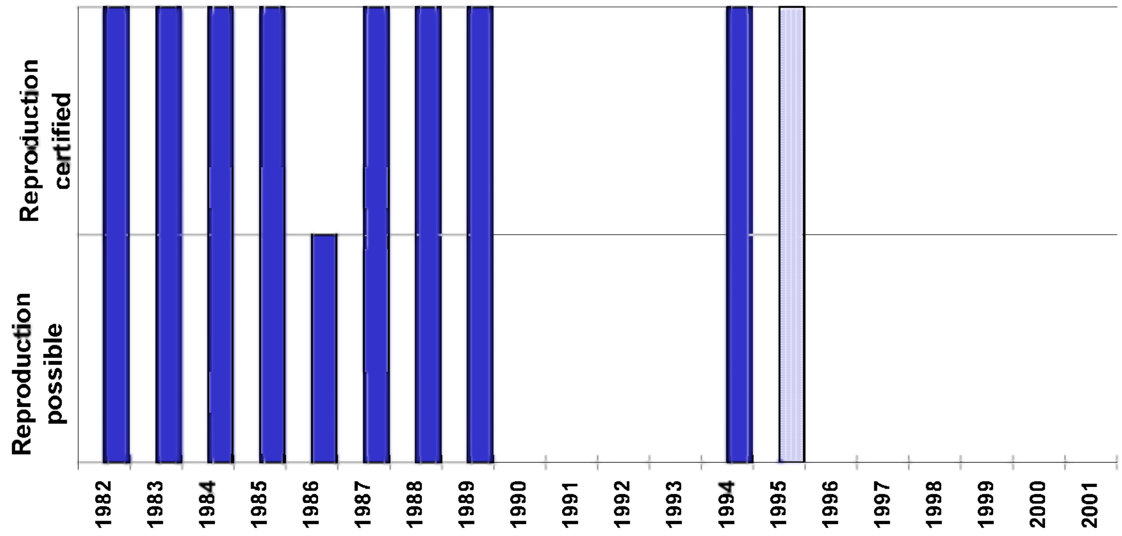


Figure 2.2.4. Deduction of the evidence of common sturgeon reproduction in the Gironde system from the monitoring of the population in the estuary. The 1995 cohort comes from artificial reproduction (Williot *et al.*, 2000) and stocking.

All species of sturgeon are now considered as endangered (Rochard *et al.*, 1990; Birstein, 1993; Hoffmann, 1996) and all populations of common sturgeon have exhibited drastic declines. Central European basins (Magnin, 1959; Kinzelbach, 1987; Gessner, 2000) were deserted at the end of the 19th century, while populations of the Rhone (Tabardel, 1994) and of the Guadalquivir (Elvira *et al.*, 1991a, 1991b) became extinct in the 1950s. However, as it is a long-lived species, vagrant large individuals from these virtually extinct populations are occasionally encountered at sea or in estuaries (Elvira and Almodovar, 1993).

The Gironde population was intensively exploited for caviar from the beginning of the 20th century (Roule, 1922; Magnin, 1962). Maximum captures probably occurred around 1955 and the decline was then drastic (Figure 2.2.2) (Trouvery *et al.*, 1984). Juveniles were also caught as well, both in fresh and marine coastal waters (Letaconnoux, 1961). Exploitation ended in 1982, when the species became completely protected in France. As the abundance continued to decrease, the species has been successively listed in all major conventions (CITES Appendix I; Bern Convention Appendix II (1998); Habitats, Fauna and Flora Directive Appendices II and IV). It is also listed in the IUCN red list of threatened animals as critically endangered.

From this time, the status of the population has been closely monitored and studied (Castelnaud *et al.*, 1991). Even though the species is now protected, the number of incidental captures of adults mentioned by fishermen has shown a drastic decline (Figure 2.2.3) (Williot *et al.*, 1997, 2002; Rochard, 2002a).

From the monitoring of the population in the estuary, it has been concluded that reproduction does not occur annually (Figure 2.2.4, Rochard *et al.*, 2001). The last natural reproduction occurred in 1994 (Elie, 1997). In 1995, 9,000 artificially reared juvenile common sturgeons were stocked (Williot *et al.*, 2000). The effectiveness of the stocking is currently being evaluated (Lochet, 2002).

By comparing the number of fish within the cohorts which are estimated to return for spawning (i.e., fish hatched before 1988) with the number of adults incidentally caught in the system by the shad commercial fishery, it was deduced that either the number of adults was underestimated or, more likely, the mortality encountered by the common sturgeon during their stay in marine areas is still very high (M>> 0.25).

Sturgeon combine both diadromy (migrations between marine and freshwater essential habitats) and gigantism (trophic strategy, late reproduction, long-living animals, iteroparity); this leads to a high level of sensitivity to habitat and connectivity alteration (Angermeier, 1995; Auer, 1996; MacDowall, 1999) and exploitation (Boreman, 1997).

Male common sturgeon mature at 8–12 years and females at 13–16 years and live to 100 years (Fishbase, 2000). In recent times, Williot *et al.* (1997, 2002) hypothesized that the physiological quality of the males could also have declined. Potential spawning habitats have been characterized (Jego *et al.*, 2002) and more than twenty sites are available either in the Garonne or the Dordogne River. Juveniles may stay several years in the estuary (Rochard, 1992; Rochard *et al.*, 2001) before they migrate to the sea. During their estuarine period, they feed on benthic organisms (Brosse *et al.*, 2000a, 2000b) and use very specific and localized habitats (Taverny *et al.*, 2002) associated with their prey.

Threats to diadromous sturgeon have been reviewed (Rochard *et al.*, 1990; Birstein, 1993). Among them, obstacles to migration are considered the major threat most often leading to the extinction of the population. In addition, historical commercial fisheries have led to the local extinction of many common sturgeon populations. Presently, a major threat is their occurrence in the by-catch in coastal fisheries operating along the European seaboard (Rochard *et al.*, 1997; Lepage *et al.*, 1998a; Mayer and Lepage 2001). Poaching activities also occur at sea at the entrance to the Gironde estuary (Mayer and Lepage, 2001; Lepage, 2002).

Habitats in rivers are legally protected in the Garonne and Dordogne, but there are new projects for gravel extraction in the estuary near one of the essential habitats for young sturgeon (Lepage *et al.*, 1998b, 2000). The incidental introduction of an alien sturgeon (*Acipenser baerii*) in the Gironde system in 2000 constitutes a new threat: these species can hybridize and probably compete for food. Moreover, fishermen have difficulties protecting one species, as the non-native species must be eradicated (Lepage, 2002).

Conclusion in relation to the Texel-Faial criteria for the identification of threatened species and habitats

The geographical distribution of the last known population of common sturgeon (spawning in the Gironde basin) is within the OSPAR area; the species is of particular importance in the Gironde system but can be encountered in most of the coastal zones. The decline in the OSPAR area, as well as in a number of other populations, is clear. The last remaining population has been monitored and still exhibits evidence of a decrease, and it may be that a viable population no longer exists.

Legally the species is completely protected, however, by-catch, all along the European coasts, is a source of mortality and poaching still occurs. Essential habitats for juveniles in the estuary have been characterized, although a new project of gravel extraction near one of their essential habitats constitutes a significant threat. In the basins where the species has become extinct, habitats used by common sturgeon have not necessarily been improved, hindering any restoration programme or any recolonization if it were to occur.

References

Almaça, C. 1988. On the sturgeon, *Acipenser sturio*, in the Portuguese rivers and sea. Folia Zoologica, 37(2): 183–191.

Almaça, C., and Elvira, B. 2000. Past and present distribution of *Acipenser sturio* L., 1758 on the Iberian Peninsula. Boletin Instituto Espanol de Oceanografia, 16(1–4): 11–16.

Angermeier, P.L. 1995. Ecological attributes of extinction-prone species: loss of freshwater fishes of Virginia. Conservation Biology, 9(1): 143–158.

Auer, N.A. 1996. Response of spawning lake sturgeons to change in hydroelectric facility operation. Transactions of the American Fisheries Society, 125: 66–77.

Birstein, V.J. 1993. Sturgeons and paddlefishes: threatened fishes in need of conservation. Conservation Biology, 7: 773–787.

Boreman, J. 1997. Sensitivity of North American sturgeons and paddlefish to fishing mortality. Environmental Biology of Fishes, 48: 399–405.

Brosse, L., Lepage, M., and Dumont, P. 2000a. First results on the diet of the young European sturgeon, *Acipenser sturio* Linnaeus, 1758, in the Gironde estuary. Boletin Instituto Espanol de Oceanografia, 16(1–4): 75–80.

Brosse, L., Rochard, E., Dumont P., and Lepage, M. 2000b. Premiers résultats sur l'alimentation de l'esturgeon européen, *Acipenser sturio* Linnaeus, 1758 dans l'estuaire de la Gironde et comparaison avec la macrofaune estuarienne présente. Cybium, 24(3)Suppl.: 49–61.

Castelnaud, G. 1988. L'opération de marquage de l'esturgeon dans l'estuaire de la Gironde (France) : une dimension européenne. ICES CM 1988/M:28, 12 pp.

Castelnaud, G., Rochard, E., Jatteau, P., and Lepage, M. 1991. Données actuelles sur la biologie d'*Acipenser sturio* dans l'estuaire de la Gironde. *In* Acipenser, pp. 251–275. Ed. by P. Williot. Cemagref Publications.

Classen, T.E. 1944. Estudio bio-estadistico del Esturion o sollo del Guadalquivir (*Acipenser sturio* L.). Minist. Mar. Inst. Español Oceanograf., Madrid. 112 pp.

Elie, P. (ed.). 1997. Restauration de l'esturgeon européen *Acipenser sturio*. Contrat Life rapport final du programme d'exécution. Etude Cemagref de Bordeaux No. 24. 381 pp.

Elvira, B., and Almodovar, A. 1993. Notice about the survival of sturgeon (*Acipenser sturio* L., 1758) in the Guadalquivir estuary (SW Spain). Archives of Hydrobiology, 129: 253–255.

Elvira, B., Almodovar, A., and Lobon-Cervia, J. 1991a. Sturgeon (*Acipenser sturio*) in Spain. The population of the river Guadalquivir: a case history and a claim for a restoration programme. *In* Acipenser, pp. 337–347. Ed. by P. Williot. Cemagref Publications.

Elvira, B., Almodovar, A., and Lobon-Cervia, J. 1991b. Recorded distribution of sturgeon (*Acipenser sturio* L., 1758) in the Iberian Peninsula and actual status in Spanish waters. Archives of Hydrobiology, 121: 253–258.

Garrido Ramos, M.A., Soriguer, M.C., de la Herran, R., Jamilena, M., Ruiz Rejon, C., Domezain, A., Hernando, J.A., and Ruiz Rejon, M. 1997. Morphometric and genetic analysis as proof of the existence of two sturgeon species in the Guadalquivir river. Marine Biology, 129: 33–39.

Gessner, J. 2000. Reasons for the decline of *Acipenser sturio* L., 1758 in central Europe, and attempts at its restoration. Instituto Español de Oceanografia, 16 (1–4): 117–126.

Grogan, C.S., and Boreman, J. 1998. Estimating the probability that historical populations of fish species are extirpated. North American Journal of Fisheries Management, 18: 522–529.

Hoffmann, R.C. 1996. Economic development and aquatic ecosystems in Medieval Europe. American Historical Review, 101: 631–669.
Jego, S., Gazeau, C., Jatteau, P., Elie, P., and Rochard, E. 2002. Les frayères potentielles de l'esturgeon européen *Acipenser sturio* L. 1758 dans le bassin Garonne-Dordogne. Méthodes d'investigation, état actuel et perspectives. Bulletin Francais de la Pêche et de La Pisciculture, 365/366: 487–505.
Kinzelbach, R. 1987. Das ehemalige vorkommen des störs, *Acipenser sturio* (Linnaeus, 1758), im Einzugsgebiet des Rheins (Chondrostei, Acipenseridae). Zeitschrift fûr Angevandte zoologie, 74(2): 167–200.
Lepage, M. 2002. Les captures par la pêcherie et l'échappement d'espèces exotiques. *In* Proceedings of the symposium « Quel avenir pour l'esturgeon européen », pp. 87–97. Epidor Publ.
Lepage, M., and Rochard, E. 1995. Threatened fishes of the world: *Acipenser sturio* Linnaeus, 1758 (Acipenseridae). Environmental Biology of Fishes, 43: 28.
Lepage, M., LeBarh, R., and Rochard, E. 1998a. Accidental catches at sea of an endangered migratory species (*Acipenser sturio*) and connections with various types of fishing. ICES CM 1998/N:24. 7 pp.
Lepage, M., Rochard, E., and Castelnaud, G. 1998b. La restauration de l'esturgeon européen (*Acipenser sturio*) et les extractions de granulats dans la Gironde. Séminaire national : Evolution naturelle et artificielle des estuaires français, Paris, 26–27 novembre 1997. Edition IFREMER, pp. 226–233.
Lepage, M., Rochard, E., and Castelnaud, G. 2000. Sturgeon *Acipenser sturio* restoration and gravel extraction in the Gironde estuary. Boletin Instituto Español de Oceanografia, 16(1–4): 175–179.
Letaconnoux, R. 1961. Note sur la fréquence de la distribution des captures d'esturgeons (*Acipenser sturio* L.) dans le Golfe de Gascogne. Revue des Travaux de l'Institut scientifique des Pêches maritimes, 25(3): 253–261.
Lochet, A. 2002. Analyse de l'intégration dans la population résiduelle de l'esturgeon européen *Acipenser sturio* d'une cohorte issue d'une reproduction en structure expérimentale. DEA thesis in Ecologie des écosystèmes aquatiques continentaux, Toulouse University. 30 pp.
Ludwig, A., Debus, L., Lieckfeldt, D., Wirgin, I., Beneckes, N., Jenneckens, I., Williot, P., Waldman, J.R., and Pitra, C. 2002. When the American sturgeon swam east. Nature, 419: 447–448.
MacDowall, R.M. 1999. Different kinds of diadromy: different kinds of conservation problems. ICES Journal of Marine Science, 56: 410–413.
Magnin, E. 1959. Répartition actuelle des Acipenseridés. Revue des Travaux de l'Institut scientifique des Pêches maritimes, 23(3): 277–285.
Magnin, E. 1962. Recherches sur la systématique et la biologie des Acipenséridés. Ann. Sta. Centr. Hydrobiol. Appl., Paris, 9: 7–242.
Mayer, N., and Lepage, M. 2001. Sauvegarde et restauration de l'esturgeon européen – action de communication et de sensibilisation. Agedra report for Life nature programme, Epidor publ. 43 pp.
Ninua, N.S. 1976. Atlanticheskij osetr reki Rioni. Metsniereba, Tbilissi, 121 pp. (in Russian).
Rochard, E. 1992. Mise au point d'une méthode de suivi de l'abondance des amphihalins dans le système fluvio-estuarien de la Gironde, application à l'étude écobiologique de l'esturgeon *Acipenser sturio*. Thèse de doctorat, Université de Rennes. I/Cemagref. 315 pp.
Rochard, E. (ed.) 2002a. Restauration de l'esturgeon européen *Acipenser sturio*. Rapport scientifique Contrat LIFE n° B – 3200/98/460, Rapport Cemagref. 300 pp.
Rochard, E. 2002b. La connaissance et le suivi de la population sauvage. *In* Proceedings of the symposium « Quel avenir pour l'esturgeon européen », pp. 47–65. Epidor Publ.
Rochard, E., Castelnaud, G., and Lepage, M. 1990. Sturgeons (Pisces Acipenseridae): threats and prospect. Journal of Fish Biology, 37(Supplement A): 123–132.
Rochard, E., Lepage, M., and Meauzé, L. 1997. Identification et caractérisation de l'aire de répartition marine de l'esturgeon européen *Acipenser sturio* à partir de déclarations de captures. Aquatic Living Resources, 10: 101–109.
Rochard, E., Lepage, M., Dumont, P., Tremblay, S., and Gazeau, C. 2001. Downstream migration of juvenile European sturgeon *Acipenser sturio* L. in the Gironde estuary. Estuaries, 24(1): 108–115.
Roule, L. 1922. Etude sur l'esturgeon du golfe de Gascogne et du bassin Girondin. Office scientifique et technique des pêches maritimes, Notes et Mémoires, 20. 12 pp.
Tabardel, M. 1994. Le point sur la situation de l'esturgeon (*Acipenser sturio* L.) en Méditerranée occidentale et possibilités de réintroduction dans le Rhône. Rennes/Arles, E.N.S.A. Rennes: 57.
Taverny, C., Lepage, M., Piefort, S., Dumont, P., and Rochard, E. 2002. Habitat selection by juvenile European sturgeon *Acipenser sturio* in the Gironde estuary (France). Journal of Applied Ichthyology, 18(4–6): 536–541
Trouvery, M., Williot, P., and Castelnaud, G. 1984. Biologie et écologie d'*Acipenser sturio*. Etude de la pêcherie. Etude n° 17, Série esturgeon n°1. Cemagref de Bordeaux, Div. ALA/AGEDRA. 79 pp.
Williot, P., Brun, R., Pelard, M., and Mercier, D. 2000. Induced maturation and spawning in an incidentally caught adult pair of critically endangered European sturgeon, *Acipenser sturio* L. Journal of Applied Ichthyology, 16: 279–281.
Williot, P., Rochard, E., Castelnaud, G., Rouault, T., Brun, R., Lepage, M., and Elie, P. 1997. Biological characteristics of European Atlantic sturgeon, *Acipenser sturio* as the basis for a restoration program in France. Environmental Biology of Fishes, 48: 359–370.
Williot, P., Rouault, T., Brun, R., Pelard, M., and Mercier D. 2002. Status of caught wild spawners and propagation of the endangered sturgeon

