

REPORT OF THE
Working Group on Fish Ecology

ICES Headquarters
3–7 March 2003

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TABLE OF CONTENTS

Section	Page
1 INTRODUCTION.....	1
1.1 Terms of Reference.....	1
1.2 Participants.....	2
1.3 Background to report.....	2
2 REFERENCE LEVELS FOR ECOQOS RELATED TO CHANGES IN THE PROPORTION OF LARGE FISH, AVERAGE WEIGHT AND AVERAGE MAXIMUM LENGTH OF THE FISH COMMUNITY.....	3
2.1 Introduction.....	3
2.2 Analysis of trends in proportion of large fish, average weight and average maximum length of the fish community these features over time.....	3
2.2.1 Introduction.....	3
2.2.2 North-western North Sea.....	4
2.2.2.1 Analytical Design.....	4
2.2.2.2 The data-sets.....	6
2.2.2.3 Community Size-composition Metrics.....	6
2.2.2.4 Results.....	6
2.2.2.5 Discussion.....	7
2.2.3 North Sea.....	12
2.2.4 North-Norway and West-Spitsbergen.....	13
2.2.5 Celtic Sea.....	15
2.2.5.1 Methods.....	15
2.2.5.2 Results.....	18
2.2.6 Portuguese continental waters.....	22
2.2.7 Summary.....	27
2.3 Evaluation of data and trends in other size and maximum-length characteristics.....	28
2.3.1 North Sea.....	29
2.3.1.1 Methods.....	29
2.3.1.2 Results.....	30
2.3.1.3 Discussion.....	38
2.3.2 Celtic Sea.....	39
2.3.2.1 Methods.....	39
2.3.2.2 Results and Discussion.....	39
2.4 Identification of the levels of the metrics that can be taken as indicative of various states of the marine ecosystem.....	45
2.4.1 A case study for the North Western North Sea.....	45
2.4.2 Discussion.....	49
3 EVALUATION OF THE QUALITY AND SUITABILITY OF DATA FOR THE LISTING OF THREATENED AND DECLINING FISH SPECIES BY OSPAR.....	52
3.1 Introduction.....	52
3.2 Sea Lamprey (<i>Petromyzon marinus</i>).....	55
3.3 Common Sturgeon (<i>Acipenser sturio</i>).....	60
3.4 Allis Shad (<i>Alosa alosa</i>).....	65
3.5 Houting (<i>Coregonus lavaretus oxyrinchus</i>).....	72
3.6 Sea horses (<i>Hippocampus</i> spp.).....	74
3.6.1 Status and distribution.....	74
3.7 Couch's goby (<i>Gobius couchi</i>).....	77
3.8 Sea turtles.....	77
3.9 Conclusions.....	81
4 REVIEW OF ESSENTIAL FISH HABITAT.....	84
4.1 Introduction.....	84
4.2 Review of the concept of Essential Fish Habitat.....	86
4.2.1 Breeding, spawning and parturition grounds.....	86
4.2.2 Nursery grounds.....	86
4.2.3 Shelter, and natural and artificial refuges.....	87
4.2.4 Feeding grounds.....	88
4.2.5 Migratory corridors.....	88
4.3 Data requirements for identifying Essential Fish Habitat.....	88
4.4 Conclusions.....	89
5 GASTRIC EVACUATION, TOTAL STOMACH CONTENT, PREY COMPOSITION AND FOOD RATION.....	92
5.1 Introduction.....	92

Section	Page
5.2 Correction Factor k	92
5.3 Food rations of saithe and whiting estimated from quarterly mean stomach data	93
5.4 Prey composition of individual stomachs	94
5.5 Estimation of food rations of fish feeding on invertebrates with a robust exoskeleton.....	94
5.6 Conclusions.....	94
6 REVIEW TOPICS OF GEOGRAPHIC AREAS WHERE ICES MAY CONTRIBUTE TO WORK ON RARE MARINE FISH SPECIES	105
7 RECOMMENDATIONS	107
ANNEX	108

1 INTRODUCTION

1.1 Terms of Reference

The Terms of Reference for the Working Group on Fish Ecology (WGFE, Chair: J. Ellis, CEFAS, UK) are that it will meet in Copenhagen, Denmark, from 3-7 March 2003 to:

(a) In preparation for development of objectives and reference levels for EcoQOs related to changes in the proportion of large fish, average weight and average maximum length of the fish community

i) analyse trends in these features over time for appropriate survey data sets from the North Sea and other seas

ii) evaluate data and trends in other characteristics of fish communities which reflect the same information as in i)

iii) where possible identify levels of the properties in i) and ii) that can be taken as indicative of various states of the marine ecosystem (i.e., serve as potential reference levels).

(b) Provide an assessment of the data on which the justification of the fish on the OSPAR priority list of threatened and endangered species and habitats will be based. This assessment should 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 fish concerned can be identified, consistently with the Texel-Faial criteria. These fish species are: sturgeon (*Acipenser sturio*), allis shad (*Alosa alosa*), houting (*Coregonus lavaretus oxyrhynchus*), Couch's goby (*Gobius couchi*), short-snouted seahorse (*Hippocampus hippocampus*), seahorse (*Hippocampus ramulosus*) and sea lamprey (*Petromyzon marinus*). In addition, a last-minute request was made to assess both the loggerhead and leatherback turtles.

(c) Review the concept of essential fish habitat and consider what information would be required in order to be able to specify essential fish habitat for individual species or stocks.

(d) Initiate for completion in 2003 analyses of:

i) ICES data on individual stomach contents as well as other relevant stomach data to obtain estimates of the correction factor k

ii) Estimates of food rations of whiting and saithe (feeding primarily on fish) using a prey energy density dependent evacuation model to quarterly mean stomach data

iii) the ICES stomach data to acquire possible information about prey composition of individual stomachs. The focus will be on major prey types that deviate significantly in energy density.

(e) Initiate for completion by 2004, estimates of food rations of cod and haddock (preying to a large extent on invertebrates with a robust exoskeleton also) using a gastric evacuation model which accounts for the effects of a robust exoskeleton on evacuation of total stomach content as well as of individual prey types. Such a model is expected to be in operation around mid-2003

(f) Review topics of geographic areas where ICES might further contribute to work on rare marine fish species.

1.2 Participants

The following scientists attended the first Working Group meeting. Full contact details are given in the Annex.

Ole Thomas Albert, Norwegian Institute of Fisheries and Aquaculture, Norway
Niels Gerner Andersen, Danish Institute for Fisheries Research, Denmark
Miran Arahamian, Environment Agency, UK
Julia Blanchard, CEFAS, UK
Niels Daan, Netherlands Institute for Fishery Research, The Netherlands
Nick Dulvy, CEFAS, UK
Jim Ellis, CEFAS UK
Helen Fraser, FRS, Scotland
Simon Greenstreet, FRS, Scotland
Louize Hill, IPIMAR, Portugal
Charlotte Mogensen, Joint Nature Conservation Committee, UK
Jens Pedersen, EUCNORD, Denmark
Eric Rochard, Cemagref, France

1.3 Background to report

The Working Group on Fish Ecology was initiated in 2002, with this the first meeting. The rationale behind the formation of the group was to support ICES on issues of fish community metrics and for providing advice on threatened marine fishes. OSPAR and HELCOM have requested advice in these areas in the recent past that ICES has been unable to respond to.

At present, fish community issues are considered by WGECO. The demands on WGECO are heavy and by establishing WGFE will enable a more focussed consideration of community issues. ICES has no scientific capacity to provide information on rarer, non-commercial fish species, and there are increasing requests for advice in this area in relation to biodiversity and nature conservation issues. The results of the work of this group might be of further interest to the fisheries assessment groups.

2 REFERENCE LEVELS FOR ECOQOS RELATED TO CHANGES IN THE PROPORTION OF LARGE FISH, AVERAGE WEIGHT AND AVERAGE MAXIMUM LENGTH OF THE FISH COMMUNITY

2.1 Introduction

ToR a: In preparation for the development of objectives and reference levels for EcoQOs related to changes in the proportion of large fish, average weight and average maximum length of the fish community:

- i. *analyse trends in these features over time for appropriate survey data sets from the North Sea and other seas,*
- ii. *evaluate data and trends in other characteristics of fish communities which reflect the same information as in i),*
- iii. *where possible identify levels of the properties in i) and ii) that can be taken as indicative of various states of the marine ecosystem (i.e., serve as potential reference levels);*

This term of reference follows on from the Bergen declaration by the North Sea Ministers in March 2002, where a framework was established to implement Ecological Quality Objectives (EcoQOs) for the North Sea. EcoQOs should include both the desired levels of ecological quality and a baseline (reference point) against which progress can be measured and must not permit any worsening of existing conditions (ICES 2002a). The ecological quality element that was agreed on for fish communities was: “*changes in the proportion of large fish and hence the average weight and average maximum length of the fish community*”.

The ICES Advisory Committee on the Environment (ACE) reviews this ecological quality element in its 2002 report (ICES 2002a). Although the EcoQ element consists of two complementary metrics that represent different aspects (average weight as a representation of size structure and average maximum length as a representation of species composition (Piet 2001)), neither of the two criteria deemed desirable (high response to signal from human activity compared to variation induced by other factors and tight time linkage to that activity) were met by these metrics. Moreover, two factors were identified that need to be addressed before further progress can be made: (a) the lack of long-term effort data with a high spatial resolution and (b) problems in evaluating management measures. The final conclusion was that “it is impossible to determine a reference level because monitoring commenced long after exploitation started but that target levels might be derived if time series are long enough”.

2.2 Analysis of trends in proportion of large fish, average weight and average maximum length of the fish community these features over time.

2.2.1 Introduction

At the meeting data sets for analyzing temporal trends in various metrics were available for five main areas. The longest one covering approximately 80 years exists for the Scottish August Groundfish Survey. Three other shorter data sets (15-30 years) for the North Sea were also analysed. The other data sets referred to the North-Norway Coastal Survey (8 years) and West-Spitsbergen Survey (10 years), the Celtic Sea Survey (16 years) and the Portuguese Autumn Demersal Survey (13 years). As far as possible the same analyses were carried out for all data sets, but availability of data was sometimes restrictive. Thus, data on maximum length were lacking for Norwegian surveys and proportions of large fish above the 60, 85 and 95th percentiles of the size compositions were absent for the North Sea.

In this section we have applied standard statistical approaches to many of our analyses. This has been done in an effort to apply a certain level of rigour to our interpretation of the results. However, the nature of the data we are analysing violates many of the assumptions underlying the major common statistical techniques. For example, we examine temporal trends in the data and in some instances apply standard linear regression analyses to the data. Such techniques assume that the data are independent, so that the assessment of the number of degrees of freedom is a standard function of the sample size (i.e. N-2). However time series data may well not be independent of one another at the temporal scale that concerns us. The value at any one point in time may well be similar to the value in the preceding time point, or points, and influence the subsequent values. This makes it difficult to assess the correct number of degrees of freedom. The groundfish survey data that we are analysing are also collected in space, leading to potential spatial auto-correlation between closely neighbouring data points. The effect of not taking such auto-correlation into account is to over-estimate the true number of degrees of freedom, thus increasing the risk of Type I errors: of rejecting the null hypothesis when this is not appropriate. On the other hand the sampling design and limited time scale of certain data sets could lead to lower statistical power increasing the likelihood of Type II errors, which is to not detect a trend that is present. To

counter these problems we have as far as possible provided actual probability values. Results of analyses where the probability is close to the 5% significance level should be treated with caution.

2.2.2 North-western North Sea

Scottish August Groundfish Survey (SAGFS) data and international effort data were analysed to examine how fishing affects three fish community size-composition metrics that could provide the basis for setting potential EcoQOs. Spatial and temporal analyses were combined in an attempt to strengthen the case that the changes observed were in fact caused by fishing.

2.2.2.1 Analytical Design

Data for 75 ICES rectangles, divided into three groups, or treatments, of low, medium and high “current” fishing disturbance were analysed. Three potential indicators of anthropogenic effects on the size structure of the fish assemblage were examined; the proportion of large fish in the assemblage, the average weight of fish in the assemblage, and the average Length_{infinity} of fish in the assemblage. The hypothesis that each metric was “most affected” in the rectangles of highest fishing disturbance, and least affected in the rectangles of lowest disturbance was tested. Even should these first hypotheses be supported by the data, this does not necessarily confirm that fishing has caused the changes. The alternative interpretation, however unlikely, that current fishing levels are highest in rectangles where the ultimate body size, average weight and proportion of large fish in the assemblage are least cannot at this point be discounted. To rule out this alternative interpretation, long-term time-series trends for each of the groups of rectangles were examined. If fishing is responsible for the change in the community characteristics, then predictable temporal trends should be apparent. Little or no long-term trend should be apparent in rectangles where fishing disturbance is low, whereas in rectangles affected by fishing, temporal trends in a predictable direction should be detected. The greater the impact from fishing, the steeper the gradient should be. An assumption underlying this analytical design is that prior to any apparent fishing effects, the community characteristics in the different treatment areas had the same start point, and that each was subjected to approximately the same low level of fishing disturbance. This is examined in a later section dealing with reference points and the setting of target values.

In adopting this analytical design, we have attempted to follow, as far as was possible, a one-way ANOVA design. However, it is important to realise that the distribution of fishing effort was not random across the 75 rectangles (Jennings *et al.* 1999b; Greenstreet *et al.* 1999b, see also Figure 2.2.2.1.1.). A true ANOVA design would have had each “treatment” set of rectangles distributed randomly across the 75 rectangles. This has two major implications.

- Spatial variation could introduce a potentially confounding effect. Concentration of the impact of fishing into restricted areas could magnify the effect of fishing on the demersal fish community. Nevertheless this is still a fishing effect. It will lead to similar distribution in the community characteristic being investigated. The question is, can this spatial factor introduce the sort of trends we anticipate independently of fishing?
- The spatial cohesion of both the “treatment” and the “effect” could, through spatial auto-correlation, reduce the independence of the data. This has consequences with respect to estimation of the actual degrees of freedom in any statistical analysis. While we have presented significance levels for the ANOVA results, some caution is necessary in interpreting these. Analysis of groundfish survey data collected at high spatial resolution (25 to 30 half-hour GOV samples collected within a 20km by 20km area) suggests that auto-correlation between species abundance is almost entirely diminished at distances of around 10 to 15km. Examination of variograms for each of the community characteristics indicates that spatial autocorrelation is diminished over a distance of around 150km (two to three ICES rectangles), suggesting that the true number of total degrees of freedom may be as few as 25 to 40.

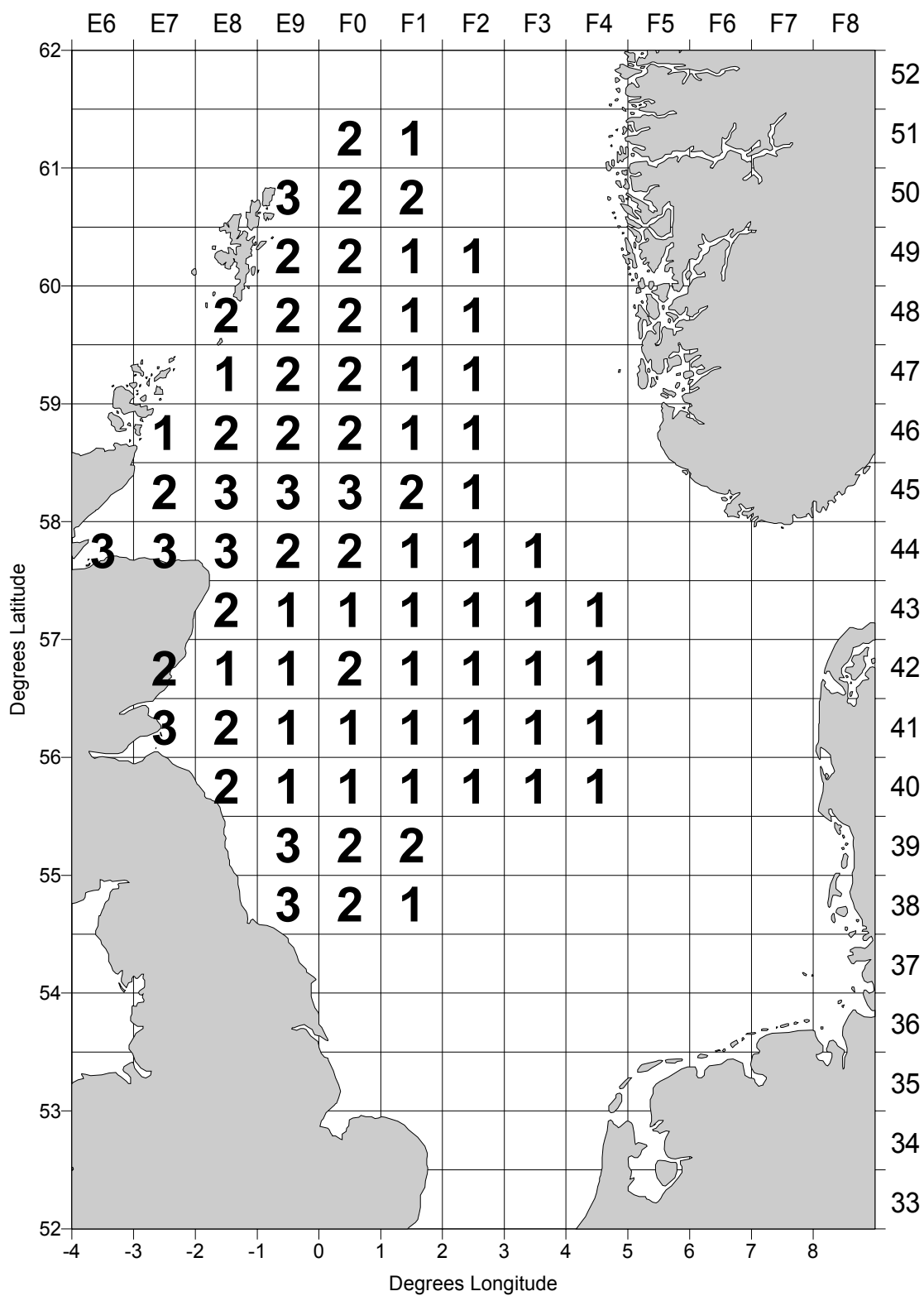


Figure 2.2.2.1.1. Chart showing the 75 ICES statistical rectangles covered by the SAGFS. Figures indicate the otter trawling effort group to which each rectangle was assigned: 1 is Low effort, 2 is Medium effort and 3 is High effort.

2.2.2.2 The data-sets

Groundfish survey data

Scottish August Groundfish Survey (SAGFS) data were examined from 75 ICES statistical rectangles located in the north-western North Sea where survey coverage was most complete (Figure 2.2.2.1.1.). Only samples collected using a 48foot Aberdeen Otter Trawl towed for one hour were included in the data set. Only data for those groundfish species likely to be well sampled by the gear were analysed. Pelagic species, such as herring, sprats and sandeels, were all excluded. The results therefore only apply to the demersal groundfish community occupying the area. For more details regarding the data see Greenstreet and Hall (1996) and Greenstreet *et al.* (1999a).

To determine “contemporary” levels of the three community size-composition metrics, species at length abundance data covering a period of 14 years from 1983 to 1996 were extracted. All samples were collected by the same survey vessel, FRV *Scotia (II)*. For one rectangle only ten trawl samples were available. This rectangle was not sampled in 1983, 1985, 1987 or 1995. To avoid sample size dependency problems, sampling effort was standardised to ten trawls in the other 74 rectangles by excluding, as necessary, trawl samples selected at random from these four years. Previous analysis of SAGFS data has indicated that it is necessary to aggregate at least five one-hour trawl samples in order to derive reliable community metrics. All ten trawl samples in each rectangle were therefore combined to provide a single aggregated, highly standardised, species abundance sample for each rectangle upon which to calculate each community size-composition metric.

For the second analysis, looking at “long-term temporal trends” in the structure and composition of the groundfish assemblage in rectangles varying in the level of fishing effort to which they had been subjected, data from the full time-series, spanning the period 1925 to 1996, were used. Following Greenstreet *et al.* (1999a), data were pooled into “groups” of two or three years to ensure adequate sampling effort in each time-period/effort “treatment” cell.

Fishing effort data

International fishing effort (hours fished) for the period 1990 to 1995 were used to define rectangles subject to three different levels of fishing disturbance (Jennings *et al.* 1999b; Jennings *et al.* 2000). Total annual average fishing effort across the 75 statistical rectangles amounted to 963,216 hours of fishing, 67% of which consisted of otter trawling, 12% beam trawling and 21% seine netting. Otter trawl is the predominant gear used in this part of the North Sea (Greenstreet *et al.* 1999b). The treatment effort levels set were therefore based solely on this gear. Average annual effort values were calculated to provide estimates of the “current” spatial distribution of fishing effort across the 75 ICES statistical rectangles for which groundfish survey data were available. These ranged from 645 h.yr⁻¹ to 63,794 h.yr⁻¹. Three broad categories (“treatments”) were defined; 40 rectangles of relatively low otter trawling intensity, from 0 to 4999 h.y⁻¹, 25 rectangles of medium otter trawl effort, from 5000 to 19,999 h.y⁻¹, and 10 rectangles of high otter trawling intensity, exceeding 20,000 h.y⁻¹. The distribution of rectangles belonging to each of these treatments is indicated in Figure 2.2.2.1.1.