Acipenser sturio in France: a synthesis. International Review of Hydrobiology, 87: 515–524.

2.3 Allis shad (*Alosa alosa*)

Current status

Allis shad historically occurred along the Atlantic coast from Norway to Morocco, extending via the British Isles, the coasts of Germany, the Netherlands, Belgium, and France, and then down to Spain, Portugal, and Morocco (Blanc *et al.*, 1971; Lelek, 1980) (Figure 2.3.1). Although less abundant than in the Atlantic, Allis shad also occurred in the Western Mediterranean along the coast of Spain and especially in the Ebro River (Lozano Cabo, 1964). Its presence along the Mediterranean coast of France was rare and even doubtful (Roule, 1925; Hoestlandt, 1958).

There has been a considerable decline in abundance of *Alosa* spp. throughout their geographic range; see reviews by Taverny *et al.* (2000) and Keith *et al.* (1992). It is for this reason that these species have been included in Appendix III of the Bern Convention and in Annexes II and V of the EC Habitats Directive. Within its distribution range, *Alosa alosa* is considered extinct in three countries, critically endangered in one, endangered in six, vulnerable in two, and not evaluated or data deficient in three (Baglinière *et al.*, in press).

Alosa alosa became extinct in the River Weser at the beginning of the twentieth century, as a result of over-fishing, channelization, and the construction of dams (Busch *et al.*, 1988, 1989). In the latter part of the eighteenth century a spawning population of *Alosa alosa* existed in the River Rhine (Hoek, 1899; Redeke, 1939).

A steep decline in numbers of *A. alosa* occurred around 1900 (Figure 2.3.2). The decline of *A. alosa* was due to overfishing, barriers to their migration and destruction of their spawning habitat (de Groot, 1989; Raat, 2001). However, the interpretation of the data is complicated by the occurrence of hybrids (Redeke, 1939).

At the turn of the nineteenth century, both species of shad were present in Belgium (Anon., 1901) and a spawning population of *Alosa alosa* was present in the River Meuse (Anon., 1888). However, by 1925 *A. alosa* was no longer found in the Walloon part of the rivers Escaut and Meuse (Poll, 1947; Philippart and Vranken, 1981, 1982) as a result of over-exploitation, pollution, habitat destruction, and the building of weirs (Philippart *et al.*, 1988).

In the UK, a spawning population of *A. alosa* existed in the River Severn until the middle of the nineteenth century. Its decline has been attributed to navigation weirs constructed around 1842 (Day, 1890). Along the northern French coast, a spawning population of both species previously existed in the Seine (Vincent, 1889; Roule, 1920), becoming extinct following the construction of the Poses and Martot dams in 1887 near Rouen (Le Clerc, 1941).

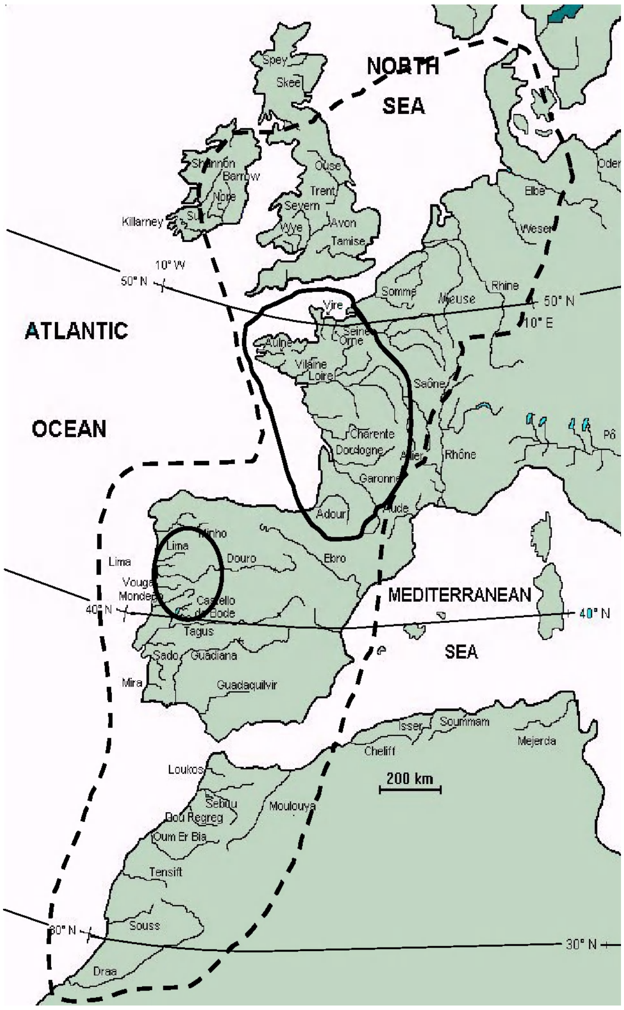


Figure 2.3.1. Historic (dashed line) and current (solid line) distribution areas of Allis shad in the Eastern Atlantic. Shown are the main rivers colonized at the end of the 19th and the beginning of the 20th century, and at the beginning of the 21st century (from Baglinière *et al.*, in press).

Of the rivers entering the Atlantic, spawning populations of *Alosa alosa* are present in the Loire, Charente, Garonne and Dordogne, Adour and Nivelle (Mennesson-Boisneau and Boisneau, 1990; Taverny, 1991; Prouzet *et al.*, 1994; Véron, 1999; Baglinière, 2000). In the Garonne and Dordogne, the original distribution of *A. alosa* had become restricted because of dams at Bazacle (1774), Mauzac (1843), and Golfech (1971). However, the construction of fish pass facilities at these obstructions since 1987 has been successful in extending access for *A. alosa* to the upper river, resulting in an increase in the size of the population, as evident from an

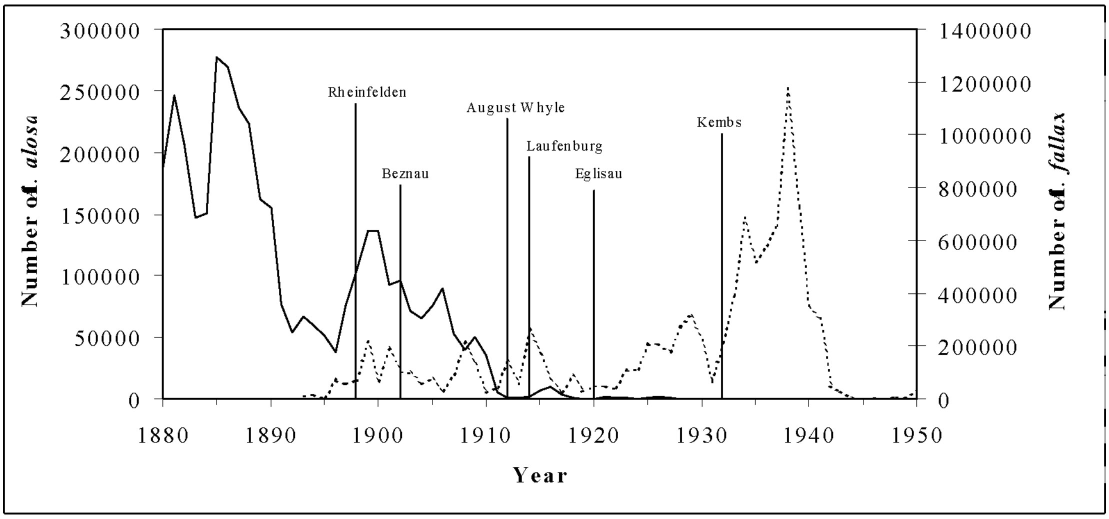


Figure 2.3.2. The catch of *Alosa alosa* (solid line) between 1880 and 1934, and of *A. fallax* (dotted line) between 1893 and 1950, from the lower Rhine (data from de Groot, 1989). Vertical lines show the dates when dams were built.

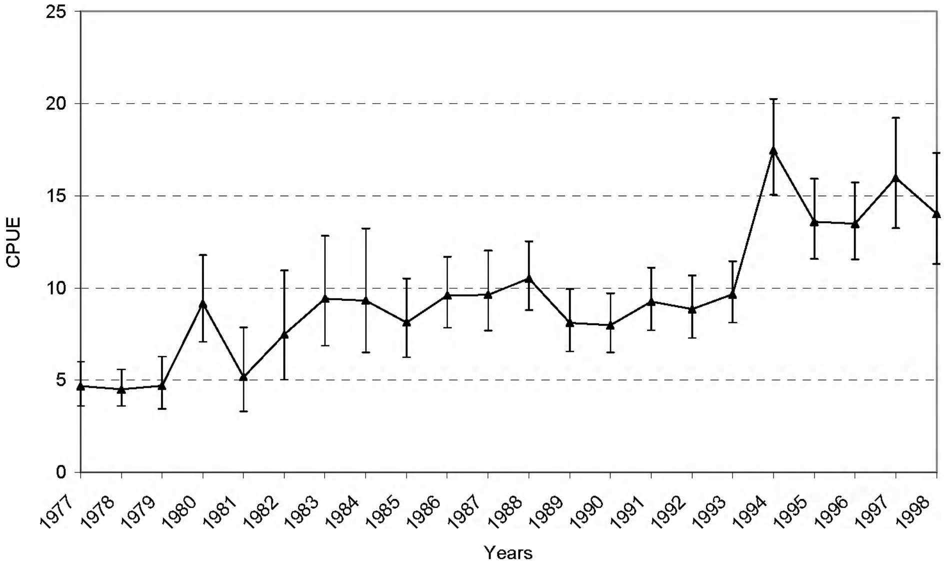


Figure 2.3.3. Annual variation in the number of *Alosa alosa* caught in the Gironde per net per day between 1977 and 1999 (Castelnaud *et al.*, 2001).

increase in the CPUE in the Gironde fishery (Figure 2.3.3).

Along the Atlantic coast of the Iberian Peninsula, spawning populations of both species of shad have been reported in the rivers Minho, Lima, Douro, Vouga, Mondego, Tagus, and the Guadiana (Capello, 1880; Regalla, 1888; Nobre, 1932; Ribeiro, 1971; Eiras 1981; Alexandrino, 1996a, 1996b; Collares-Pereira *et al.*, 2000; Costa *et al.*, 2001). However, dam construction on a number of Portuguese rivers has had a dramatic impact on populations (Costa *et al.*, 2001). In the Douro, the Crestuma-Lever dam constructed in 1985, 21 km upstream from the river mouth, has resulted in the populations of *A. alosa* virtually becoming extinct (Alexandrino, 1996b). In the rivers Tagus and Minho, the populations of *A. alosa* have declined dramatically (Figure 2.3.4) to the extent that only a residual population now exists in the Tagus (Alexandrino, 1996b). In the Tagus, this decline is associated with the construction of the Castelo de Bode and Belver dams in 1951 and 1952, respectively (Costa *et al.*, 2001). Although a Borland fish lift has been installed on the Douro at Crestuma-Lever and on the Tagus at Belver (170 km from the mouth of the Tagus), they do not appear to be effective in passing *Alosa* spp. upstream (Bochechas, 1995).

At the southern limit of their distribution in Morocco, the total annual catch of shad from Moroccan waters (including the Oued Moulouya) was in the region of 1000 t (Watier, 1918) at the start of the 20th century. However, the construction of barrages and degradation of the habitat have resulted in a number of populations becoming extinct. A spawning population of the anadromous form of *Alosa alosa* existed in the Sebou but became extinct following the construction of the barrage Idriss ler and from pollution derived from the processing

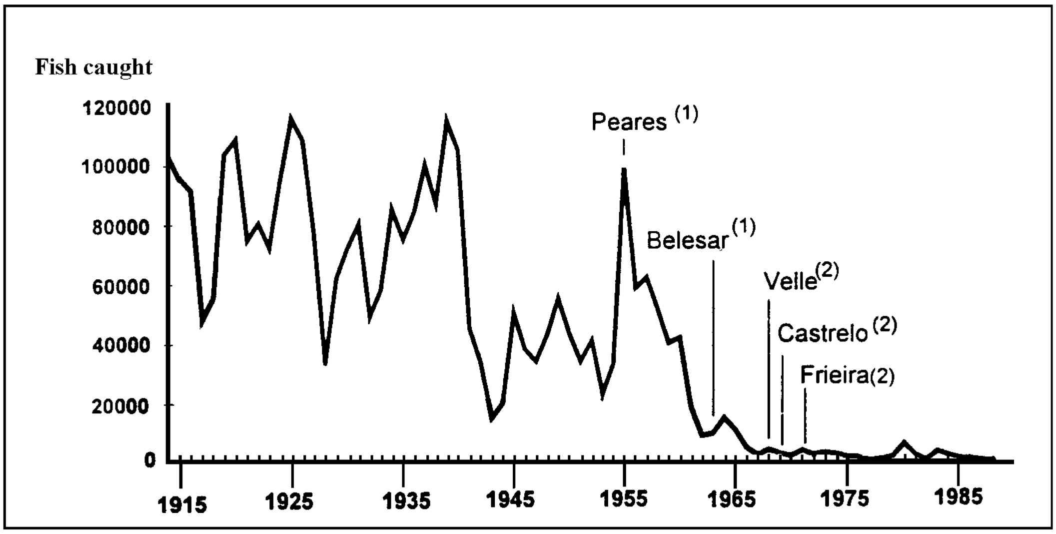


Figure 2.3.4. The catch of *Alosa alosa* between 1914 and 1988 from the River Minho, and the dates when barrages were constructed; 1 = Lugo province, 2 = Orense province (Alexandrino, 1996b).