2.2.2.3 Community Size-composition Metrics

Species abundance at length data in the recent effort standardised database were examined to determine an appropriate length at which a fish might be considered to be large. 95% of all the individual fish recorded were 30cm in length or less. The top five percentile of the length distribution, consisting of fish >30cm, was therefore defined as “large”. Length-weight relationships were used to convert abundance at length to weight at length. Average weight in a treatment was determined simply by dividing the total weight in the sample by the total number of individuals. Length_{Infinity} data, determined from the von Bertalanffy growth equation calculated for each species, were available for 28 of the 56 species included in the SAGFS database (Jennings *et al.* 1998; Jennings *et al.* 1999a). These 28 species accounted for over 98% of the individuals sampled by the SAGFS in any of the spatial/temporal “treatments”. Species abundance data were converted to the number of individuals with particular Length_{Infinity} values, and the mean value for each spatial/temporal “treatment” was computed.

2.2.2.4 Results

The following hypotheses regarding the three groundfish assemblage size-composition metrics were examined:

- In contemporary times the proportion of large fish, average weight and average Length_{Infinity}, of fish in the groundfish assemblage should be least in rectangles where current levels of otter trawling are highest, and highest in rectangles where current levels of otter trawling are lowest.

- Temporal trends in the proportion of large fish, average weight and average Length_{Infinity}, of fish in the groundfish assemblage should be more steeply negative in rectangles where current levels of otter trawling are highest. No, or little, temporal trend should be apparent in rectangles where otter trawl effort levels are low.

The first analysis used the contemporary effort-standardised extraction from the SAGFS database. Mean (± 1 S.D. of the mean) contemporary size-composition metrics values were determined for the three group of rectangles varying in their current exploitation levels (Figure 2.2.2.4.1.). Differences were tested using one-way ANOVA. Within the contemporary demersal fish assemblage in the north-western North Sea, the percentage of fish greater than 30cm in length and the average weight of fish were least in areas of high otter trawling disturbance and highest in areas with low fishing effort. Both ANOVAs were significant, but only at the 5% significance level with respect to average weight. Bearing in mind the problems involved in assessing the true number of degrees of freedom discussed above, this latter result should perhaps be treated with some caution. In post-hoc comparisons between the different treatments, Tukey's tests suggested that only in comparisons between the high effort and the low effort treatment rectangles were the differences significant (Table 2.2.2.4.1.). Once again this difference was only significant at the 5% level for the average weight comparison. One way ANOVA for Length_{Infinity} was highly significant. However, little difference between the high effort and medium effort treatments was apparent (confirmed by the high Tukey's test probability obtained). Indeed, Tukey's test revealed that the post-hoc comparison between High and Low effort rectangles was also not significant – no doubt explained by the small sample size of High effort rectangles imparting insufficient power. The significant ANOVA was entirely due to the highly significant comparison of the average Length_{Infinity} in fish assemblages in Medium and Low effort rectangles (Table 2.2.2.4.1.).

Table 2.2.2.4.1. Results of post-hoc Tukey analyses. Probability values are shown for each effort-treatment comparison.

Effort Treatments	% Fish >30cm	Average Weight	Length _{Infinity}
High v Low	0.005	0.038	0.130
High v Medium	0.294	0.319	0.969
Medium v Low	0.063	0.381	0.008

Long-term trends in the three demersal fish assemblage size-composition metrics were determined for each effort treatment group of rectangles using the full time series of available SAGFS data, from 1925 to 1996. LOWESS smooths were fitted to the data so as to highlight non-linear variation over the time span of the data set, but in each case, a linear regression analysis was also carried out to assess and quantify the overall trend (Figures 2.2.2.4.2. to 2.2.2.4.4.).

In rectangles of low otter trawl effort, no overall long-term trend was detected in the percentage of large fish in the demersal assemblage. Overall negative trends were detected in both medium and high effort treatments, although in the case of the former, the trend was not statistically significant. The LOWESS smooth suggests that in rectangles of medium otter trawl effort, the apparent negative trend was due to a decline in the percentage of large fish over the last two decades of the time series. The LOWESS smooth suggested that a linear fit was a close approximation to the actual trend in the data observed in rectangles of high fishing effort. For both average fish weight and Length_{Infinity}, the overall negative trends were statistically significant in rectangles of both medium and high otter trawl effort. In both cases the LOWESS smooth indicated that a linear fit was reasonable. This was true for the high effort treatments over the whole duration of the time series, and over the last 40 to 50 years of the time series for the medium effort treatment. Overall negative trends were also observed for both size composition metrics in rectangles of low otter trawl effort. In neither case, however, was this trend significant. In both cases the LOWESS smooth suggested that much of the cause of the overall negative trend was due to low metric values in the last 20 years of the time series.

2.2.2.5 Discussion

These analyses support the contention that fishing affects the size composition of the demersal fish assemblage in the north-western North Sea. The percentage of large fish, average weight of fish and average Length_{Infinity} of fish in the community were all lower in areas of high fishing effort and highest in areas least affected by fishing. The time-series analyses confirm the cause and effect relationship. In areas of low fishing effort no significant overall trend was observed for any of the three size composition metrics. Conversely, in areas affected by otter trawling, the predicted negative trends were observed, and the gradients of these trends tended to be steeper the greater the fishing impact. All three of these metrics appear sensitive to the effects of fishing on the size composition of the demersal fish assemblage, and could therefore provide the basis for potential operational EcoQOs with respect to this element of ecological quality.

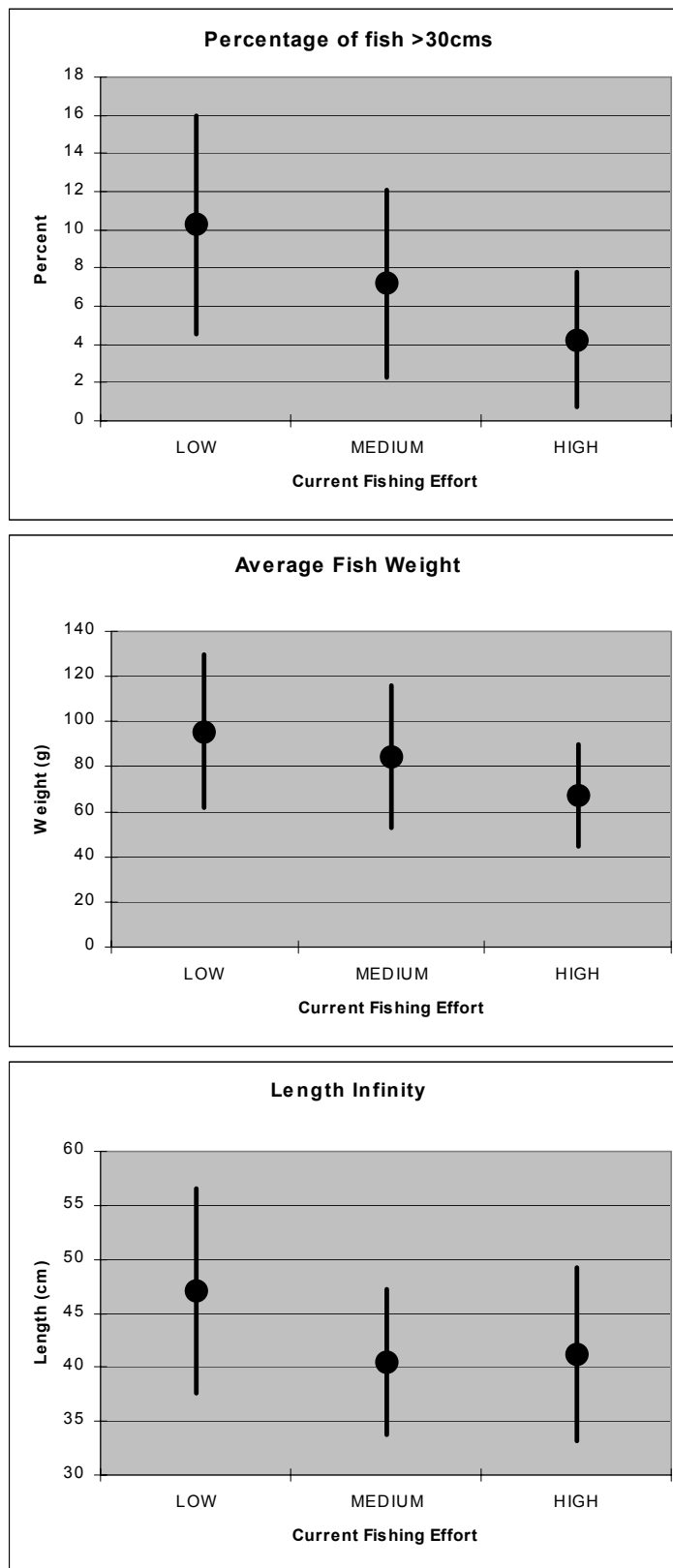


Figure 2.2.2.4.1. Mean (± 1 Standard Deviation) of the proportion of fish larger than 30cm (ANOVA $P=0.005$), the average weight (ANOVA $P=0.041$), and the average Length_{∞} (ANOVA $P=0.007$) of fish in the demersal fish assemblage in rectangles differing in their current levels of fishing effort.

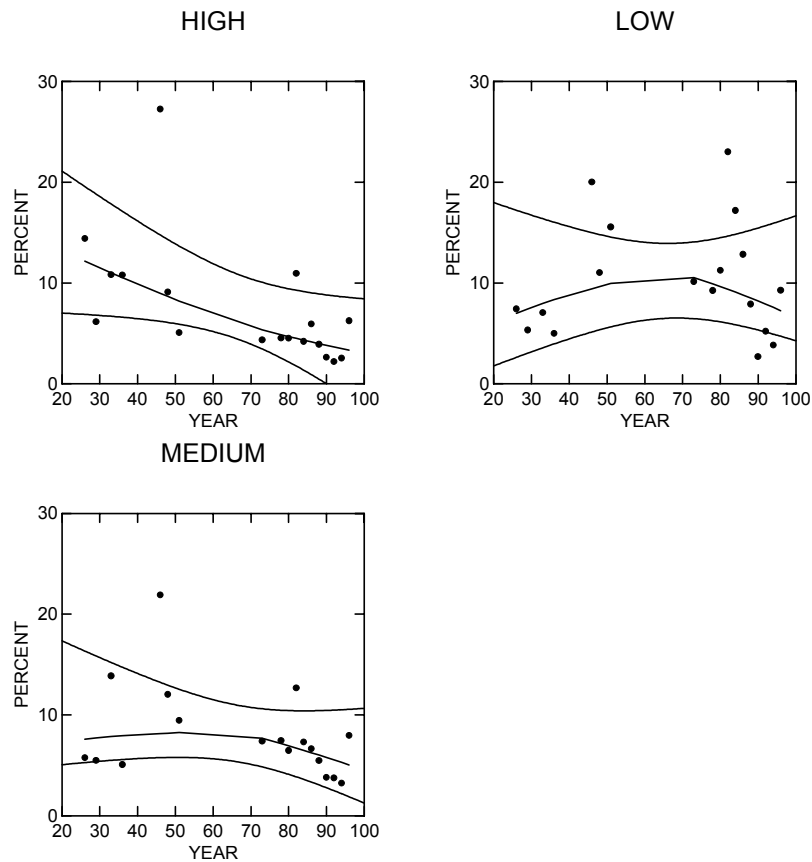


Figure 2.2.2.4.2. Long-term trends in the proportion of fish >30cm in length in rectangles of high, medium and low otter trawl effort. Trend lines are LOWESS smooths with 95% confidence limits. Linear regressions revealed overall negative trends in areas of high ($b = -0.138$, $P = 0.013$) and medium ($b = -0.066$, $P = 0.145$ not significant) fishing effort. The LOWESS smooth shown here suggests that most of the negative slope in rectangles of medium otter trawl effort was due to a sharp decline in the percentage of large fish in these areas from the early 1980s onwards. Linear regression revealed no overall trend in rectangles of low otter trawl effort ($b = 0.007$, $P = 0.897$ not significant).

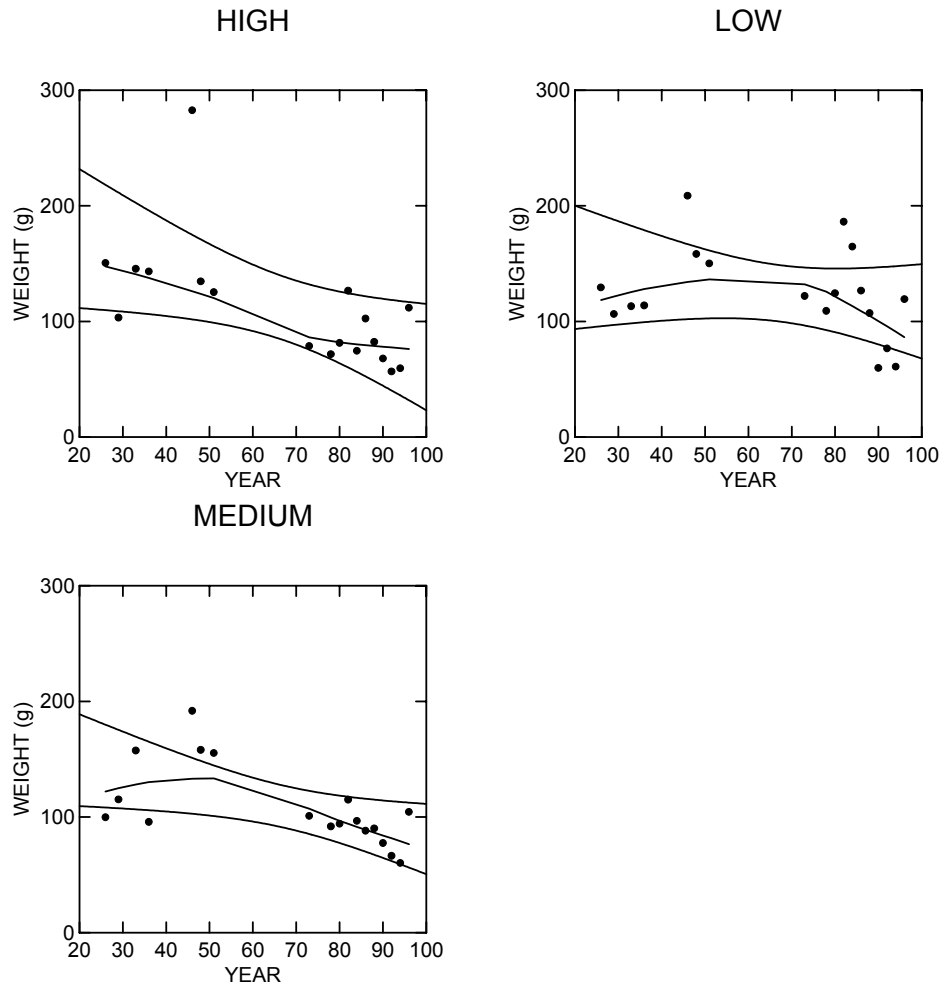


Figure 2.2.2.4.3. Long-term trends in the average weight of fish in rectangles of high, medium and low otter trawl effort. Trend lines are LOWESS smooths with 95% confidence limits. Linear regressions revealed overall negative trends in all three effort treatments (High: $b = -1.281$, $P = 0.008$; Medium $b = -0.852$, $P = 0.007$; Low $b = 0.475$, $P = 0.219$ not significant), but these trends were only significant with respect to High and Medium fishing effort rectangles. The LOWESS smooths shown here suggest that the linear fit is reasonably close in areas of High otter trawling effort. However, most of the negative slope in rectangles of Medium and Low otter trawl effort were due to sharp declines in the average weight of fish in these areas during the latter stages of the time series.

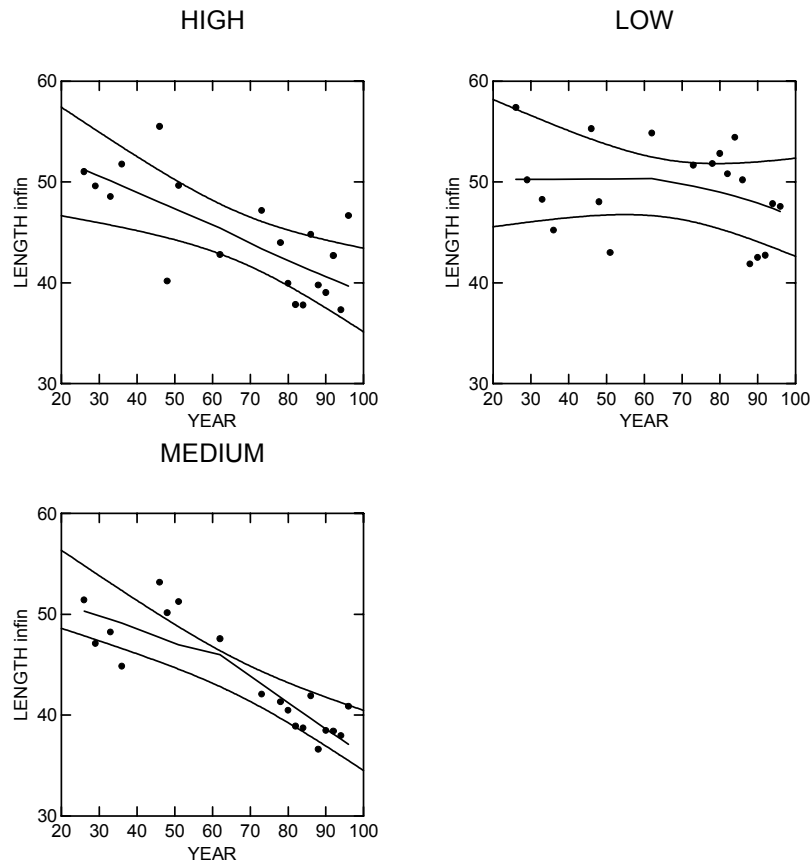


Figure 2.2.2.4.4. Long-term trends in the average $\text{Length}_{\text{infinity}}$ determined for the demersal fish assemblage in rectangles of high, medium and low otter trawl effort. Trend lines are LOWESS smooths with 95% confidence limits. Linear regressions revealed overall negative trends in areas of high ($b = -0.159$, $P=0.001$), medium ($b = -0.187$, $P<0.001$) and low otter trawl effort ($b = -0.055$, $P = 0.238$) otter trawl fishing effort, but in the latter case, this relationship was not significant. The LOWESS smooths shown here suggest that most of the negative slope in rectangles of low otter trawl effort was due to a sharp decline in assemblage $\text{Length}_{\text{infinity}}$ from the mid 1970s onwards. For rectangles of medium and high otter trawl effort, the LOWESS smooths suggest that linear fits were fair approximations for the trends in the data.

2.2.3 North Sea

Piet (2002) evaluated trends in average weight and average maximum length in three monitoring surveys (IBTS, BTS and SNS) with particular reference to their performance as indicators of changes in fishing effort. Figure 2.2.3.1 shows the significant long-term and downward trends that have been observed consistently for the two metrics in all surveys (for further details see working document by G.J.Piet).

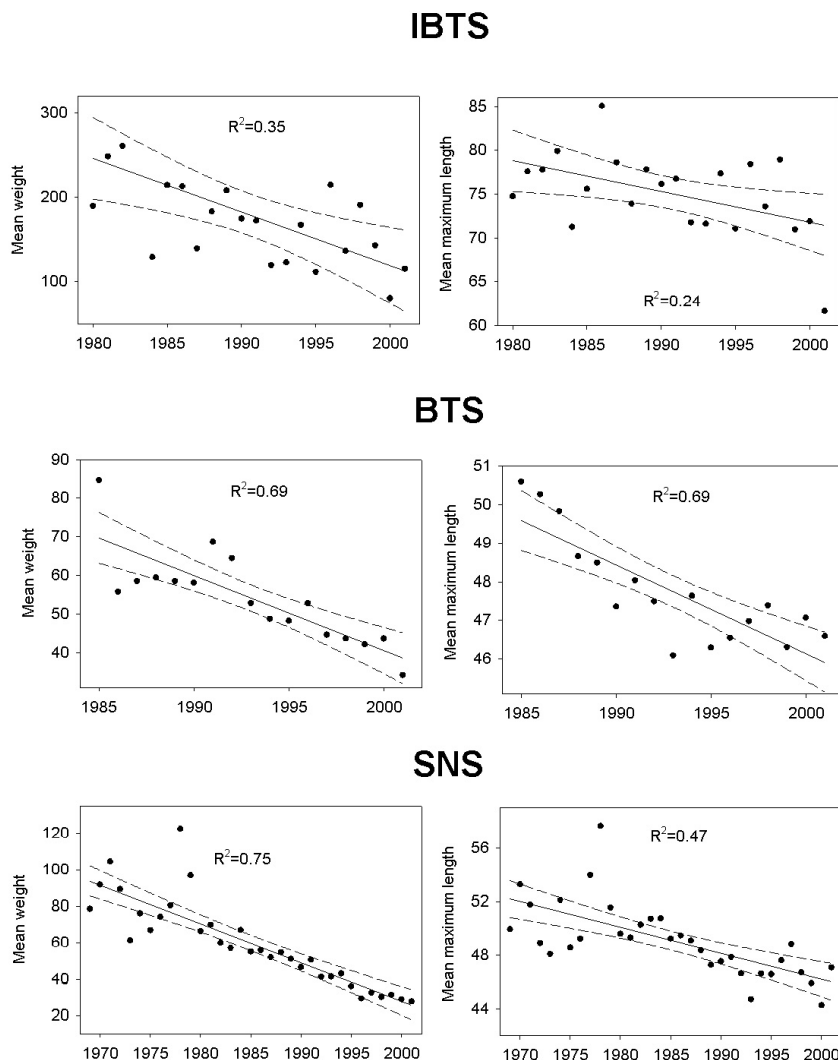


Figure 2.2.3.1. Long-term trends (with confidence limits) in proposed EcoQO metrics mean weight and mean maximum length for three North Sea surveys.