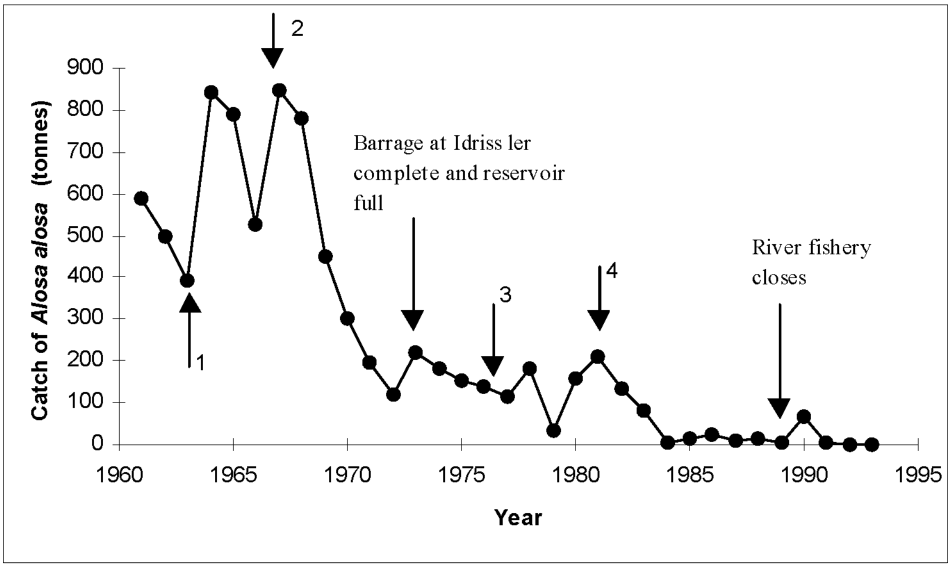


Figure 2.3.5. The combined marine and river catch of *Alosa alosa* from the Oued Sebou (Morocco) between 1961 and 1993. Numbers refer to when different sugar refineries were constructed (Sabatié, 1993).

of sugar (Figure 2.3.5, Sabatié, 1993). Other Oueds which used to support spawning populations of *Alosa alosa* were the Bou Regreg, Oum er Rbia, and the Massa. The populations have become extinct following the construction of weirs in 1968–1969, 1929, and 1973, respectively.

Conclusion in relation to the Texel-Faial criteria for the identification of threatened species and habitats

There is extensive evidence that the OSPAR area is of global importance to *Alosa alosa*. Though in the past the species had been recorded outside this area, i.e., in Morocco, the present situation is that all the remaining self-sustaining populations are confined mainly to France and Portugal and complete their life cycle within the OSPAR area. There is good evidence for a reduction in their range and in certain rivers the population (as evidenced from catches) has declined to such a level that it is extremely unlikely that a self-sustaining population still exists and the population may well be extinct. Examples, where the data can be considered reasonably robust, include the rivers Weser, Rhine and Meuse, Severn, Seine, Tagus, Minho and Sebou.

The main threats to Allis shad in Europe are obstruction of migration routes, pollution of lower river reaches, impingement at water intakes, and damage to spawning grounds. The majority of these threats take place in estuarine and freshwater environments used by the migrating fish. The construction of dams and artificial embankments prevents the fish from migrating freely (Taverny *et al.*, 2000) and can lead to hybridization with *Alosa fallax* (Boisneau *et al.*, 1992). The effect of barriers may have been exacerbated by overfishing, as fish congregate and become easier to capture below obstructions, and such a situation may well have developed on the Rhine. For fish passes to be fully effective, they must be designed with shad as the target species (Larinier and Travade, 2002), and when this is the case they can be effective (Travade *et al.*, 1998); see also Figure 2.3.3. However, there are instances where the installation of a fish pass has not proved effective for passing shad (Bochechas, 1995).

Extraction of water for irrigation can also make spawning grounds inaccessible. Impingement at water intakes provides a potential threat for the fish returning downstream and for the juveniles (Taverny, 1990). In 1986, Taverny (1990) estimated that 434,860 juvenile *Alosa alosa* were entrained by the Blayais power station in the Gironde estuary (France).

The spawning grounds themselves have been degraded by the extraction of gravel and stones from the river bed, and by modifications in water flow caused by channelling and fluctuating water levels below dams. Poor water quality is another concern affecting the fish directly (Sabatié, 1993) and indirectly through effects on their food (e.g., Berg *et al.*, 1996). A small marine fishery for Allis shad exists (Baglinière *et al.*, in press); their accidental capture during trawling or in coastal gillnets appears to be low.

At present, except for the Gironde, there is little information on the impact of a fishery on a population. In the Gironde, the present level of exploitation is considered sustainable (Martin-Vandembulcke, 1999). However, if market conditions change and there is an increase in demand for *Alosa alosa*, then there is an increased risk from overfishing.

Most of the environmental problems affecting shads are in freshwater and estuarine environments, and there is no evidence that anthropogenic activities in fully marine environments are major threats to their populations, although they are occasionally taken in marine fisheries. It is suggested that in the future both shad species should be protected, as protection measures for *A. alosa* will also afford protection to the twaite shad *A. fallax*.

References

Alexandrino, P. 1996a. Genetic and morphological differentiation among some Portuguese populations of allis shad *Alosa alosa* (L., 1758) and twaite shad *Alosa fallax* (Lacépède, 1803). Publicaciones Especiales Instituto Español de Oceanografía, 21: 15–24.

Alexandrino, P. 1996b. Estudo de populações de sável *(Alosa alosa* L.) e savelha *(Alosa fallax* Lacépède*)*. Análise da diferenciação interspecífica, substruturação e hibridaçao. [Study of allis shad *(Alosa alosa* L.) and twaite shad *(Alosa fallax* Lacépède) populations: Population structure, differentiation between the two species and hybridisation]. 185 pp. Tese de doutoramento. Universidade do Porto, Porto.

Anon. 1888. Bevattende een rapport over Ankerkuil - en Staalboomen-Visscherij. Tijdschrift der Nederlandsche Dierkundige Vereeniging, Supplement Deel II: 1–331.

Anon. 1901. Les aloses en Belgique. Pêche et Pisciculture, 12 Avril 1901. pp. 65–70.

Baglinière, J.L. 2000. Le genre *Alosa* sp. *In* Les aloses *(Alosa alosa* et *Alosa fallax* spp.*)*. Écobiologie et variabilité des populations, pp. 3–30. Ed. by J.L. Baglinière and P. Elie. INRA-CEMAGREF, Paris.

Baglinière, J.L., Sabatié, M.R., Rochard, E., Alexandrino, P., and Aprahamian, M.W. In press. The Allis shad (*Alosa alosa* Linneus, 1758): biology, ecology, range and status of populations. Transactions of the American Fisheries Society.

Berg, S., Krog, C., Muss, B., Nielsen, J., Fricke, R., Berghahn, R., Neudecker, T., and Wolff, W.J. 1996. Red List of Lampreys and Marine Fishes of the Wadden Sea. H. von Nordheim, O. Norden Andersen, and J. Thissen. *In* Red Lists of Biotopes, Flora and Fauna of the Trilateral Wadden Sea Area, 1995. Helgoländer Meeresuntersuchungen, 50(Suppl): 101–105.

Blanc, M., Banarescu, P., Gaudet, J.L., and Hureau. J.C. 1971. European inland water fish. A multilingual calatogue. FAO, Fishing News (Books) Ltd, London.

Bochechas, J. 1995. Preliminary data on Borland fish pass efficiency for non-salmonids into Portuguese large rivers. Proceedings of the International Symposium on Fishways '95 in Gifu, pp. 377–383. Gifu, Japan.

Boisneau, P., Mennesson-Boisneau, C., and Guyomard, R. 1992. Electrophoretic identity between Allis shad *Alosa alosa* L. and twaite shad, *Alosa fallax* (Lacépède). Journal of Fish Biology, 40: 731–738.

Busch, D., Haeslop, U., Scheffel, H.-J., and Schirmer, M. 1988. Fish and their environment in large European river ecosystems: The river Weser, FRG. Sciences de L'eau, 7: 75–94.

Busch, D., Schirmer, M., Schuchardt, B., and Ullrich, P. 1989. Historical changes of the River Weser. *In* Historical Change of Large Alluvial Rivers: Western Europe, pp.297–321. Ed. by G.E. Petts, H. Möller, and A.L Roux. John Wiley and Sons Ltd, Chichester.

Capello, F.B. 1880. Catálogo dos Peixes de Portugal. Catalogue Academia Real Sciencias de Lisboa.

Castelnaud, G., Rochard, E., and Le Gat, Y. 2001. Analyse de la tendance de l'abondance de l'alose *Alosa alosa* en Gironde à partir de l'estimation d'indicateurs halieutiques sur la période 1977–1998. Bulletin Français de la Pêche et de la Pisciculture, 362/363: 989–1015

Collares-Pereira, M.J., Cowx, I.G., Ribeiro, F., Rodrigues, J.A., and Rogado, L. 2000. Threats imposed by proposed water resource development schemes on the conservation of endangered fish species in the Gaudiana River Basin in Portugal. Fisheries Management and Ecology, 7: 167–178.

Costa, M.J., Almeia, P.R., Domingos, I.M., Costa, J.L., Correia, M.J., Chaves, M.L., and Teixeira, C.M. 2001. Present status of the main shads' populations in Portugal. Bulletin Français de la Pêche et de la Pisciculture, 362/363: 1109–1116.

Day, F. 1890. Notes on the Fish and Fisheries of the Severn. Proceedings of the Cotswold Naturalists Field Club, 9: 202–219.

de Groot, S.J. 1989. The former allis and twaite shad fisheries of the lower Rhine, The Netherlands. ICES CM 1989/M:19.

Eiras, J.C. 1981. Contribuição para o conhecimento da biologia de *Alosa alosa* L. Estudo de algumas modificaçoes somáticas, fisiológicas e bioquimicas durante a migraçao anádroma no Rio Douro. 228 pp. Ph.D. Thesis. University of Porto, Portugal.

Hoek, P.P.C. 1899. Neuere Lachs- und Maifisch-studien. Tijdschrift der Nederlandsche dier kundige Vereeniging, 2: 156–242.

Hoestlandt, H. 1958. Reproduction de l'Alose Atlantique (*Alosa alosa* Linné) et transfert au bassin méditerranéen. Verhandlungen, Internationale Vereineigung für theorische und angewandte Limnologie, 13: 736–742.

Keith, P., Allardi, J., and Moutou, B. 1992. Livre Rouge: Des especes menacées de poissons d'eau douce de France et bilan des introductions. Museum National d'Histoire Naturelle, Secretariat de la Faune et de la Flore, Conseil Supérieur de la Pêche, CEMAGREF and Ministère de l'Environnement humaines. *In* Les aloses *(Alosa alosa* and *Alosa fallax* spp.). Écobiologie et variabilité des populations, pp. 227–248. Ed. by J.L. Baglinière and P. Elie. INRA-CEMAGREF, Paris.

Larinier, M., and Travade, F. 2002. The design of fishways for shad. Bulletin Français de la Pêche et de la Piscicultura, 364 Supplément: 135–146.

Le Clerc, M. 1941. Note sur les essais de multiplication artificielle de l'alose dans le bassin de la Loire. Bulletin Français de Pisciculture, 123: 27–37.

Lelek, A. 1980. Les poissons d'eau douce menacés en Europe. Collection Sauvegarde de la Nature, Volume 18.

Lozano Cabo, F. 1964. El sabalo. *In* Los peces de las aguas continentales espanolas, pp. 91–95. Servicio Nacional de Pesca fluvial y Caza, Madrid.

Martin-Vandembulcke, D. 1999. Dynamique de population de la grande Alose *(Alosa alosa* L. 1758) dans le bassin versant Gironde-Garonne-Dordogne (France): Analyse et prévision par modélisation. 114 pp. + annexes. Ph.D Thesis. Institut National Polytechnique de Toulouse, Toulouse.

Mennesson-Boisneau, C., and Boisneau, P. 1990. Migration, répartition, reproduction, caracté-ristiques biologiques et taxonomie des aloses (*Alosa* sp.) dans le bassin de la Loire. 143 pp. + Annexes. Thèse de Docteur en Sciences de l'Université. University of Rennes I, Paris XII Val de Marne.

Nobre, A. 1932. Peixes de água doce de Portugal. Seperata do Boletim, 13 (2), Ministerio da Agricultura.

Philippart, J.-C., and Vranken, M. 1981. Pour la conservation de notre faune ichtyologique. Bulletin des Réserves Naturelles et Ornithologiques de Belgique**,** 28: 41–50.

Philippart, J.C., and Vranken, M. 1982. Les poissons d'eaux douce menacés en region wallone. Université de Liège, Ministère des Affaires Wallones, Leige.

Philippart, J.C., Gillet, A., and Micha, J.-C. 1988. Fish and their environment in large European river ecosystems: The River Meuse. Sciences de L'eau, 7: 115–154.

Poll, M. 1947. Poissons marins. *In* Faune de Belgique. 452 pp. Musée Royal d'Histoire Naturelle de Belgique, Bruxelles.

Prouzet, P., Martinet, J.P., and Badia, J. 1994. Caractérisation biologique et variation des captures de la grande alose (*Alosa alosa*) par unité d'effort sur le fleuve Adour (Pyrénées-atlantiques, France). [Biological characteristics and catch variation of Allis shad (*Alosa alosa*) from commercial catches in the Adour River (Pyrenées atlantiques, France)]. Aquatic Living Resources, 7: 1–10.

Raat, A.J.P. 2001. Ecological rehabilitation of the Dutch part of the River Rhine with special attention to the fish. Regulated Rivers: Research and Management, 17: 131–144.

Redeke, H.C. 1939. Über den bastard Clupea *Alosa finta* Hoek. Archives Néerlandaise de Zoologie, 3: 148–158.

Regalla, F.A.F. 1888. Relatório sobre a pesca no rio Minho em 1884. 22 pp. Ministério da marinha e Ultramar.

Ribeiro, S. 1971. Importancia económica e social da pesca nas áquas interiores. I Simpásio nacional sobre poluicao des áquas interiores, 1: 25–34.

Roule, L. 1920. Les espèces d'aloses du bassin de la Seine. Bull. Mus. Hist. Nat. Paris, 26: 610–611.

Roule, L. 1925. Les poissons des eaux douces de France. Presses Universitaires de France editor, Paris.

Sabatié, M.R. 1993. Recherches sur l'écologie et la biologie des aloses du Maroc (*Alosa alosa* Linné, 1758 et *Alosa fallax* Lacépède, 1803). Exploitation et taxinomie des populations atlantiques; bioécologie des aloses de l'oued Sebou. [Ecological and biological research on shad in Marocco (*Alosa alosa* Linné, 1758 and *Alosa fallax* Lacépède, 1803), fishing and taxonomy of Atlantic populations, bioecology of shad in Sebou River]. 326 pp. + annexes. Thèse de Doctorat en Océanologie Biologique. Université de Bretagne Occidentale, Brest.