Although the paper assumes that fishing effort has continued to increase over the entire period, data to support this view are lacking. Nevertheless, there are some clear signals of changes in the size and species composition that are picked up by these metrics.

The analysis then went on to compare temporal (IBTS) trends in each metric for six North Sea areas that are distinguished on the basis of two depth classes (essentially the northern and southern North Sea, respectively) and superimposed on these three effort classes, based on average spatial distribution of total international otter and beam trawl effort (Jennings *et al.*, 1999), and (2) to compare (BTS) trends in the Plaice Box and a reference area before the closure (1985-1988), during the partial closure (1989-1994) and during the full closure (1995-2001).

However, by splitting the survey data in subsets, the variability in the haul data became progressively more important and in fact few of the temporal trends in either metric were significantly different from zero. Altogether, it remains unclear to what extent the two metrics respond at small spatial scales to differences in effort or to even big changes in effort as caused by the introduction of the Plaice Box.

2.2.4 North-Norway and West-Spitsbergen

Data from two Norwegian surveys were analysed. The North-Norway Coastal survey covers coastal banks and larger fjords from 62°N to the Russian border, mainly at 60-400m depths. The West-Spitsbergen survey covers the 100-500 m deep shelf area from Bear Island to 80°N. The standard Norwegian survey bottom-trawl (Campelen 1800) with a Rockhopper ground-gear was used on both surveys. For both surveys data for 1995-2002 were used. Although survey data from West-Spitsbergen are available further back in time, in years prior to 1995 several small species were recorded without length measurements being made. Since 1995 the sampling protocol with respect to length measurements of individual species has been unchanged and similar for both surveys.

The data analysed were numbers per 1 cm length-group per species per trawl haul. Calculations were made for all species combined and for species of each of the four species groups: Gadoids, Flatfish, Elasmobranchs, and Other demersal. Mean weight of each species group and of the total fish community was calculated from the total catch in weight and numbers per species group. Pelagic fish (herring and capelin) and invertebrates were excluded from the analyses.

Large fish were defined according to three cut off limits, namely the 60, 85 and 95 percentiles of the total combined cumulative length distribution. This corresponded to 15, 31 and 59 cm for the Svalbard data and 19, 36 and 52 cm for the North-Norway data. The upper and middle panels of Figure 2.2.4.1 show the mean fish weight and the proportion by numbers of fish larger than these cut off levels for the two surveys respectively. There does not appear to be any consistent and continuing trends in any of these measures during the time period.

The lower panels of Figure 2.2.4.1 show the development of the absolute abundance of each 20-cm length group for the two surveys respectively. It appears that the abundance of fish larger than 40 cm has increased in the North-Norway Coastal Survey while being stable in the West-Spitsbergen Survey. The abundance of the smallest size group was highly variable, but the data still indicate an increase in the West-Spitsbergen Survey.

Further studies are necessary to evaluate the effect that fishing might have on these trends. Data on L_{max} of the northern and arctic species that dominate in these surveys were not available to the working group. Before next years meeting these data should be established together with other and preferably longer survey series.

West-Spitsbergen Survey North-Norway Coastal Survey

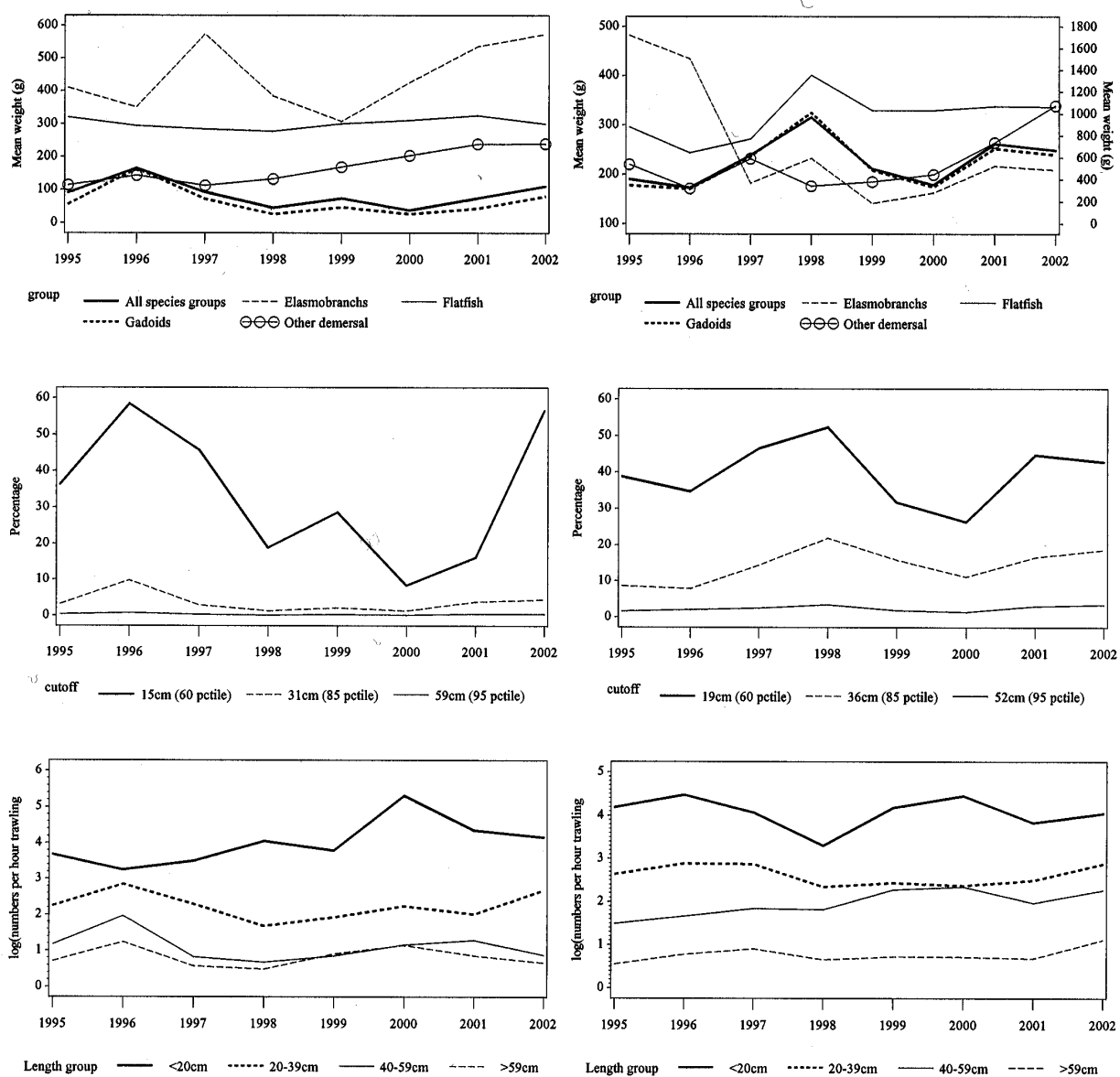


Figure 2.2.4.1 Temporal trends in three measures of community size structure: Mean weight of individual fish (upper), proportion of large fish (middle), and mean catch rate of individual length-groups (bottom). The right axis on the upper right figure is for the elasmobranchs.

2.2.5 Celtic Sea

2.2.5.1 Methods

Abundance of each Celtic Sea fish species was collected using standardised trawl data from ICES divisions VII-f-j (Warnes and Jones 1995; Pinnegar *et al.* 2002). Trawl samples were collected from using Portuguese high headline gear towed for one hour at between 46-64 stations each year, except for 1994 when 32 stations were sampled (Figure 2.2.5.1.1.). Pelagic species are likely to be well sampled as the surveys were originally designed to survey mackerel. Annual trawl surveys were conducted in quarter 1 and only data from 1987-2002 were analysed. Individual fish was measured to the centimetre below. Unfortunately no comparable data on levels of fishing effort were available.

Proportion of large fish

The three categories of 'large' fish were defined as the 60th, 85th and 95th percentiles of a cumulative frequency distribution of the total number of individuals caught throughout the survey. Proportion data were arcsine square root transformed for regression analysis.

Average weight

The average weight of all individuals of all species caught each year was estimated as the total weight (kg) of species divided by the total number of individuals caught in year (Zwanenburg 2000). All species captured in the survey were retained for analysis (N = 117), separate analyses were conducted for pelagic teleosts, demersal teleosts, elasmobranchs and the boarfish (*Capros aper*).

Assemblage L_{max}

The average maximum length of the Celtic Sea fish assemblage caught by year was estimated as:

$$L_{max} = \sum n_{ij} \cdot L_i / N_j$$

Where n_{ij} is the relative abundance (no. hr⁻¹) of species i caught in each year j , L_i is the maximum length reported in the literature and N_j is the relative abundance summed across all species. Information on the maximum length of each species was collated from a regional faunal reference (Table 2.2.5.1.1.). Species were excluded from the analysis if they (a) were captured in only one year, and (b) if the identified to genus only. A total of 84 species remained for analysis (Table 2.2.5.1.1.).

Table 2.2.5.1.1 The scientific and common names and their maximum recorded lengths of Celtic Sea fishes. From Wheeler (1978) except for Mediterranean horse mackerel, *Trachurus mediterraneus* from Fishbase (Froese and Pauly 2000).

Scientific name	Common name	Maximum length (cm)
<i>Agonus cataphractus</i>	pogge	20
<i>Alosa alosa</i>	Allis shad	60
<i>Alosa fallax</i>	Twaite shad	55
<i>Arnoglossus imperialis</i>	imperial scaldfish	25
<i>Arnoglossus laterna</i>	scaldfish	19
<i>Aspitrigla cuculus</i>	red gurnard	40
<i>Belone belone</i>	garfish	94
<i>Blennius ocellaris</i>	butterfly blenny	20
<i>Buglossidium luteum</i>	solenette	13
<i>Callionymus lyra</i>	common dragonet	30
<i>Callionymus maculatus</i>	spotted dragonet	14
<i>Cepola rubescens</i>	red bandfish	70
<i>Clupea harengus</i>	herring	43
<i>Conger conger</i>	conger eel	274
<i>Ctenolabrus rupestris</i>	goldsinny	18
<i>Dicentrarchus labrax</i>	bass	100
<i>Dipturus batis</i>	common skate	285
<i>Echiichthys vipera</i>	lesser weever	14
<i>Echiodon drummondi</i>	pearlfish	32
<i>Enchelyopus cimbrius</i>	fourbearded rockling	41
<i>Engraulis encrasicolus</i>	European anchovy	20
<i>Eutrigla gurnardus</i>	grey gurnard	45
<i>Gadiculus argenteus</i>	silvery pout	25
<i>Gadus morhua</i>	cod	120
<i>Gaidropsarus mediterraneus</i>	shore rockling	35
<i>Gaidropsarus vulgaris</i>	three bearded rockling	53
<i>Galeorhinus galeus</i>	tope	167
<i>Galeus melastomus</i>	blackmouth dogfish	75
<i>Glyptocephalus cynoglossus</i>	witch	55
<i>Helicolenus dactylopterus</i>	bluemouth rockfish	46
<i>Hippoglossoides platessoides</i>	longrough dab	48
<i>Lepidorhombus boscii</i>	fourspot megrim	41
<i>Lepidorhombus whiffiagonis</i>	megrim	61
<i>Leucoraja fullonica</i>	shagreen ray	120
<i>Leucoraja naevus</i>	cuckoo ray	70
<i>Limanda limanda</i>	dab	42
<i>Lophius budegassa</i>	black-bellied anglerfish	82
<i>Lophius piscatorius</i>	anglerfish	200
<i>Macrorhamphosus scolopax</i>	snipefish	15
<i>Malacocephalus laevis</i>	softhead grenadier	56
<i>Maurollicus muelleri</i>	pearlside	6.5
<i>Melanogrammus aeglefinus</i>	haddock	76
<i>Merlangius merlangus</i>	whiting	70
<i>Merluccius merluccius</i>	hake	180
<i>Microchirus variegatus</i>	thickback sole	33
<i>Micromesistius poutassou</i>	blue whiting	45
<i>Microstomus kitt</i>	lemon sole	66
<i>Molva molva</i>	ling	200
<i>Mullus surmuletus</i>	striped red mullet	40

Scientific name	Common name	Maximum length (cm)
<i>Mustelus asterias</i>	starry smooth hound	180
<i>Mustelus mustelus</i>	smoothhound	160
<i>Nezumia aequalis</i>	common Atlantic grenadier	41
<i>Pagellus bogaraveo</i>	red sea bream	51
<i>Petromyzon marinus</i>	sea lamprey	91
<i>Phycis blennioides</i>	greater forkbeard	75
<i>Platichthys flesus</i>	flounder	51
<i>Pleuronectes platessa</i>	plaice	91
<i>Pollachius pollachius</i>	pollack	130
<i>Pollachius virens</i>	saithe	130
<i>Raja brachyura</i>	blonde ray	113
<i>Raja clavata</i>	thornback ray	120
<i>Raja microocellata</i>	smalleyed ray	82
<i>Raja montagui</i>	spotted ray	75
<i>Sardina pilchardus</i>	pilchard	25
<i>Scomber japonicus</i>	Spanish mackerel	40
<i>Scomber scombrus</i>	mackerel	66
<i>Scophthalmus maximus</i>	turbot	100
<i>Scophthalmus rhombus</i>	brill	75
<i>Scorpaena scrofa</i>	largescaled scorpionfish	51
<i>Scyliorhinus canicula</i>	lesser spotted dogfish	70
<i>Scyliorhinus stellaris</i>	greater spotted dogfish	152
<i>Solea solea</i>	sole	60
<i>Spondyliosoma cantharus</i>	black sea bream	51
<i>Sprattus sprattus</i>	sprat	16.5
<i>Squalus acanthias</i>	spurdog	122
<i>Torpedo nobiliana</i>	Atlantic torpedo	180
<i>Trachinus draco</i>	greater weaver	41
<i>Trachurus mediterraneus</i>	Mediterranean horse mackerel	67
<i>Trachurus trachurus</i>	horse mackerel	50
<i>Trigla lucerna</i>	tub gurnard	75
<i>Trigloporus lastoviza</i>	streaked gurnard	36
<i>Trisopterus esmarki</i>	Norway pout	25
<i>Trisopterus luscus</i>	bib	41
<i>Trisopterus minutus</i>	poor cod	26
<i>Zeus faber</i>	John Dory	40

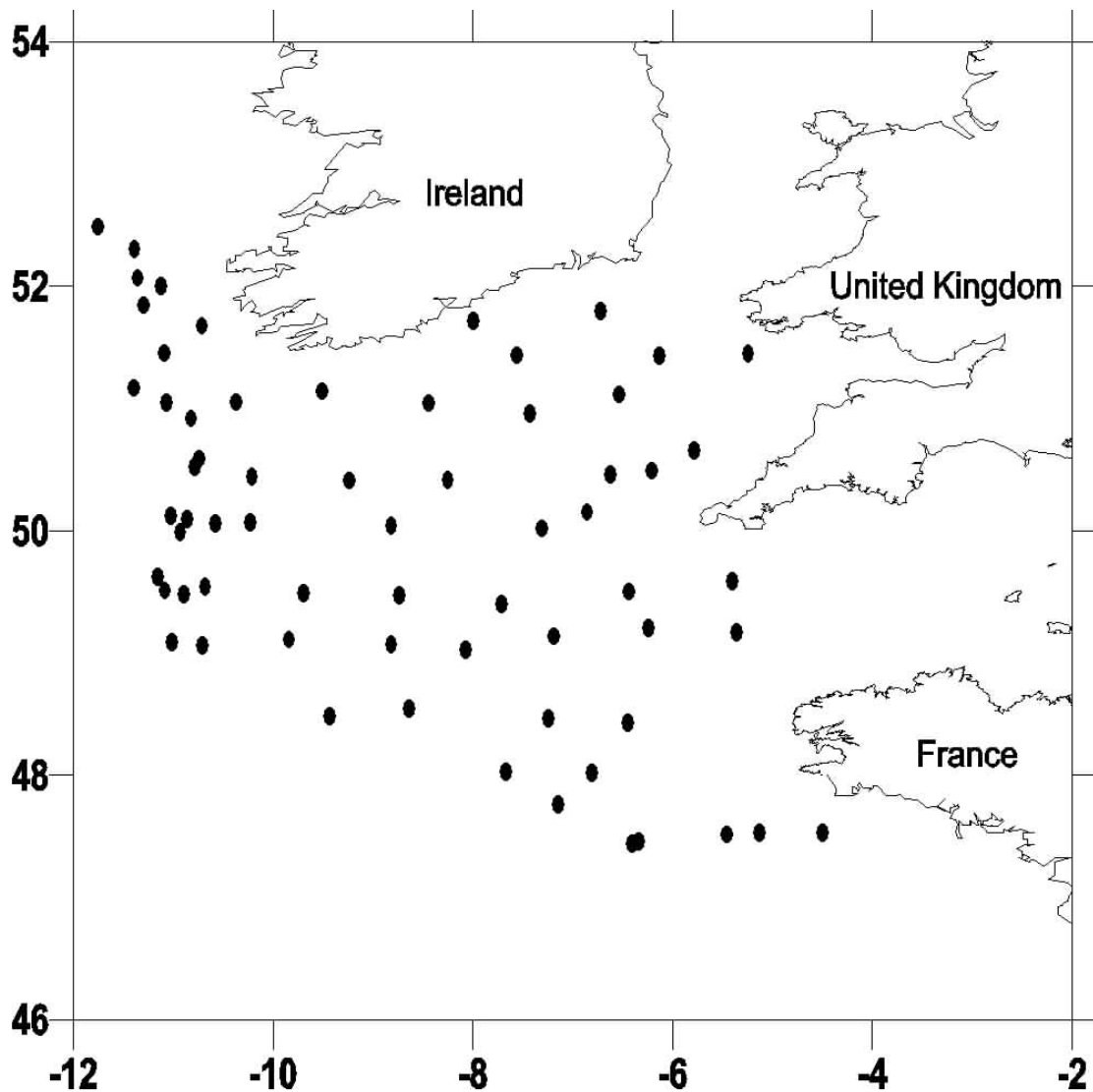


Figure 2.2.5.1.1. The position of CEFAS groundfish survey stations (points) in the Celtic Sea.

Results

Proportion of large fish

There is a slight declining and significant trend observable in proportion of large fish above the 60th percentile for all species combined ($r^2 = 0.34$, $df = 14$, $F = 7.01$, $P = 0.02$), but there are two peaks in the proportion of large fish centred on 1991 and 1997-8. There are significant declines in the proportion of large elasmobranchs, but only when using the 60th and 85th percentile cut off (60th: $r^2 = 0.35$, $df = 14$, $F = 7.6$, $P = 0.02$; 85th: $r^2 = 0.34$, $df = 14$, $F = 7.02$, $P = 0.02$; 95th: $r^2 = 0.04$, $df = 14$, $F = 0.6$, $P = 0.46$). No clear pattern was observed in the proportion of large pelagic and demersal teleosts (Figure 2.2.5.2.1.).