Taverny, C. 1990. An attempt to estimate *Alosa alosa* and *Alosa fallax* juvenile mortality caused by three types of human activity in the Gironde Estuary, 1985–1986. *In* Management of Freshwater Fisheries. Proceedings of a Symposium organised by the European Inland Fisheries Advisory Commission, pp. 215–229. Göteborg, Sweden, 31 May–3 June 1988. Pudoc, Wageningen.

Taverny, C. 1991. Pêche, biologie, écologie des Aloses dans le Système Gironde-Garonne-Dordogne (Fishing, biology, ecology of shads in the system of Gironde, Garonne-Dordogne). 4, 392 pp. CEMAGREF, Bordeaux.

Taverny, C., Belaud, A., Elie, P., and Sabatié, M.R. 2000. Influence des activités humaines. *In* Les aloses (*Alosa alosa* and *Alosa fallax* spp.). Écobiologie et variabilité des populations. pp. 227–248. Ed. by J.L. Baglinière and P. Elie. INRA-CEMAGREF, Paris.

Travade, F., Larinier, M., Boyer-Bernard, S., and Dartiguelongue, J. 1998. Performance of four fish pass installations recently built on two rivers in south-west France. *In* Fish migration and fish bypasses, pp. 146–170. Ed. by M. Jungwirth, S. Schmutz, and S. Weiss. Fishing News Books-Blackwell Science Ltd, Oxford.

Véron, V. 1999. Les populations de grande Alose (*Alosa alosa*, L.) et d'Alose feinte (*Alosa fallax* Lacépède) des petits fleuves français du littoral Manche Atlantique. 81 pp. Diplome d'Agronomie Approfondie en Halieutique. ENSAR Laboratoire Halieutique and INRA, Rennes.

Vincent, P. 1889. La propagation artificielle de l'alose. Bulletin Agricole France, 7.

Watier, C. 1918. La pêche de l'alose au Maroc. Annales de Université de Grenoble, 29: 5–27.

2.4 Houting (*Coregonus lavaretus oxyrhinchus*)

Status and distribution

Habitats Directive: Priority species. Annexes II and IV: Rare.

The houting is an anadromous whitefish, which spawns in rivers from which the young migrate to the sea to develop and grow to maturity. They then return to their natal rivers to breed. The species can tolerate high salt concentrations.

Distribution, population size and status

The houting is a fish species whose known distribution is restricted to the Wadden Sea and the adjacent streams (Pihl *et al.*, 2001; Figure 2.4.1). Spawning takes place in fresh water in fast-running streams over firm bottom substrates. At the beginning of the 19th century, it was a common species in the Dutch, German, and Danish Wadden Sea. However, during the 1920s and 1930s the species gradually disappeared. In Great Britain, this species (which some authorities regard as merely a subspecies of *Coregonus lavaretus*) is only known to have occurred as a vagrant in coastal waters off the southeast coast of England and in a few estuaries there (e.g., the Colne and Medway). None has been recorded in British waters for several decades (Ratcliffe, 1977).

Previously the two taxa *C. oxyrhynchus* and *C. lavaretus* were classified as two different species, as their appearances were different, but genetic analyses have shown that there are no genetic differences between these two taxa in Northern Europe (Hansen, 1997). The houting is now classified as *C. l. oxyrhynchus.*

In the late 1980s, the population of the North Sea houting in the Wadden Sea area was nearly extinct. By 1979–1980 the River Vidå was believed to be the only

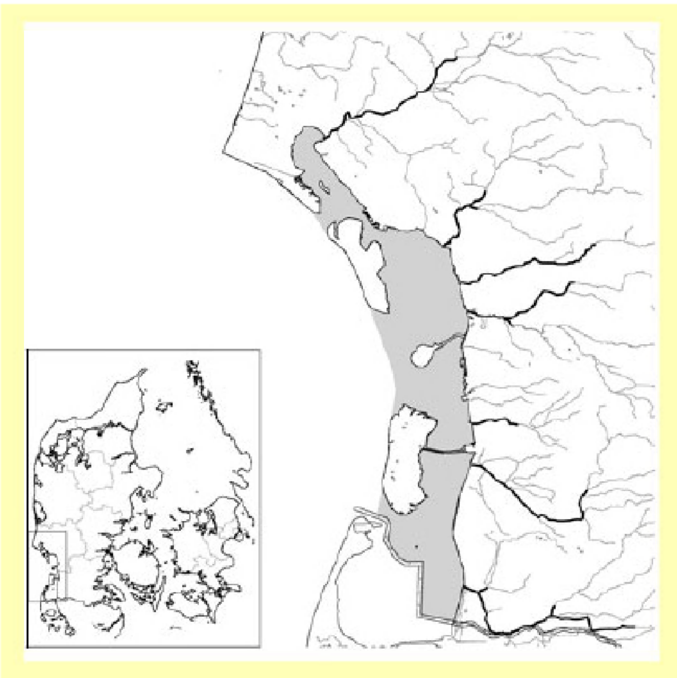


Figure 2.4.1. Known general distribution in Denmark in 1999. Streams with confirmed breeding are shown in black; the grey shaded area delimits coastal areas thought to constitute the major areas where fry develop to the adult stage (Pihl *et al.*, 2001).

stream with a stable and reproducing population of houting. Between 1987 and 1992, 1.7 million fry were stocked into the six largest streams in Denmark (Vidå, Brede Å, Ribe Å, Kongeå, Sneum Å, and Varde Å), which drain into the Wadden Sea. Pihl *et al.* (2001) reported that the Vidå is the only stream system where this action has resulted in an acceptable production of subadult houtings. In Brede Å and Varde Å, reproduction does take place, but it is uncertain whether the population there can be made self-maintaining. No self-maintaining population has been established in Sneum Å or Kongeå.

The restocking together with general protection, which included a ban on fishing, has rehabilitated the species. It is now common in the Wadden Sea area, but is still protected. The main threat to the species is the destruction of spawning grounds by engineering works, illegal fishing during the spawning migration, and the construction of artificial barriers. The conservation status of this species is primarily based on its ability to establish self-sustaining populations.

Conclusion in relation to the Texel-Faial criteria for the identification of threatened species and habitats

There is evidence of a decrease in both the area of distribution and the abundance of houting within the OSPAR maritime area, and the species is restricted to very few locations. A great part of the population is found in Danish waters. In Great Britain this species is classified as *Extinct* and in Europe, as a whole, it is considered *Endangered*. It is listed in Appendix III of the Bern Convention. In the UK, a Biodiversity Action Plan has been developed to raise awareness that the species will need protection if it becomes re-established.

The species is protected in the Danish Red List and has been protected since 1983 by the Danish Ministry of Fisheries, making it illegal to deliberately catch houting. Habitat degradation is still a major threat to the survival of the species. Essential habitats for juveniles in estuaries have been characterized, however, pollution and by-catch pose a continual threat.

References

Hansen, M.M. 1997. En lang næse til snæblen. Ferskvandsfiskeribladet, 12: 280–285.

Pihl, S., Ejrnæs, R., Søgaard, B., Aude, E., Nielsen, K.E., Dahl, K., and Laursen, J.S. 2001. Habitat and Species Covered by the EC Habitats Directive. A Preliminary Assessment of Distribution and Conservation Status in Denmark. NERI Technical Report, No. 365 (only available in electronic version).

Ratcliffe, D.A. 1977. A Nature Conservation Review: The selection of biological sites of national importance to nature conservation in Britain. Vol I–II, 401 pp, 320 pp. Cambridge University Press.

2.5 Seahorses (*Hippocampus* spp.)

Status and distribution

The short-snouted seahorse (*Hippocampus hippocampus*) and long-snouted seahorse (*Hippocampus guttulatus* (formerly *H. ramulosus*)) were each nominated by one country (Portugal). The nominations for both species cited regional importance, decline, and sensitivity.

Both of these species are distributed from the Mediterranean and Northwest Africa to the English Channel, with *H. guttulatus* extending further north along the western coasts of the British Isles and in the southern North Sea (Wheeler, 1978). *H. guttulatus* also occurs in the Black Sea. They tend to be most frequently recorded from shallow sub-tidal waters among algae, but may over-winter in deeper waters (Fishbase, 2002; Lourie *et al.*, 1999). Other habitats (e.g., macroalgae) are also occupied by seahorses and, in the eastern English Channel, *Hippocampus* spp. are occasionally caught in beam trawl surveys (Figure 2.5.1) where they tend to occur with hydroids. There are sporadic records of seahorses around the British Isles, but most are recorded along southern coasts of the UK, including the Channel Islands, and in the Wadden Sea. The size and distribution of seahorse populations in the OSPAR area are not known.

Life history characteristics

The life history characteristics of seahorses, which involve relatively protracted parental care, low fecundity, monogamy, low mobility, and small range size (Vincent, 1996; Schmid and Senn, 2002) would make them sensitive to overexploitation in areas of high relative abundance. The seahorses' low fecundity (<1000 young per year) means that populations may find it more difficult to recover from overfishing. The male seahorse undergoes a full pregnancy, as in other fishes with obligate paternal care, so taking the male will also remove the dependent offspring. Seahorses have low mobility and are site faithful. This means that any fishermen targeting and being skilled at the practice can eliminate local seahorse populations. It also means that recolonization of depopulated areas is very slow.

Seahorses have highly structured social behaviour. They form long-term faithful pair bonds that enhance their reproductive output. If one member of a pair is fished, its partner also stops reproducing for a prolonged period. When it eventually does find another mate, reproductive output of the new pair may be lower. Seahorse monogamy means that fishers finding one seahorse will search carefully for its partner, thus frequently catching both.

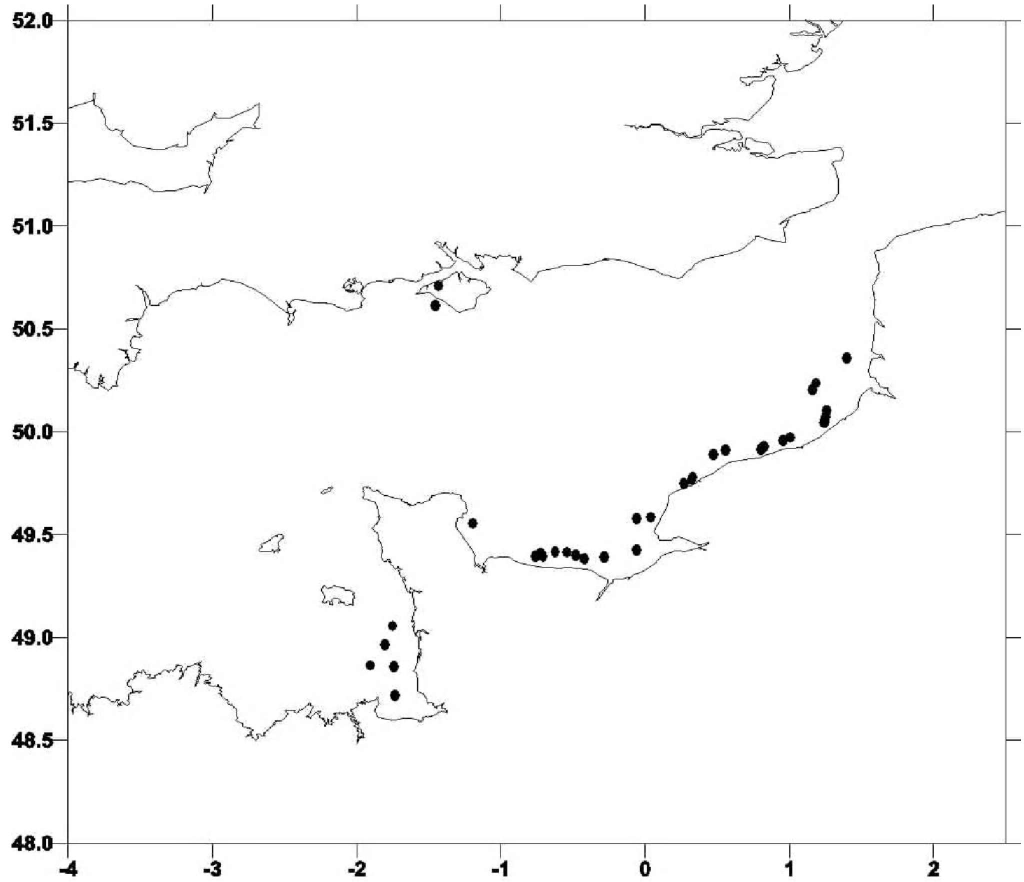


Figure 2.5.1. Occurrence of seahorses (*Hippocampus* spp.) in the eastern English Channel, as observed from beam trawl surveys (1991–2002).

The short-snouted seahorse lives in mixed habitats of macroalgae and rocky areas during the spring, summer, and early autumn. The long-snouted seahorse occupies predominantly eelgrass beds during the spring, summer, and early autumn, and migrates to deeper waters in winter. Thus, conservation of seagrass beds is essential for these species. Because seahorses live in areas along the coast, the potential for impact from human activities is great. Very few studies have been carried out on wild seahorse populations and the basic biology of the species is unknown. This lack of information makes it extremely difficult to predict how seahorse populations will be affected by exploitation. Presently, the exact size and distribution of the populations of seahorses in the OSPAR area are unknown.

Background information on international trade and conservation of seahorses

International trade in seahorses is regulated and most species are listed on Appendix II of CITES. Control of international trade in animals and plants falls under the mandate of the Convention on the International Trade in Endangered Species of Wild Fauna and Flora. CITES has been ratified by 160 nations, all committed to ensuring that international trade does not impose unsustainable pressures on wild populations of fauna and flora. Where trade must be controlled in order to ensure that use is compatible with survival of a species, then CITES includes provision for species to be listed on Appendix II of the Convention. Member countries shall issue export permits for international trade, and show that trade is not detrimental to the persistence of wild populations. Some countries would also require import permits for species included on Appendix II.

The exploitation of seahorses and their pipefish relatives for traditional Chinese medicine is large enough to threaten wild populations, causing declines in the number and size of seahorses. Seahorse biology is such that populations will be particularly susceptible to overfishing: (a) pregnant seahorses must survive if the young are to survive; (b) lengthy parental care combined with small brood size limit reproductive rates; (c) low mobility and small home ranges restrict recolonization of depleted areas; (d) sparse distribution means that lost partners are not quickly replaced; (e) strict monogamy means that social structure is easily disrupted; and (f) typically low rates of adult mortality mean that fishing exerts a relatively substantial selective pressure. Key parameters such as growth rates, longevity, and juvenile dispersal remain unstudied.

Extensive trade surveys have been carried out which have revealed that more countries are trading seahorses. A TRAFFIC trade report has revealed large and growing exploitation of seahorses for traditional medicines, curiosities, and ornamental display (Vincent, 1996). Seahorses are used in traditional medicines from many cultures to cure a variety of ailments. They are typically wild-caught, dried, and sold in pairs. At least 32 countries had traded syngnathids by 1995, but this increased to 75 countries during 1996–2001, with much of the expansion in Africa and Latin America.

The impacts of global trade on seahorse populations are considerable, and the fear is that seahorses within the OSPAR area will be targeted in the future. Combined with damage to their vulnerable inshore marine habitats, the effects on the populations could be detrimental. Many species of seahorses are also listed on the IUCN Red List.

Conservation action requires an understanding of the threats. It is therefore important to document the trade for seahorses and their relatives. Although trade regulation has been implemented, it will be insufficient for seahorse conservation. Countries should promote population assessments and undertake conservation measures for seagrass and estuarine habitats.

Conclusion in relation to the Texel-Faial criteria for the identification of threatened species and habitats

The nomination stated that there was "strong circumstantial evidence" of declines of *Hippocampus* spp., although this was based on species of *Hippocampus* from other parts of the world that are commonly traded. There are no available data to determine whether the abundance or spatial extent of this species has changed. Seagrass beds are known to be an important habitat for seahorses and declines in the extent of such habitats may have an impact on seahorse populations. There are ongoing conservation initiatives to protect such habitats.

Seahorses (Family Syngnathidae, Genus *Hippocampus*) may be exploited for medicines, marine curios, and for the aquarium trade. Worldwide, there is increased concern on the conservation and trade of seahorses (Vincent, 1996). The Convention on the International Trade in Endangered Species of Wild Fauna and Flora (CITES) lists seahorses on Appendix II (i.e., trade is controlled in order to ensure that exploitation is from sustainable sources) and many seahorse species are listed as threatened by the IUCN, and are also included on national Red Lists.

The UK is considering listing both species in Schedule 5 of the UK Wildlife and Countryside Act, 1980. The nomination for the short-snouted seahorse and the long-snouted seahorse fulfils the criteria for sensitivity and rarity. The listing could be used to limit any future expansion of the commercial fishery targeting seahorses.

References

Lourie, S.A., Vincent, A.C.J, and Hall, H.J. 1999. Seahorse: an identification guide to the world's species and their conservation. Project Seahorse, London. 214 pp.

Schmid, M.S., and Senn, D.G. 2002. Seahorses – Masters of Adaptation. Vie Milieu, 52: 201–207.

Vincent, A.C.J. 1996. The international trade in seahorses. TRAFFIC International, 163 pp.

Wheeler, A. 1978. Key to the fishes of northern Europe. Frederick Warne, London, 380 pp.

2.6 Couch's goby (*Gobius couchi*)

Status and distribution

First described as a species in 1974 by Miller and El-Tawil, the distribution of Couch's goby is uncertain. The species is known to be present at three locations in the UK and Ireland: Helford in south Cornwall; Lough Hyne, Co. Cork, Ireland; and Mulroy Bay, Co. Donegal, Ireland. More recently, it has been recorded from the Mediterranean (Iscia Island near Naples, Stefanni and Mazzoldi, 1999), the northern Adriatic (Kovacic, 2001), and the Rio Formosa coastal lagoon in southern Portugal (Ribeiro *et al.*, 2001), suggesting that the distribution of the species may be wider than first noted.

Potts and Swaby (1991) indicated that at the Helford site the population had decreased over the previous ten years. No information was available on the situation for any of the other sites.

No fishery is known to target this species. Couch's goby uses shallow coastal habitats (1–3 m depth) and is often associated with substrates made of stones, boulders or shell debris or sandy bottoms (Minchin, 1987), in holes under roots of *Cymodocea nodosa*. Thus, the species may be affected by human activity, specifically modification to its habitat and pollution, but no particular threat has been identified. It is an omnivorous species feeding on polychaetes, algae, crustaceans, and bivalves.

Conclusion in relation to the Texel-Faial criteria for the identification of threatened species and habitats

Couch's goby is a little-known species that was nominated by one country (Germany), due to its global importance, rarity, decline, and sensitivity. However, this review revealed that the status and distribution of this species are poorly known, such that even the distribution area is doubtful. From the precautionary perspective, it must therefore be considered as rare. At present, this species should be classified as naturally rare, but it may not be particularly threatened by human activities. Evidence for any decline in abundance is lacking, with the only evidence provided based on mortality following an oil spill. Such a decline is not relevant at the population level. The evidence for sensitivity cited desiccation, heavy metal pollution, and removal, yet there was no evidence that these were significant threats affecting this species.

References

Kovacic, M. 2001. The Kvarner population of *Gobius couchi* (Teleostei, Gobiidae), a fish new to the Adriatic fauna. Nat. Croat., 10(1): 1–10, Zagreb.

Miller, P.J., and El-Tawil, M.Y. 1974. A multidisciplinary approach to a new species of *Gobius* (Teleostei: Gobiidae) from southern Cornwall. Journal of Zoology, London, 174: 539–574.

Minchin, D. 1987. Fishes of the Lough Hyne Marine Reserve. Journal of Fish Biology, 31: 343–352.

Potts, G.W., and Swaby, S.E. 1991. Evaluation of the conservation requirements of rarer British marine fishes. Final report to the Nature Conservancy Council, Peterborough, UK.

Ribeiro *et al.* 2001. Study of the ichthyofauna of the Rio Formosa coastal lagoon (southern Portugal, Algarve). http://www.ualg.pt/fcma/cfrg/pdfs/ir_2001eci.pdf .

Stefanni, S., and Mazzoldi, C. 1999. The presence of Couch's goby in the Mediterranean Sea. Journal of Fish Biology, 54(5): 1128–1131.

2.7 Conclusions

ACE considered that the nominations for sturgeon, houting, and both species of turtles and seahorses met the Texel-Faial criteria and that threats in the maritime area were relevant. Sturgeon is locally important in the OSPAR area and globally, rare, very sensitive to certain human activities, and the declines are marked. Houting is of local importance, occurring only in parts of OSPAR Region II, although there are populations in the Baltic Sea. Once again, this species is sensitive and has declined. Both species of seahorse occur in the OSPAR area and in the Mediterranean, and it is unclear as to the proportion of the global population that resides in the OSPAR area. They are considered sensitive, due to their life history traits, and are perceived as rare (although they are cryptic species). Although there are no accurate data to illustrate a decline, both species are known to utilize seagrass habitats, and such habitats have declined.

ACE also felt that sea lamprey and Allis shad met the Texel-Faial criteria, although their primary threats are in estuarine and freshwater ecosystems. Whereas there is evidence that improved management can result in improved local populations, the overall populations and their spatial extent have declined.

The data were insufficient to support the view that Couch's goby is threatened and, although it may be considered rare, its rarity may be natural or apparent and not necessarily anthropogenically induced.

ACE considered that the original classification according to the Texel-Faial criteria was inconsistent and a revised version is presented in Table 2.7.1.

Table 2.7.1. Revised classification of proposed fish species under the Texel-Faial criteria.

	Global importance	**Local Importance**	**Rarity**	**Sensitivity**	**Keystone species**	**Decline**
Sturgeon (*Acipenser sturio*)	✓	✓	✓	✓		✓
Houting (*Coregonus lavaretus oxyrhinchus*)	?	✓	✓	✓		✓
Short-snouted seahorse (*Hippocampus hippocampus*)	?	?	✓	✓		?
Long-snouted seahorse (*Hippocampus guttulatus*)	?	?	✓	✓		?
Allis shad (*Alosa alosa*)	✓		✓	✓		✓
Sea lamprey (*Petromyzon marinus*)	✓			✓		✓
Couch's goby (*Gobius couchi*)	?	?	?			?

3 Sea turtles

3.1 Introduction

Nesting sites

Beach nourishment impacts turtles by burial of nests, disturbance to nesting turtles, and changes in sand compaction and temperature, which may affect embryo development. Artificial lighting can cause disorientation or misorientation of both adults and hatchlings. Turtles are attracted to light, coming out of the ocean to go towards a light source, thus increasing their chances of death or injury. In addition, as nesting females avoid areas with intense lighting, highly developed areas may cause problems for turtles trying to nest. A serious threat from nighttime use of a beach is the disturbance of nesting females. Heavy utilization of nesting beaches by humans may also result in lowered hatchling success due to sand compaction. The placement of physical obstacles on a beach can hamper or deter nesting attempts as well as interfere with incubating eggs and the sea approach of hatchlings. The use of off-road vehicles on beaches is a serious problem in many areas. It may result in decreased hatchling success due to sand compaction, or directly kill hatchlings. Tire ruts may also interfere with the ability of hatchlings to get to the ocean. The invasion of a nesting site by non-native beach vegetation can lead to increased erosion and destruction of a nesting habitat. Neither the loggerhead turtle nor the leatherback turtle nests in the OSPAR region, although the loggerhead turtle has important nesting beaches in the Mediterranean Sea and both species have minor nesting grounds along the western coast of Africa (Márquez, 1990).

By-catch

By-catch is an important factor affecting sea turtles (e.g., Hodge, 1979; Yeung, 1999; Pierpoint, 2000). Most by-catch records involve the leatherback turtle (94% of records identified to species), the turtle species most frequently reported from UK and Irish waters. The most common method of incidental capture for leatherback turtles is entanglement in ropes, particularly those used in pot fisheries targeting crustaceans and whelks. Rope entanglement occurs predominantly between July and October, on the north, west, and southwest coasts of the UK and the south and west coasts of Ireland. Leatherbacks have also been caught and drowned in pelagic driftnets and by-catch has been documented in the Northwest Atlantic pelagic longline fisheries (e.g., Witzell, 1984, 1996). However, the significance of marine turtle by-catch in the OSPAR area is not known. Leatherback turtles are globally endangered and Spotila *et al.* (1996) suggest that the impact of by-catch on Atlantic leatherback populations may be unsustainable. The threat of by-catch therefore encompasses many fishing methods and may affect marine turtles throughout their range, close inshore as well as in deep-water pelagic fisheries.

Turtles are taken by gillnet fisheries in the Atlantic and Gulf of Mexico, but the number is currently not known. Several thousand vessels are involved in hook and line fishing for various coastal species. The capturing of turtles is not uncommon, but the number is currently not known. Pound net fisheries are primarily a problem in waters off Virginia and North Carolina, however generally turtles are released alive. From 1978–1981, 330 turtles were captured in the Atlantic and Gulf of Mexico EEZ in the Japanese tuna longline fishery. Due to expansion of this fishery, it may have a large impact on turtle recovery.

In European drifting longline fishery, a total of 23 turtle catches were observed in the Greek monitoring programme (22 loggerhead *Caretta caretta* and one leatherback *Dermochelys coriacea*), 220 turtles in the Italian programme (218 loggerheads and two green turtles *Chelonia mydas*), and 2,127 turtles in the Spanish programme (2,125 loggerheads and two leatherbacks). Turtle catch rates were highly variable depending on the fishing areas and fishing seasons, being higher during quarters 2 and 3 in Spain. Set operations as well as gear characteristics also affected the outcome (Laurent *et al.*, 2001).

In the OSPAR area the greatest threat appears to come from by-catch in fishing gear, most often in fishing lines, including those used in trap fisheries. Loggerheads and leatherbacks also ingest a wide variety of marine debris such as plastic bags, plastic and styrofoam pieces, tar balls, balloons, and raw plastic pellets. Effects of consumption include interference in metabolism or gut function, even at low levels of ingestion, as well as absorption of toxic by-products. The impact on the population of the effects of these contaminants has not been determined. In areas where recreational boating and ship traffic are intense, propeller and collision injuries are not uncommon. Sea turtles in general are also at risk when encountering an oil spill, which affects respiration, skin, blood chemistry, and salt gland functions. Pesticides, heavy metals, and polychlorinated biphenyls (PCBs) have been detected in turtles and eggs but their effect is unknown.

The UK government has, as an obligation as signatory to the Rio Convention, developed a Biodiversity Action Plan (BAP) (Biodiversity: The UK Action Plan (DOE, 1994)) for turtles, which also partly fulfils requirements under the EC Habitats Directive 92/43/EEC in relation to marine turtles. The biodiversity plan addresses the need to understand the use and importance of UK waters to turtles. Furthermore, by understanding the relative importance of the OSPAR area to marine turtles, implementation of international conservation measures can be facilitated.

3.2 Loggerhead turtle (*Caretta caretta*)

Current status

The loggerhead turtle is found in temperate and subtropical waters throughout most of the world, but can range far north and south. In the Western Atlantic, they are found from Newfoundland to Argentina (Frazer, 1995). The southern Florida loggerhead turtle subpopulation, which is the largest in terms of nesting females, appears to be stable or may be increasing. The northern subpopulation, which breeds on the coasts north of Florida, has declined since the 1970s but may now have stabilized. This species is the most common turtle in the Mediterranean, with most nesting at sites in Greece, Turkey, and Tunisia (Argano and Baldari, 1983; Groombridge, 1990). Individuals from Atlantic populations are also present in the western Mediterranean during the spring and summer (Laurent and Lescure, 1995). Loggerheads do not nest in the OSPAR area, but individuals from the Mediterranean and the western population forage in OSPAR waters outside the breeding season (June to August).

Loggerhead turtle populations are not monitored everywhere, but of those populations that may use the OSPAR area, numbers in Honduras, Mexico, Israel, Turkey, Bahamas, Cuba, Greece, and Panama have been declining. However, since loggerheads take approximately 20–30 years to mature, the effects of elevated juvenile/subadult mortality may not become apparent for up to 30 years, the approximate age of maturation and first nesting. The most significant threats to loggerhead turtle populations are coastal development and tourism, commercial fisheries (particularly shrimp trawling in nearshore areas), and various forms of pollution.

Conclusion in relation to the Texel-Faial criteria for the identification of threatened species and habitats

The data provided in the nomination for loggerhead turtles meets the Texel-Faial criteria for declining and threatened species, although some available data on by-catch should be added for the OSPAR area.

In UK waters there have been a small number of records of loggerhead turtles, although most specimens are thought to have been carried north by adverse currents from their usual habitats (Carr, 1987; Penhallurick, 1991; Mallinson, 1991).

3.3 Leatherback turtle (*Dermochelys coriacea*)

Current status

Leatherback turtles breed circumglobally within latitudes between approximately 40°N and 35°S, but range widely to forage in temperate and boreal waters outside the nesting season (Eckert, 1995). They are the only species of marine turtle to have adapted to life in cold water (see, for example, Greer *et al.*, 1973; Goff and Stenson, 1988). Leatherbacks have been recorded to 71°N in the OSPAR area (Prichard and Trebbau, 1984). The total number of leatherbacks nesting worldwide in 1995 was estimated at 34,529 females (±8,000 95% confidence interval) (Spotila *et al.*, 1996). About 80% of these animals were reported from sites in the Atlantic. Within this region, the largest nesting aggregations occur in French Guiana (Fretey and Girondot, 1989) and Surinam in northern South America, and in Gabon on the West African coast. There are other important nesting sites in the Caribbean (particularly Trinidad, the Dominican Republic, and the U.S. Virgin Islands) and leatherbacks also nest annually in southern Florida. In French Guiana, the nesting season extends from March to mid-August (Girondot and Fretey, 1996). Using data from a number of colonies, Spotila *et al.* (1996) assumed an inter-nesting interval of two and a half years. The leatherback is not believed to be nesting in the Mediterranean, but is present in the region throughout the year (Camiñas, 1998). Leatherback numbers are declining rapidly throughout their range (Spotila *et al.*, 1996). Populations in the Pacific and Indian Oceans have crashed dramatically in recent years (Eckert, 1997). Some important Atlantic colonies appear stable (French Guiana/Surinam: Girondot and Fretey, 1996). Loggerheads and leatherbacks accounted for 52% and 42% of observed animals, respectively (Johnson *et al.*, 1999). Observed mortality ranged from 0–60 each year.

Long-distance migration has been documented from tag returns and satellite telemetry. Turtles tagged in French Guiana have been recorded in Europe and north and west Africa (Girondot and Fretey, 1996). An indication of the origin of some leatherbacks recorded in British waters was provided by a female turtle found in Carmarthen Bay, South Wales, in September 1997, that had previously nested and been tagged in French Guiana (R. Penrose, Marine Environmental Monitoring, pers. comm.). Satellite transmitters placed on two leatherbacks in Trinidad by Eckert (1998) functioned successfully for twelve months. These turtles initially swam northeast beyond Barbados before diverging. One turtle remained in the central Atlantic until the end of November before migrating directly to the African coast. The second animal swam east and then north into the Bay of Biscay. There are distinct seasonal peaks in the occurrence of leatherback turtles in northern waters. Around the UK, most turtles are reported between August and October (Gaywood, 1997; Godley *et al.*, 1998).