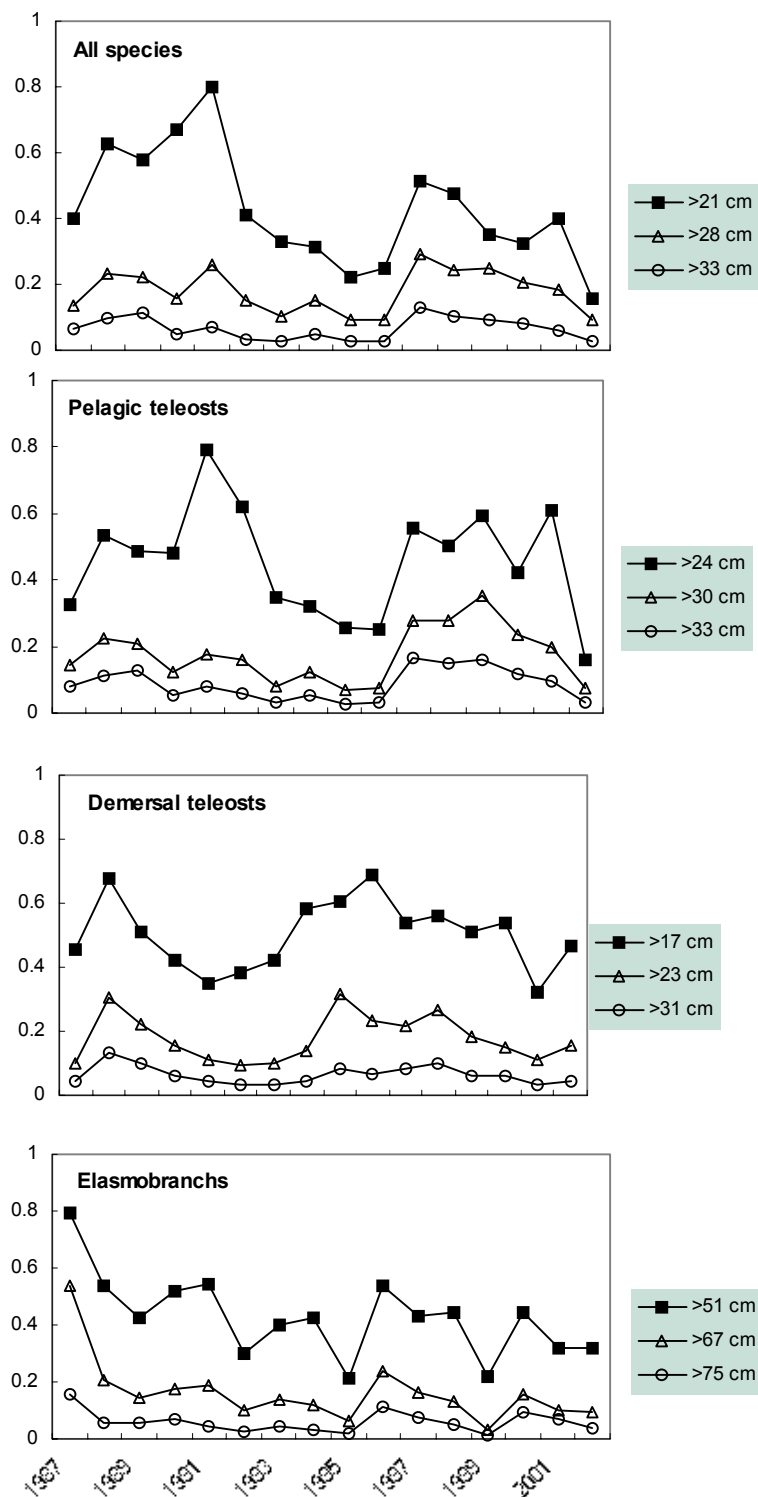


Figure 2.2.5.2.1. Trends in the proportion of large fish above the 60th, 85th and 95th percentile between 1987-2002 in the Celtic Sea for (a) all species, (b) pelagic teleosts, (c) demersal teleosts and (d) elasmobranchs.

Average weight of fishes

The average weight of fishes in the Celtic Sea was calculated for (i) all fishes, (ii) elasmobranch fishes, (iii) pelagic fishes, (iv) boarfish and (v) demersal fishes (Figure 2.2.5.2.2.). Regression analysis was used to determine whether or not average weight (Log_{10} transformed) showed a significant linear trend over time. Although the average weight of boarfish increased significantly, none of the other groups showed a significant linear trend over the time period examined. What was apparent for the average weight of pelagic fishes was that oceanographic factors may also be important.

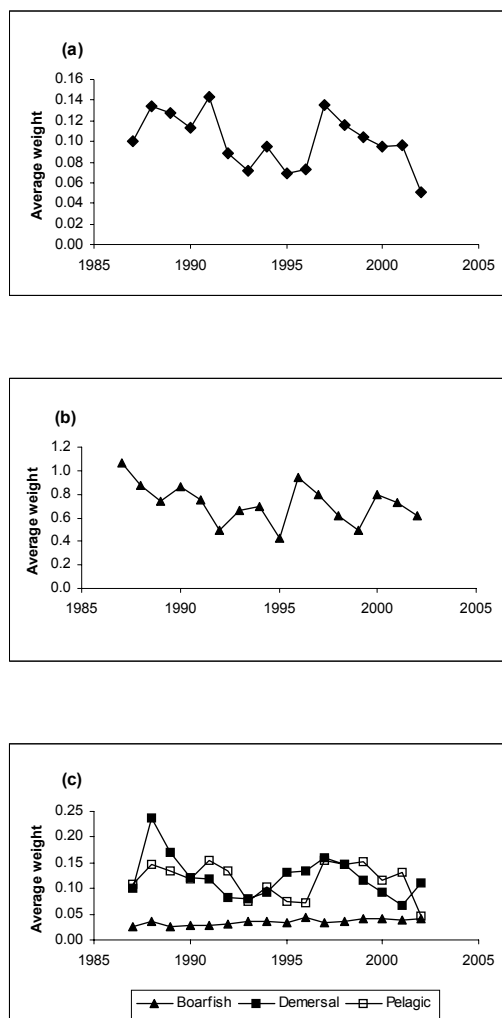


Figure 2.2.5.2.2. Temporal change in the average weight of (a) all fishes ($r^2=0.20$; $df=14$; $F=3.44$; $p=0.085$), (b) elasmobranchs ($r^2=0.13$; $df=14$; $F=2.14$; $p=0.17$), and (c) demersal fishes ($r^2=0.11$; $df=14$; $F=1.72$; $p=0.21$), pelagic fishes ($r^2=0.07$; $df=14$; $F=1.07$; $p=0.32$) and boarfish ($r^2=0.58$; $df=14$; $F=19.6$; $p=0.001$) caught in the Celtic Sea survey (CEFAS Q1 survey with Portuguese High Headline Trawl).

Average maximum length

Overall there was a slight decline in the average maximum length (L_{\max}) of the Celtic Sea assemblage but the data were highly variable in time exhibiting peaks around 1988-1991 and 1997-9. (Figure 2.2.5.2.3a). This pattern was also apparent in pelagic species, but demersal teleosts exhibited a significant increase in average maximum length (Figure 2.2.5.2.3b). Elasmobranchs exhibited a declining assemblage L_{\max} from 1987-1995 whereupon a increase was observed up to 2000, which has subsequently exhibited decline (Figure 2.2.5.2.3c).

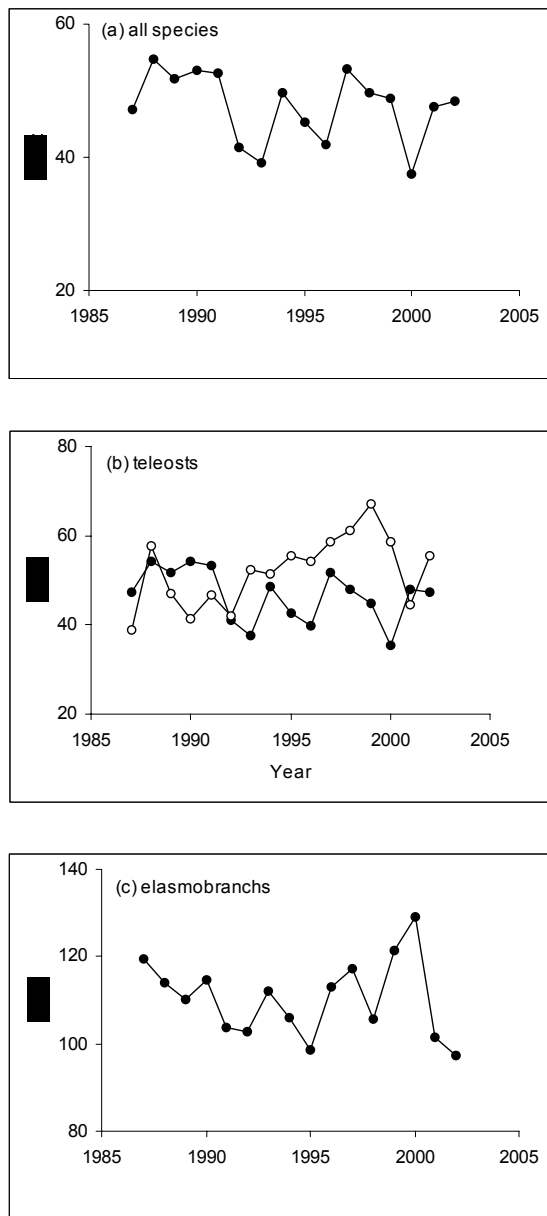


Figure 2.2.5.2.3. Trends in mean maximum length (L_{\max}) of Celtic Sea fish assemblage from 1986-2002 for (a) all 84 species, (b) pelagic (solid symbols) and demersal (hollow) teleosts and (c) elasmobranchs.

2.2.6 Portuguese continental waters

The same data set was used as has already been analysed in two previous working group meetings (ICES 2001; ICES 2002b). These data are from IPIMAR autumn demersal surveys that have been carried out annually since 1982 (Cardador *et al.* 1997). Here, however, only data from 1990 on have been used as length distribution data was not available for all species for the 1980's. Three metrics were tested with these data: the proportion of large fish, trends in L_{\max} over time and average weight over time.

Proportion of large fish

The proportion of large fish in the whole community was calculated by calculating the cumulative frequency of the total number of individuals caught during the entire data set. Three cut off limits were chosen: 60, 85 and 95% of the total, above which fish were considered "large". These gave large fish values of 20, 25 and 33 cm respectively. Two species, snipefish (*Macroramphosus sp.*) and boarfish (*Capros aper*) were excluded from the data set as these two species are extremely variable and abundant small migratory species that follow processes that are not fully understood, and which dominate the Portuguese assemblage, constituting up to 76% and 8% of all individuals respectively. When they were included in the data set, large fish values were 12.5, 14.5 and 18 cm. Figure 2.2.6.1. shows the abundance of both boar and snipe fish caught during the autumn surveys during this period. The cumulative frequency of the total number of individuals caught per year was calculated and the proportion of fish at the three sizes plotted (Figure 2.2.6.2.).