Leatherback turtles feed primarily on jellyfish. Their diet in temperate and boreal waters is known to include cnidarians (siphonophores as well as medusae) and tunicates (salps, pyrosomas) (den Hartog and van Nierop, 1984; Davenport and Balazs, 1991). In UK and Irish waters, they are often reported in the vicinity of jellyfish swarms, and there are several observations of leatherbacks feeding on jellyfish at the surface.

Conclusion in relation to the Texel-Faial criteria for the identification of threatened species and habitats

The data provided in the nomination for leatherback turtles meets the Texel-Faial criteria for declining and threatened species, although some available data on by-catch should be added for the OSPAR area.

References

Argano, R., and Baldari, F. 1983. Status of western Mediterranean sea turtles. Rapports et procès verbaux des réunions - Commission internationale pour l'exploration scientifique de la mer Méditerranée, 28(5): 233–235.

Camiñas, J.A. 1998. Is the leatherback *(Dermochelys coriacea* Vandelli, 1761) a permanent species in the Mediterranean Sea? *In* Proceedings of the XXXV Congress CIESM, Dubrovnik.

Carr, A.F. 1987. New perspectives on the pelagic stage of marine turtle development. Conservation Biology, 1(2): 103–121.

Davenport, J., and Balazs, G.H. 1991. “Fiery bodies”– are pyrosomas an important component part of the

diet of leatherback turtles? British Herpetological Society Bulletin, 31: 33–38.
den Hartog, J.C., and van Nierop, M.M. 1984. A study of the gut contents of six leathery turtles *Dermochelys coriacea* (Linnaeus) (Reptilia: Tesudines: Dermochelydae) from British waters and from the Netherlands. Zoologische Verhandlingen, 200. Rijksmuseum van Natuurlijke Historie, Leiden.
Eckert, K.L. 1995. Leatherback sea turtle, *Dermochelys coriacea*. *In* National Marine Fisheries Service and U.S. Fish and Wildlife Service status reviews for sea turtles listed under the Endangered Species Act of 1973, pp. 37–75. Ed. by P.T. Plotkin. National Marine Fisheries Service, Silver Spring, Maryland, USA.
Eckert, S.A. 1997. Distant fisheries implicated in the loss of the world's largest leatherback nesting population. Marine Turtle Newsletter, 78: 2–7.
Eckert, S.A. 1998. Perspectives on the use of satellite telemetry and other electronic technologies for the study of marine turtles, with reference to the first year-long tracking of leatherback sea turtles. *In* Proceedings of the 17th Annual Sea Turtle Symposium, pp. 46–48. Ed. by S.P. Epperly and J. Braun. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-SEFSC-415.
Frazer, N.B. 1995. Loggerhead sea turtle, *Caretta caretta*. *In* National Marine Fisheries Service and U.S. Fish and Wildlife Service status reviews for sea turtles listed under the Endangered Species Act of 1973, pp. 1–23. Ed. by P.T. Plotkin. National Marine Fisheries Service, Silver Spring, Maryland, USA.
Fretey, J., and Girondot, M. 1989. L'activité de ponte de la tortue luth, *Dermochelys coriacea* (Vandelli 1761), pendant la saison 1988 en Guyane Française. Revue d'Ecologie–la Terre et la Vie, 44: 261–274.
Gaywood, M.J. 1997. Marine turtles in British and Irish waters. British Wildlife, 9(2): 69–77.
Girondot, M., and Fretey, J. 1996. Leatherback turtle, *Dermochelys coriacea*, nesting in French Guiana, 1978–1995. Chelonian Conservation and Biology, 2: 204–208.
Godley, B., Gaywood, M., Law, R., McCarthy, C., McKenzie, C., Patterson, I., Penrose, R., Reid, R., and Ross, H. 1998. Patterns of Marine Turtle Mortality in British Waters 1992–96 with reference to tissue contaminant levels. Journal of the Marine Biological Association of the United Kingdom, 78: 973–984.
Goff, G.P., and Stenson, G.B. 1988. Brown adipose tissue in leatherback sea turtles: a thermogenic organ in an endothermic reptile? Copeia, 1988: 1071–1075.
Greer, A.E., Lazell, J.D., and Wright, R.M. 1973. Anatomical evidence for counter-current heat exchanger in the leatherback turtle *(Dermochelys coriacea*). Nature, 244: 181.
Groombridge, B. 1982. The IUCN Amphibia – Reptilia Red Data Book Part 1: Testudines, Crocodylia, Rhynchocephalia., International Union for the Conservation of Nature, Gland, Switzerland.
Groombridge, B. 1990. Marine turtles in the Mediterranean: distribution, population status, conservation. Report to the Council of Europe, Environmental Conservation and Management Division, Strasbourg.
Hodge, R.P. 1979. *Dermochelys coriacea schlegeli* (Pacific leatherback) USA: Alaska. Herpetological Review, 10(3): 102.
Hoey, J.J. 1997. A summary of pelagic longline-sea turtle interactions based on U.S. observer data. *In* Proceedings of the 17th Annual Sea Turtle Symposium. Ed. by S.P. Epperly and J. Braun. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-SEFSC-415: 209–212.
Johnson, R., Yeung, C., and Brown, C.A. 1999. Estimates of marine mammal and marine turtle by-catch by the U.S. Atlantic pelagic longline fleet in 1992–97. NOAA Technical Memorandum NMFS-SEFSC-418.
Laurent, L., Camiñas, J.A., Casale, P., Deflorio, M., De Metrio, G., Kapantagakis, A., Margaritoulis, D., Politou, C.Y., and Valeiras, J. 2001. Assessing marine turtle by-catch in European drifting longline and trawl fisheries for identifying fishing regulations. Project EC-DG Fisheries 98-008. Joint project of BIOINSIGHT, IEO, IMBC, STPS and University of Bari. Villeurbanne, France, 267 pp. (C.E.E.P.), 12: 76–90.
Laurent, L., and Lescure, J. 1995. Attempt of spatial-temporal pattern distribution of Loggerhead Turtle in the Mediterranean. Scientia Herpetologica, 1995: 324–327.
Lazar, B., and Tvrtkovic, N. 1997. Results of marine turtle research and conservation program in Croatia. *In* Proceedings of the 17th Annual Sea Turtle Symposium. Ed. by S.P. Epperly and J. Braun. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-SEFSC-415.
Mallinson, J.J. 1991. Stranded juvenile loggerheads in the United Kingdom. Marine Turtle Newsletter, 54: 14–16.
Márquez, M.R. 1990. FAO Species catalogue, Volume 11: Sea turtles of the world. An annotated and illustrated catalogue of sea turtle species known to date. FAO Fisheries Synopsis No. 125, Vol. 11. 81 pp.
Penhallurick, R.D. 1991. Turtle occurrences off Cornwall and Scilly in 1990 with a note on newly discovered reports of earlier date. Zoological Cornwall and the Isles of Scilly, 1: 6–10.
Pierpoint, C. 2000, By-catch of marine turtles in UK and Irish waters. JNCC Report No 310. Peterborough, UK.
Plotkin, P.T. 1995. National Marine Fisheries Service and U.S. Fish and Wildlife Service status reviews for sea turtles listed under the Endangered Species Act of 1973. National Marine Fisheries Service, Silver Spring, Maryland, USA.
Prichard, P.C.H., and Trebbau, P. 1984. The turtles of Venezuela. Society for the Study of Amphibians and Reptiles.
Reynolds, D.P., and Sadove, S.S. 1997. Wild captures of sea turtles in New York: species composition shift

examined. *In* Proceedings of the 17th Annual Sea Turtle Symposium, pp. 269–270. Ed. by S.P. Epperly and J. Braun. U.S Department of Commerce. NOAA Technical Memorandum NMFS-SEFSC-415.

Spotila, J.R., Dunham, A.E., Leslie, A.J., Steyermark, A.C., Plotlin, P.T., and Paladino, F.V. 1996. Worldwide population decline of *Dermochelys coriacea*: are leatherback turtles going extinct? Chelonian Conservation and Biology, 2: 209–222.

Witzell, W.N. 1984. The incidental capture of sea turtles in the Atlantic U.S. Fishery Conservation Zone by the Japanese tuna longline fleet, 1979–81. Marine Fisheries Review, 46: 56–58.

Witzell, W.N. 1996. The incidental capture of sea turtles by the U.S. pelagic longline fleet in the western Atlantic Ocean. *In* Pelagic longline fishery–sea turtle interactions, pp. 32–33. Ed. by P. Williams, P.J. Anninos, P.T. Plotkin, and K.L. Salvini. Proceedings of an industry, academic and government experts, and stakeholders workshop, Silver Spring, Maryland, 24-25 May 1994. U.S. Department of Commerce, NOAA Technical Memorandum. NMFS-OPR-7.

Yeung, C. 1999. Estimates of marine mammal and marine turtle by-catch by the U.S. Atlantic pelagic longline fleet in 1998. NOAA Technical Memorandum NMFS-SEFSC-430.

4 Marine mammals

This section contains the detailed assessments of the data used in the choice of marine mammal species for the draft OSPAR Priority List of Threatened and Declining Species and Habitats. These assessments served as the basis for the ICES advice contained in Section 5 of this report, and are reproduced here as a supplement to that advice. For each marine mammal species considered, the background material is a quotation from Document BDC 03/3/2f “Mammals”, which was presented at the 2003 meeting of the OSPAR Biodiversity Committee (BDC); this material is shown in italics. The references that were reviewed in this document and subsequently considered by the OSPAR Biodiversity Committee have been indicated by an asterisk (*); the other references have been provided during the ICES review.

4.1 Bowhead whale (*Balaena mysticetus*)

a) Description

Bowhead whales inhabit Arctic and sub-Arctic waters between 55° and 80°N. There are believed to be four different stocks; Spitzbergen, Hudson Bay/Davis Strait, Bering/Chukchi/Beaufort Seas, and Okhotsk Sea. The animals migrate to northerly feeding grounds in spring and summer, returning to the southern parts of their range in late autumn (Christensen et al., *1992). The Spitzbergen stock is found in the waters around Greenland, Norway, and Russia but centred in the Greenland and the Barents Seas (IUCN, 2002).*

<u>Decline</u> *Before hunting started in the 17th century, the population of the Spitzbergen stock of the bowhead whale was estimated to be about 25,000 (Klinowska, 1991). Populations were quickly depleted because of the ease with which this species could be caught. Today there are believed to be only a few tens of individuals (only 24 sightings, including one dead animal, between 1958 and 1983) (Klinowska, 1991; Zeh* et al., *1993). Sightings in the Russian region of the Arctic suggest that there may be more whales in this area, but it is unclear whether these are a few remaining individuals from the original Spitzbergen stock or immigration from another stock.*

<u>Sensitivity</u> *Ice-associated animals, such as the bowhead whale, may be sensitive to changes in Arctic weather, sea-surface temperatures, or ice extent. Like other marine mammals they are generally characterized by low annual mortality and long life spans. There are believed to be less than 50 mature individuals in the Spitzbergen stock, which makes the whole stock very vulnerable to extinction.*

Cetaceans use sound to provide information about the physical environment, to communicate between individuals and for the detection of potential prey. Baleen whales, such as the bowhead, emit low-frequency sound that can travel hundreds of kilometres (Evans, 2000). This makes them sensitive to acoustic disturbance from military activities such as naval sonars (particularly low-frequency acoustics), as well as other sources such as seismic exploration. The whales will be particularly vulnerable if the zone of influence coincides with migration and breeding areas.

<u>Threat</u> *In the past, the main threat to this species was commercial whaling, whereas today it is pollution. Oil pollution is of particular concern because oil spilled in polar regions tends to accumulate at the ice edges, the preferred habitat of these whales. One of their main methods of feeding involves skimming the water at the surface, making them more likely to ingest oil.*

Synthetic toxins such as DDT and PCBs are another threat. High levels of these compounds have been found in the blubber of several whale species. Although the detrimental effects of chlorine compounds on whales has not been proven, birth abnormalities have been reported in seals in association with high levels of these chemicals. The population may also be exposed to radionuclides in the food chain in Arctic waters. Acoustic disturbance from shipping, military and research activities adds to the pressures on this species.

Any shifts in regional weather patterns which affect sea-surface temperature and the extent of sea ice, are another potential threat but it is not possible to make reliable predictions of the effects of Arctic climate change on bowhead whales at the present time.

<u>Relevant additional considerations</u> *Data from past whaling activities in the Arctic confirm that large numbers of bowhead whale were taken by whalers. There is some uncertainty about the precise size of the population today as the species is very rare.*

<u>Changes in relation to natural variability</u> The large numbers of bowhead whales that were fished during earlier centuries will have masked any changes in the population caused by natural variability. With such a small number remaining, natural variability may however become a major contributory factor in its local extinction.

<u>Threat and link to human activities</u>

Relevant human activity: Shipping and navigation, military activities; fishing, hunting, harvesting, research. Category of effect of human activity: Physical – Noise disturbance, Biological – removal of target species, removal of non-target species, physical damage to species.

Commercial whaling, and therefore human activity, is known to have caused the significant decline of the bowhead whale. Current threats from poor water quality and acoustic disturbance are also linked to human activities.

<u>Management considerations</u> All states whose waters this species is found in are members of the International Whaling Commission (IWC), and two of them (Norway and Greenland) are also members of the North Atlantic Marine Mammal Commission (NAMMCO). The IWC has banned commercial whaling of the bowhead whale since 1975, however, some aboriginal whaling does take place. Apart from protection from whaling, other measures that would help safeguard this species are more indirect such as minimizing the risk of marine pollution and ensuring a high water quality in the Arctic. OSPAR does not deal with whaling issues directly, but can communicate an opinion on it to the IWC and members of the North Atlantic Marine Mammal Commission (NAMMCO).

The Spitzbergen stock of bowhead whale has been classified as Critically Endangered by IUCN (IUCN, 2002).

b) Literature used (*below)

The literature used is adequate.

c) Literature interpretation

A good interpretation has been made of the literature.

d) ICES conclusions

The OSPAR assessment of the status and threats facing this species is good.

Location: The species occurs in Region I of the OSPAR area.

Decline: There is good evidence of decline in the past.

Threats: There is no evidence of current direct threat, but indirect threats such as pollutant effects may be present.

e) ICES overall evaluation

ICES finds that there is good evidence of decline, but rather little evidence of direct threat currently.

References

*Christensen, I., Haug, T., and Øien, N. 1992. Seasonal distribution, exploitation and present abundance of stocks of large baleen whales (Mysticeti) and sperm whales (*Physeter macrocephalus*) in Norwegian waters. ICES Journal of Marine Science, 49: 341–355.

*Evans, P.G.H. 2000. Biology of cetaceans of the North East Atlantic (in relation to seismic energy). *In* Proceedings of the Seismic and Marine Mammals Workshop, 25–28 June 1998. Ed. by M.L. Tasker and C. Weir.

*IUCN. 2002. 2002 IUCN Red list of threatened species. IUCN, Gland, Switzerland.

*Klinowska, M. 1991. Dolphins, porpoises and whales of the world. The IUCN Red Data Book. IUCN, Gland, Switzerland.

Ridgway, S.H., and Harrison, R. (eds.). 1985. Handbook of marine mammals. Vol. 3. The sirenians and baleen whales. Academic Press, New York. 362 pp.

*Zeh, J.E., Clark, C.W., George, J.C., Withrow, D., Carroll, G.M., and Koski, W.R. 1993. Current population size and dynamics. *In* The bowhead whale, pp. 409–489. Ed. by J.J. Burns, J.J. Montagu, and C.J. Cowles. Society for Marine Mammalogy, Special Publication No. 2. Lawrence, Kansas.

4.2 Blue whale (*Balaenoptera musculus*)

a) Description

The blue whale is found in all major oceans of the world. There are considered to be two stocks in the North Atlantic and it is the east Atlantic stock that occurs in the OSPAR Maritime Area. The migration patterns of this stock are poorly known. Some blue whales are known to winter off the Azores and Cape Verde Islands. In spring, blue whales migrate to the productive feeding grounds around Iceland, in the Barents Sea and around Spitzbergen.

<u>Decline</u> The blue whale has been severely depleted throughout its range. Whaling during the late 1800s and early 1900s targeted stocks in the North Atlantic and the North Pacific and then moved to other areas leading to a drastic reduction of the population throughout the world. The North Atlantic stock is estimated to have been made up of around 3,500 whales in Northern Norway and 10,000 in the Denmark Strait (FAO, 1978; Yochem and Leatherwood, 1985). Large-scale sightings surveys in the North Atlantic in 1987 and 1989 gave estimates of the population around Iceland as 442 and 878, respectively, with very few observations in other parts of the survey area (i.e., off Norway, Greenland, the Faeroes, and

Spain) (Gunnlaugsson and Sigurjónsson, 1990; Sigurjónsson and Víkingsson, 1997).

There are no agreed figures for the population of the blue whale in the Northern Hemisphere at the present time. The IWC only makes an estimate for blue whales in the southern hemisphere and Randall et al. *(2002) have recently suggested that there are perhaps a few hundred to a thousand blue whales remaining in the North Atlantic.*

Keystone species The blue whale is a baleen whale that feeds almost exclusively on a few species of euphausiids and copepods in highly productive polar waters. There is evidence to suggest that they also feed on shallow banks in the Azores before resuming migratory movements and where they probably have a significant impact on plankton numbers, consuming around 2–4 tonnes of food a day.

Sensitivity Like other cetaceans, the blue whale has a low reproductive rate and late age of maturity. This means that recovery of depleted populations will take many decades rather than years. Cetaceans use sound to provide information about the physical environment, to communicate between individuals, and for the detection of potential prey. Baleen whales, such as the blue whale, emit low-frequency sound that can travel hundreds of kilometres. This makes them sensitive to acoustic disturbance from military activities such as naval sonars (particularly low-frequency acoustics), as well as other sources such as seismic exploration. The whales will be particularly vulnerable if the zone of influence coincides with migration and breeding areas. In the case of the blue whale, this would include the edge of the continental shelf that may be an important migration route for this species (Evans, 2000).

Threat The blue whale was the preferred target of modern whalers because of its size and, once they could be taken and processed on factory ships, they were hunted in all the world's oceans. Catches peaked in 1930–1931 when nearly 30,000 were taken worldwide. It has also been estimated that over 280,000 blue whales (including pygmy blues) were taken between 1924–1925 and 1970–1971, mostly in the Southern Hemisphere (Chapman (1974) in Klinowska, 1991). Commercial whaling was therefore the overriding threat to this species until it was banned in 1964. Current threats come from acoustic disturbance and habitat degradation. Depletion of food resources is an issue in the Antarctic where krill are harvested. This is not the case in the Arctic, however, there are also other influences on krill abundance and therefore it is not clear if this is a threat to blue whales in the Arctic. Boat collisions also pose some threat to the whales during their spring and autumn migrations.

Relevant additional considerations Data are available on the numbers of whales taken during the period when they were subject to commercial exploitation. Since then, sightings data have been collected to determine population size and trends. Given the current rarity of the species, with the exception of a few areas, the population density is too thin to enable any recovery to be detected from surveys except over a very long period (Klinowska, 1991).

Changes in relation to natural variability The large numbers of blue whales that were taken by commercial whalers will have masked any changes caused by natural variability. With such a small number remaining, natural variability may however become a major contributory factor in any further decline.

Threat and link to human activities

Cross-reference to checklist of human activities in OSPAR MPA Guidelines.

Relevant human activity: Shipping and navigation; military activity; fishing, hunting, harvesting; research. Category of effect of human activity: Physical – Noise disturbance. Biological – removal of target species, removal of non-target species, physical damage to species.

Commercial whaling was undoubtedly the cause of the decline in blue whale numbers in the last century and therefore there was a clear link between the threat to this species and human activities. Today the threats that may lead to further decline or failure to recover are more indirect unless whaling resumes. They include marine pollution, poor water quality, acoustic disturbance, and collisions with vessels.

Management considerations The population was severely depleted before it was given protection by the IWC in 1964 and, while it is too rare to be the main target species of any fishery, it is vulnerable to illegal whaling. OSPAR does not deal with whaling issues directly but can communicate an opinion on it to the IWC and members of the North Atlantic Marine Mammal Commission (NAMMCO). Management measures need to be geared towards enabling the recovery of the population and, apart from direct protection, this could include actions to minimize acoustic disturbance.

The IUCN has classified the blue whale as an endangered species (IUCN, 2002).

b) Literature used (*below)

The literature used is adequate.

c) Literature interpretation

A good interpretation has been made of the literature.

d) ICES conclusions

The OSPAR assessment of the status and threats facing this species is good.

Location: The species occurs in all regions of the OSPAR area, but Region II is peripheral to the range of the species.

Decline: There is good evidence of decline in the past. The decline attributed to "Northern Norway" in the BDC document relates to the entire Northeast Atlantic.

Threats: There is no evidence of current direct threat, but indirect threats such as pollutant effects may be present.

e) ICES overall evaluation

ICES finds that there is good evidence of decline, but rather little evidence of direct threat currently.

References

*Chapman, D.G. 1974. Estimation of population parameters of Antarctic baleen whales. *In* The whale problem: a status report. Ed. by W.E. Schevill. Harvard University Press, Cambridge, MA, USA.

*Evans, P.G.H. 2000. Biology of cetaceans of the North East Atlantic (in relation to seismic energy). *In* Proceedings of the Seismic and Marine Mammals Workshop, 25–28 June 1998. Ed. by M.L. Tasker and C. Weir.

*FAO. 1978. Large whales. Proceedings of the Scientific Consultation on the Conservation and Management of Marine Mammals and their Environment. Mammals in the Seas. Vol. 1. FAO Fisheries Series, 5(1): 51–96.

*Gunnlaugsson, T., and Sigurjónsson, S. 1990. NASS-87: Estimation of whale abundance based on observations made on board Icelandic and Færoese survey vessels. Report of the International Whaling Commission, 40: 571–580.

*IUCN. 2002. 2002 IUCN red list of threatened species. IUCN, Gland, Switzerland.

*Klinowska, M. 1991. Dolphins, porpoises and whales of the world. The IUCN Red Data Book. IUCN, Gland, Switzerland.

*Randall, R.R., Brent, S., Stewart, P.J., Clapham, P.J., and Powell, A.J. 2002. Sea mammals of the world. A & C Black, London.

*Sigurjónsson, S., and Gunnlaugsson, T. 1990. Recent trends in abundance of blue and humpback whales west and southwest of Iceland based on systematic sightings records with a note on occurrence of other cetacean species. Report of the International Whaling Commission, 40: 537–552.

*Sigurjónsson, J., and Víkingsson, G.A. 1997. Seasonal abundance of and estimated food consumption by cetaceans in Icelandic and adjacent waters. Journal of Northwestern Atlantic Fisheries Science, 22: 271–287.

*Yochem, P.K., and Leatherwood, S. 1995. Blue whale *Balaenoptera musculus* (Linnaeus, 1785). *In* Handbook of marine mammals, Vol. 3. Ed. by S.H. Ridgway and R.J. Harrison. Academic Press, London. 362 pp.

4.3 Northern right whale (*Eubalaena glacialis*)

a) Description

The North Atlantic population of this species is usually divided into an eastern and western stock, although photo-identification and preliminary genetics data from recent work suggest that there may be links between animals found in the western and eastern Atlantic (Knowlton et al., *1992; Evans, 2000).*

In the OSPAR Maritime Area, there have been sightings of the northern right whale on or near continental shelf edges off the Iberian Peninsula, the Irish Sea, west of Scotland and Ireland, in Norway, and south of Iceland (Evans, 2000). The whales use northern feeding grounds in the spring, then move to temperate waters in autumn and winter. Historically, the main calving grounds included the Bay of Biscay and there were feeding areas in Scandinavian waters (Collet, 1909; Thompson, 1928; Fairley, 1981).

Regional importance *The historic distribution of the eastern stock of* E. glacialis *included areas both inside and outside the OSPAR Maritime Area. Given the current endangered status of this species, the remaining whales within the OSPAR Area are of regional importance.*

Decline *Tens of thousands of northern right whales were caught in earlier centuries (mostly before 1800), but historic records are not complete enough for pre-whaling population numbers to be estimated accurately. The current size of the North Eastern Atlantic population is unknown, but it is estimated to be no more than the low tens of individuals (Brownell* et al., *1986; Kraus* et al., *in Evans, 2000). The species was believed to be near extinction in the late 1980s, with possibly only a few individuals remaining, and there is no evidence of recovery (Klinowska, 1991).*

Sensitivity *Many populations of* E. glacialis *occurred in coastal waters of temperate regions and appeared to depend on inshore areas for reproductive activities. This species may therefore be more vulnerable to the detrimental effects of human activity than many other cetaceans (Klinowska, 1991).*

Cetaceans use sound to provide information about the physical environment, to communicate between individuals, and for the detection of potential prey. Baleen whales, such as the northern right whale, emit low-frequency sound that can travel hundreds of kilometres. This makes them sensitive to acoustic disturbance from military activities such as naval sonars (particularly low-frequency acoustics), as well as other *sources such as seismic exploration. The whales will be particularly vulnerable if the zone of influence coincides with migration and breeding areas (Evans, 2000).*

<u>Threat</u> The northern right whale has been hunted in the North Atlantic since the 10–11th centuries. The population has been severely depleted as a result and it is now probably the most endangered of the large whale species (Klinowska, 1991). The main current threats are from entanglement in fishing gear, ship strikes, and pollution (bioaccumulation of heavy metals and organochlorines, oil pollution, and radioactivity), and acoustic disturbance.

<u>Relevant additional considerations</u>

<u>Sufficiency of data</u> Most of the historic data on northern right whales come from whaling records. Sightings schemes are a more recent source of information, but it is difficult to determine population size from these data as the animals are so rare.

<u>Changes in relation to natural variability</u> The large numbers of northern right whales that were fished during earlier centuries will have masked any changes in the population caused by natural variability. With such a small number remaining, natural variability may however become a major contributory factor in its local extinction.

<u>Expert judgement</u> Historic records show that tens of thousands of whales were caught when it was the target of whaling during earlier centuries, leading to the historic decline in this species. It is also clear that it remains vulnerable today, and that there is a threat of it becoming extinct in the OSPAR Maritime Area.

<u>Threat and link to human activities</u>

Cross-reference to checklist of human activities in OSPAR MPA Guidelines.

Relevant human activity: Shipping and navigation, military activity, research; fishing, hunting, harvesting; Category of effect of human activity: Physical – noise disturbance. Biological – removal of target species, removal of non-target species, physical damage to species.

Whaling, and therefore human activity, is known to have caused the significant decline of the northern right whale. Current threats from ship collisions, marine pollution, water quality (through bioaccumulation), acoustic disturbance, and entanglement in fishing gear are also linked to human activities.

<u>Management considerations</u> The population was severely depleted before it was given protection by the International Whaling Commission (IWC). The ban needs to remain in place and management measures need to be geared towards enabling the recovery of the population. OSPAR does not deal with whaling issues directly, but can communicate an opinion on it to the IWC and members of the North Atlantic Marine Mammal Commission (NAMMCO).

The IUCN has classified the northern right whale as an endangered species (IUCN, 2002).

b) Literature used (*below)

The literature used is adequate.

c) Literature interpretation

A good interpretation has been made of the literature

d) ICES conclusions

The OSPAR assessment of the status and threats facing this species is good.

Location: The species occurs in all regions of the OSPAR area, but Region II is peripheral to the range of the species.

Decline: There is good evidence of decline in the past.

Threats: The threats are impossible to evaluate in the OSPAR area, but in the Western Atlantic, ship strikes and entanglement in fishing gear continue to threaten possibly the same stock.

e) ICES overall evaluation

ICES finds that there is good evidence of decline, but little evidence of direct threats currently, owing to the extremely low population size.

References

Aguilar, A. 1986. A review of old Basque whaling and its effects on the right whales (*Eubalaena glacialis*) of the North Atlantic. Report of the International Whaling Commission Special Issue, 10: 191–200.

Brown, S.G. 1986. Twentieth-century records of right whales (*Eubalaena glacialis*) in the north-eastern Atlantic Ocean. Report of the International Whaling Commission Special Issue, 10: 121–128.

*Brownell, R.L., Best, P.B., and Prescott, J.H. (eds.). 1986. Right whales: past and present status. Report of the International Whaling Commission, Special Issue, 10: 1–289.

*Collett, R. 1909. A few notes on the whale *Baleana glacialis* and its capture in recent years in the North Atlantic by Norwegian whalers. Proceedings of the Zoological Society of London, 1909: 91–98.

*Evans, P.G.H. 2000. Biology of cetaceans of the North East Atlantic (in relation to seismic energy). *In* Proceedings of the Seismic and Marine Mammals Workshop, 25–28 June 1998. Ed. by M.L. Tasker and C. Weir.

*Fairley, J.S. 1981. Irish whales and whaling. Longstaff Press, Dublin.

*IUCN. 2002. 2002 IUCN Red List of Threatened Species. IUCN, Gland, Switzerland.

*Klinowska, M. 1991. Dolphins, porpoises and whales of the world. The IUCN Red Data Book. IUCN, Gland, Switzerland.

Knowlton, A.R., Kraus, S.D., Meck, D.F., and Mooney-Seus, M.L. 1997. Shipping/right whale workshop. New England Aquatic Forum Series, Report 1997-3. Boston, MA.

Knowlton, A.R., Sigurjónsson, J., Ciano, J.N., and Kraus, S.D. 1992. Long distance movements of North Atlantic right whales. Marine Mammal Science, 8(4): 397–405.

*Kraus, S.D., Hamilton, P.K., Kenney, R.D., Knowlton, A.R., and Slay, C.K. 2001. Status and trends in reproduction of the northern right whale. Journal of Cetacean Research and Management, Special Issue, 2: 231–236.

*Thompson, D'A.W. 1928. On whales landed at the Scottish whaling stations during the years 1908–1914 and 1920–1927. Fisheries Board of Scotland, Scientific Investigations 3. 39 pp.

4.4 Harbour porpoise (*Phocoena phocoena*)

a) Description

The harbour porpoise is generally a coastal species distributed in cold temperate and subarctic waters in the Northern Hemisphere (Klinowska, 1991). In the eastern North Atlantic, it is common and widely distributed on the continental shelf from the Barents Sea and Iceland south to the coasts of France and Spain. There are thought to be a number of subpopulations in the Atlantic and possibly also in the North Sea and adjacent waters, with separate populations occurring in the Irish Sea, northern North Sea, and southern North Sea (Kinze, 1990; IWC, 1996; Walton, 1997; Lockyer, 1999, Andersen et al., *1999; Rosel* et al., *1999).*

<u>Decline</u> *A number of surveys covering different parts of the OSPAR Maritime Area have been carried out to determine the size and trends in the population of the harbour porpoise. Surveys carried out in 1988/1989 estimated harbour porpoise numbers of 10,994 in the Lofoten-Barents Sea area and 82,619 in the northern North Sea, although these may be underestimates (Bjørge and Øien, 1995; IWC, 1990). The only dedicated survey for estimating harbour porpoise abundance in the region was conducted in 1994 and covered the North Sea, English Channel, and Celtic Sea (Hammond* et al., *2002). This resulted in an abundance estimate of between 260,000 and 449,000 (a suggested population of approximately 350,000), of which around 300,000 occurred in the North Sea and the remainder in the Channel and Celtic Sea. Estimates for the Barents Sea and Northern Norwegian waters were 11,000 and for southern Norway and the northern North Sea, 82,600 (Bjørge and Øien, 1995).*

Declines in abundance have been reported since the 1940s, as well as in more recent studies in various parts of the range of P. phocoena. *The harbour porpoise has become scarce in the southernmost North Sea, English Channel, and Bay of Biscay, for example (Evans, 2000), and has declined in the Skagerrak and Kattegat (Berggren and Arrhenius, 1995a, 1995b). It was considered to be one of the most common cetaceans in Region IV of the OSPAR Maritime Area, but sightings and strandings are now only common in certain areas, e.g., the western Galician and northern Portuguese coasts (OSPAR, 2000).*

The harbour porpoise is believed to have been common in waters off the coast of Belgium in the 19th and first half of the 20th century, with data suggesting a decline in the southern North Sea between the 1970s and 1990s. Since 1997, there has been an increase in the number of sightings and strandings in Belgian waters and the Netherlands, but it is not clear whether this reflects an improvement in the status of the population in this area (Haelters et al., *2000; Camphuysen, 1994, Witte* et al., *1998).*

<u>Sensitivity</u> *The harbour porpoise is known to be sensitive to poor water quality, especially toxic contaminants which bioaccumulate and, in the case of organochlorine contamination, this has been linked to reproductive failures (Addison, 1989).*

Like all cetaceans, they use sound for navigation, finding food, and communication and are therefore sensitive to acoustic pollution. Harbour porpoises are amongst the fastest reproducing cetaceans, but depleted populations are nevertheless likely to take decades rather than years to recover.

<u>Threat</u> *Small cetaceans, including the harbour porpoise, were taken for human consumption from the OSPAR Maritime Area until this was made illegal from 1970 (Klinowska, 1991).*

The main threat to this species in the OSPAR Maritime Area today is incidental capture and drowning in fishing nets. For example, the Danish gillnet fishery has been estimated to take more than 4,600 animals a year (IWC, 1996); in the Celtic Sea, by-catch rates have been estimated at more than 6% of the population per year (Tregenza et al., *1997), while in the Swedish Kattegat, Kattegat surveys in 1996 and 1997 calculated by-catch levels of 1.2% and 2.4% of the population in the set-net fishery for cod and pollock. The International Whaling Commission/ASCOBANS working group on harbour porpoise advised a maximum annual by-catch, assuming no uncertainty in any parameter, of 1.7% of the population size per year if the population is to be sustainable (ASCOBANS, 2000).*

Other threats to this species are marine pollution, for example, from toxic substances that bioaccumulate and are known to reduce reproductive fitness, as well as acoustic disturbance (from shipping traffic, oil exploration, military activities, etc.), which may reduce available habitat. A reduction in prey species may also be a threat as the diet of harbour porpoises includes herring, mackerel, and sandeel, which are also targeted by commercial fisheries in the North Sea.

<u>Sufficiency of data</u> *Data on the status and trends of the harbour porpoise have come from sightings programmes and from observers at sea. This includes information on by-catch that has been used to estimate the impact on the population of harbour porpoises in parts of the OSPAR Maritime Area. Tagging studies have also been a source*

of information on the range and behaviour of harbour porpoises. The SCANS survey (Hammond et al., *2002) yielded the first reliable abundance estimate of harbour porpoises in the North Sea and adjacent waters. This estimate is a good basis for estimating the threat imposed by the by-catch rates in the region and in the long run to detect changes in abundance by repeating the survey.*

Changes in relation to natural variability Little is known about the natural variability of harbour porpoise populations or whether such variability has played a role in the decline of this species in particular areas.

Threat and link to human activities

Cross-reference to checklist of human activities in OSPAR MPA Guidelines.

Relevant human activity: Fishing, hunting, harvesting. military activity, research. Category of effect of human activity: Physical – noise disturbance. Biological – removal of target species, removal of non-target species.

The most significant threat to harbour porpoises at the present time is fishing because of the large numbers of animals that are taken as by-catch by a variety of fisheries. This threat is clearly linked to human activity and one which can be addressed through management actions directed at these fisheries.

Management considerations The top priority for management to improve the status of this species must be aimed at reducing the incidental capture of harbour porpoises. This may include technical measures, such as acoustic deterrents, closed areas or closed seasons. More general measures concerned with fisheries management such as effort control may also be required. Other management measures should be targeted at improving coastal water quality by reducing the discharge of substances that are toxic, persistent, and liable to bioaccumulate.

In the North Sea, the harbour porpoise is covered by the terms of the Agreement on the Conservation of Small Cetaceans of the Baltic and North Seas (ASCOBANS), a regional agreement under the Bonn Convention. Many of the useful potential measures fall within the remit of fisheries organisations or ASCOBANS. OSPAR can, however, communicate an opinion on its concern about this species to the relevant bodies and introduce any relevant supporting measures that fall within its own remit if such measures exist or are introduced in the future.

The harbour porpoise is listed on Appendix II of the Bern Convention and Annexes II and IV of the Bonn Convention. IUCN assess the global status of the harbour porpoise as Vulnerable (IUCN, 2002).

b) Literature used (*below)

The literature used is adequate.

c) Literature interpretation

A good interpretation has been made of the literature.

d) ICES conclusions

The OSPAR assessment of the status and threats facing this species is good.

Location: The harbour porpoise occurs in all regions, but the core of the range is Regions II and III. The population structure in the OSPAR area is complex.

Decline: There is good evidence of declines in the past in the Channel and southern North Sea (and more recently in the Baltic). Information on trends in other areas is not clear.

Threats: There is good evidence that the main threat is by-catch, particularly in bottom-set gillnets. The by-catch is likely to be unsustainable on the Celtic shelf, in the Baltic, and probably in parts of the North Sea.

e) ICES overall evaluation

ICES finds that there is good evidence of decline in certain parts of the OSPAR area and good evidence of current threat to harbour porpoises.

References

*Addison, R.F. 1989. Organochlorine and marine mammal reproduction. Canadian Journal of Fisheries and Aquatic Sciences, 46: 360–368.

*Andersen, L.W., Walton, M., Berggren, P., and Lockyer, C. 1999. A comprehensive microsatellite analysis of harbour porpoise *Phocoena phocoena*, sub-populations in eastern North Atlantic including inner Danish waters and the Swedish Baltic. Unpublished Report for Project BYCARE.

*ASCOBANS. 2000. Resolution 3, Incidental take of small cetaceans. Third Session of the Meeting of Parties, Bristol, UK, 26–28 July 2000.

*Berggren, P., and Arrhenius, F. 1995a. Sightings of harbour porpoises (*Phocoena phocoena*) in Swedish waters before 1990. Report of the International Whaling Commission Special Issue, 16: 99–107.

*Berggren, P., and Arrhenius, F. 1995b. Densities and seasonal distribution of harbour porpoises (*Phocoena phocoena*) in the Swedish Skagerrak, Kattegat and Baltic Seas. *In* Biology of the Phocoenids. Ed. by A. Bjørge and G.P. Donovan. International Whaling Commission, Cambridge. 552 pp.

*Bjørge, A., and Øien, N. 1995. Distribution and abundance of harbour porpoise, *Phocoena phocoena*, in Norwegian waters. Report submitted to the International Whaling Commission, SC/42/SM3.

*Clausen, B., and Andersen, S.H. 1988. Evaluation of bycatch and health status of the harbour porpoise (*Phocoena phocoena*) in Danish waters. Danish Review of Game Biology, 13(5): 1–20.

*Camphuysen, C.J. 1994. The harbour porpoise *Phocoena phocoena* in the southern North Sea II: A come-back in Dutch coastal waters? Lutra, 37: 54–61.

*Evans, P.G.H. 2000. Biology of cetaceans of the North East Atlantic (in relation to seismic energy). *In* Proceedings of the Seismic and Marine Mammals Workshop, 25–28 June 1998. Ed. by M.L. Tasker and C. Weir.

*Haelters, J., Jauniaux, T., and Van Compel, J. 2000. Harbour porpoises on Belgian beaches from 1990 to 1999. ASCOBANS Advisory Committee Meeting, Belgium, 13–16 March 2000. Document AC7/Doc.12(P). 5 pp.

*Hammond, P.S., Benke, H., Berggren, P., Borchers, D.L., Collet, A., Heide-Jørgensen, M.P., Heimlich, S., Hiby, A.R., Leopold, M.F., and Øien, N. 2002. Abundance of the harbour porpoise and other cetaceans in the North Sea and adjacent waters. Journal of Applied Ecology, 39: 361–376.

*IUCN. 2002. 2002 IUCN Red List of Threatened Species. IUCN, Gland, Switzerland.

*IWC. 1990. Report of the Subcommittee on Small Cetaceans. IWC/42/4. Report of the International Whaling Commission, 46: 160–179.

*IWC. 1996. Forty-Sixth Report of the International Whaling Commission. Report of the Sub-Committee on Small Cetaceans. Annex H.

*Kinze, C.C. 1990. Non-metric analyses of harbour porpoises (*Phocoena phocoena*) from the North and Baltic Seas: implications for stock identity. IWC document SC/42/SM 35.

*Klinowska, M. 1991. Dolphins, porpoises and whales of the world. The IUCN Red Data Book. IUCN, Gland, Switzerland.

*Lockyer, C. 1999. Application of a new method of investigating population structure of harbour porpoise *Phocoena phocoena*, with special reference to the North and Baltic Seas. Working paper to the 1999 meeting of the ICES Working Group on Marine Mammal Habitats.

*OSPAR. 2000. Quality Status Report Region IV. Bay of Biscay and Iberian Coast. OSPAR Commission, London.

*Tregenza, N.J.C., Berrow, S.D., Hammond, P.S., and Leaper, R. 1997. Harbour porpoise (*Phocoena phocoena* L.) by-catch in set gillnets in the Celtic Sea. ICES Journal of Marine Science, 54: 896–904.

*Walton, M.J. 1997. Population structure of harbour porpoises *Phocoena phocoena* in the seas around the UK and adjacent waters. Proceedings of the Royal Society, London B, 264: 89–94.

*Witte, R.H., Baptist, H.J.M., and Bot, P.V.M. 1998. Increase of the harbour porpoise *Phocoena phocoena* in the Dutch sector of the North Sea. Lutra, 40: 33–40.

ACRONYMS

ACE	Advisory Committee on Ecosystems
ACFM	Advisory Committee on Fishery Management
ACME	Advisory Committee on the Marine Environment
AGDS	acoustic ground-discrimination systems
ASC	Annual Science Conference
ASCOBANS	Agreement on Small Cetaceans of the Baltic and North Seas
BDC	OSPAR Biodiversity Committee
BEWG	Benthos Ecology Working Group
BOOS	Baltic Ocean Observing System
BSRP	Baltic Sea Regional Project
COMBINE	Cooperative Monitoring in the Baltic Marine Environment (HELCOM)
DDT	dichlorodiphenyltrichloroethane
DEFRA	Department for Environment, Food and Rural Affairs (UK)
DG	Directorate General
EC	European Commission
EcoQ	Ecological Quality
EcoQO	Ecological Quality Objective
EEA	European Environment Agency
ELIFONTS	Effects of Large-Scale Industrial Fisheries on Non-target Species
EU	European Union
EUNIS	European Nature Information System
GEF	Global Environment Facility
GIS	Geographical Information System
GIWA	Global International Waters Assessment
GLOBEC	Global Ocean Ecosystem Dynamics
GOOS	Global Ocean Observing System
HELCOM	Helsinki Commission (Baltic Marine Environment Protection Commission)
IBSFC	International Baltic Sea Fisheries Commission
IBTS	International Bottom Trawl Survey
ICES	International Council for the Exploration of the Sea
IMPRESS	Interactions between the marine environment, predators and prey: implications for sustainable sandeel fisheries
IOC	Intergovernmental Oceanographic Commission
IWC	International Whaling Commission
JAMP	OSPAR Joint Assessment and Monitoring Programme
JNCC	Joint Nature Conservation Committee (UK)
MarLIN	Marine Life Information Network programme (UK)
MCAP	Management Committee on the Advisory Process
MONAS	HELCOM Monitoring and Assessment Group
MSVPA	Multispecies Virtual Population Analysis
MPAs	Marine Protected Areas
NAO	North Atlantic Oscillation
NASCO	North Atlantic Salmon Conservation Organization
NGOs	non-governmental organizations
NMMP	National Marine Monitoring Programme (UK)
NORSEPP	Planning Group on the North Sea Pilot Project
NSBP	North Sea Benthos Project
NSBS	North Sea Benthos Survey
OSPAR	OSPAR Commission
PA	Precautionary Approach
PCBs	polychlorinated biphenyls
PODs	porpoise detectors
PDV	Phocine Distemper Virus
PGIBSRP	Planning Group on the Implementation of the Baltic Sea Regional Project
PP	Precautionary Principle
QSR	Quality Status Report
REGs	Regional Ecosystem Groups
REGNS	Regional Ecosystem Study Group for the North Sea

RMC	Resource Management Committee
SACs	Special Areas of Conservation
SCOR	Scientific Committee on Oceanic Research
SGAWWP	Study Group on ACFM, ACE, and ACME, and Working Group Protocols
SGCOR	Study Group on Mapping the Occurrence of Cold-Water Corals
SGEF	Study Group on Elasmobranch Fishes
SGFEN	Subgroup on Fishery and Environment
SGGROMAT	Study Group on Growth, Maturity and Condition in Stock Projections
SGPBI	Study Group on Modelling of Physical-Biological Interactions
SGPRISM	Study Group on Incorporation of Process Information into Stock Recruitment Models
SGPRP	Study Group on Precautionary Reference Points for Advice on Fishery Management
SSB	spawning stock biomass
STECF	Scientific, Technical and Economic Committee for Fisheries
TAC	Total Allowable Catch
TEGs	Thematic Ecosystem Groups
UK	United Kingdom
U.S.	United States
USA	United States of America
VHVO	very high vertical opening
VPA	Virtual Population Analysis
WFD	Water Framework Directive
WGAGFM	Working Group on the Application of Genetics in Fisheries and Mariculture
WGDEEP	Working Group on the Biology and Assessment of Deep-Sea Fisheries Resources
WGECO	Working Group on Ecosystem Effects of Fishing Activities
WGFE	Working Group on Fish Ecology
WGMHM	Working Group on Marine Habitat Mapping
WGMME	Working Group on Marine Mammal Ecology
WGSE	Working Group on Seabird Ecology
WMO	World Meteorological Organization
WWF	Worldwide Fund for Nature