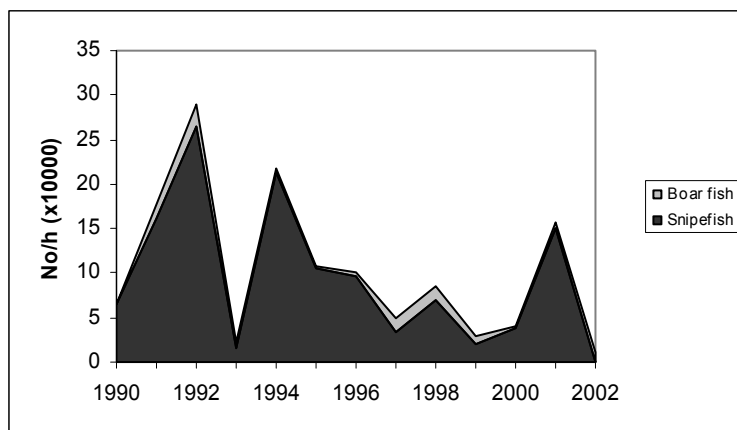


Figure 2.2.6.1. Abundance (No/h x 10000) for snipefish and boarfish for the period in consideration.

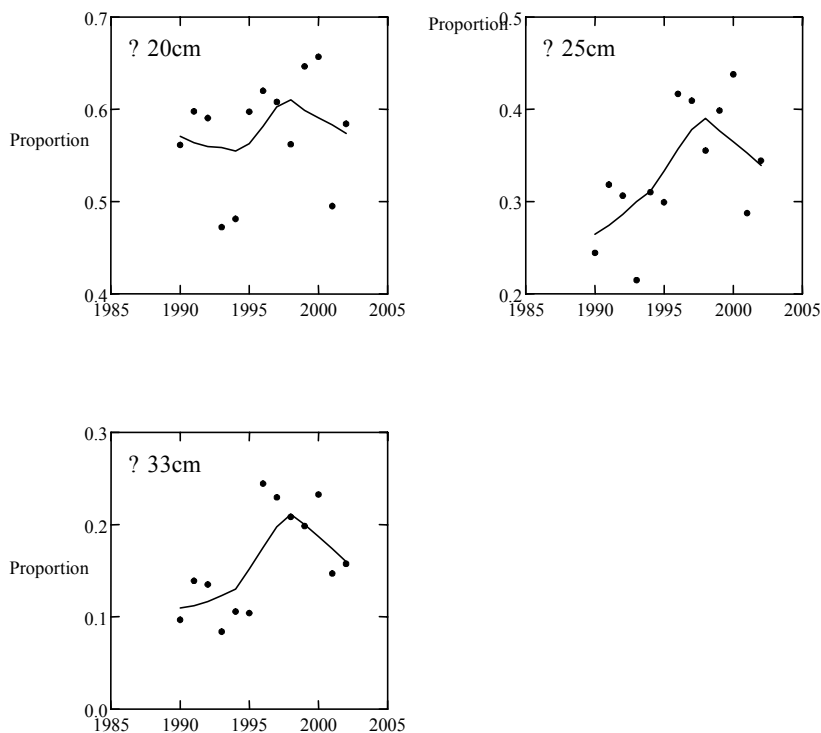


Figure 2.2.6.2.. Trend over time of the proportion of large fish. Cut off limits at 60, 85 and 95th percentiles fitted with LOWESS curves.

From this figure it can be seen that the proportion of large fish has increased slightly over time when large fish are considered to be greater than 33cm. A linear fit to the data was statistically significant (P value: 0.048; slope: 0.008). The relationship was not significant when large fish are considered to be greater than 20cm or greater than 25 cm. There is however high variability in all these data.

An increase in the proportion of large individuals could be considered to be indicative of a recovering ecosystem. Considering landings over the past 50 years (Figure 2.2.6.3.), it can be seen that there was a large (5-fold) increase in landings in the early 1970's, but since then there has been a steady decrease to present levels (17 thousand tonnes in 1999). Effort data (as hours trawling per year) are available for the period 1950 to 1987 (Cardador 1988), and it can be seen that during this time effort follows the same pattern exhibited by landings.

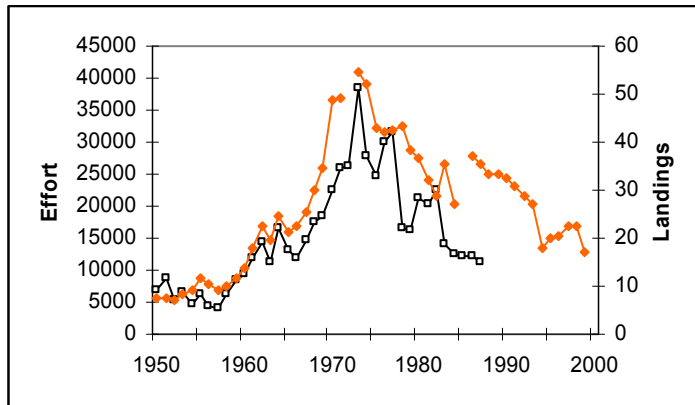


Figure 2.2.6.3. Evolution of effort (open squares) as hours trawling (x 10000) and landings (solid diamonds) as thousand tonnes over time in Portuguese continental waters for the period 1950 to 1999.

L_{max} over time

Average L_{\max} per year was calculated using L_{\max} values from the literature and using total numbers of individuals per year. These data were compiled during the 2001 WGECO meeting (ICES 2001). For these analyses snipefish and boarfish were excluded from the data set for the reasons explained above. Average L_{\max} over time is shown in Figure 2.2.6.4., and data were also plotted for four assemblage groups (demersal species, pelagic species, elasmobranchs and snipefish and boarfish) (Figure 2.2.6.5.).

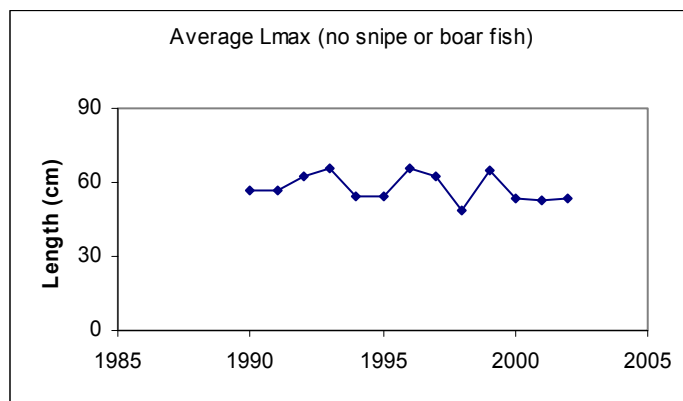


Figure 2.2.6.4. Average L_{\max} over time for all species (except snipe and boarfish) in Portuguese continental waters.

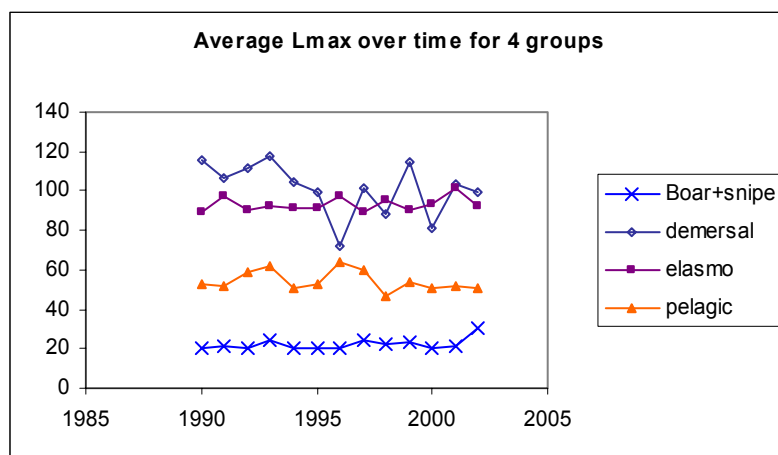


Figure 2.2.6.5. Average L_{\max} over time for four assemblage groups over time in Portuguese Continental waters.

No trends were observed for any of the groups, although for boar and snipefish it can be seen that in 2002 Average L_{\max} increases. This is because during this year more boarfish were caught than snipefish (about 11,000 boarfish/h compared to 42 snipefish/h), and boarfish has an L_{\max} of 30cm, whereas snipefish's L_{\max} is 20cm.

Average weight over time

Average weight per year was calculated using the total weight divided by the number of individuals per year. For these analyses snipe and boar fish were excluded from the data set for the reasons explained above (Figure 2.2.6.6.). Average weight over time was also plotted for three assemblage groups (demersal species, pelagic species, elasmobranchs), snipe fish and boarfish were not included in this graph as they have similar and constant weights over time (Figure 2.2.6.7.). None of the analyses showed any type of trend.

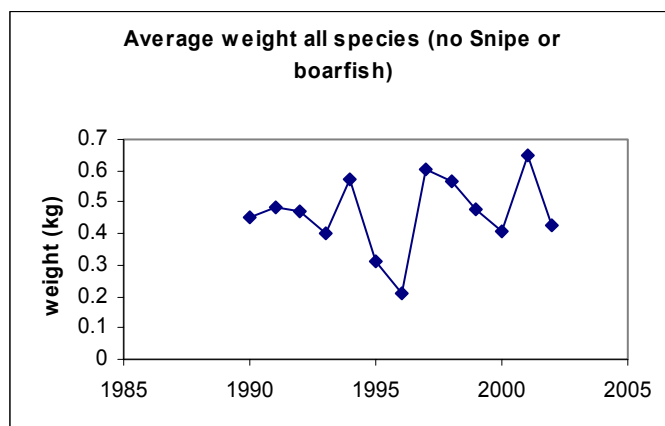


Figure 2.2.6.6. Average weight over time for all species except snipe and boarfish in Portuguese Continental waters.

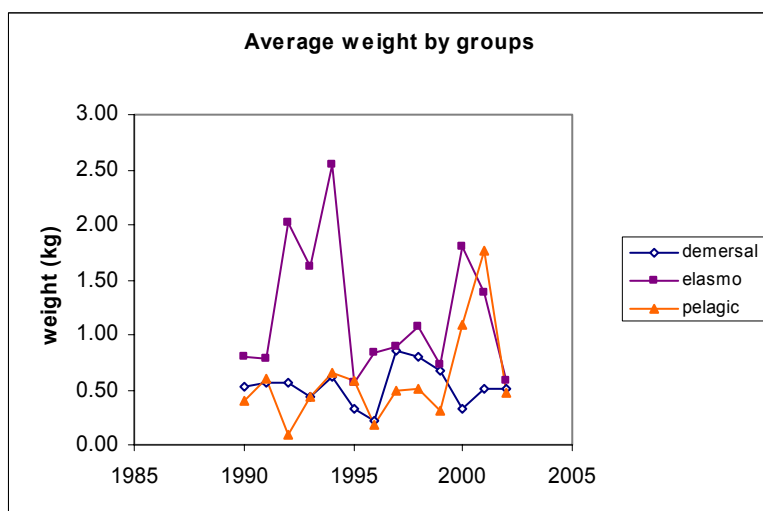


Figure 2.2.6.7. Average weight over time for three assemblage groups over time in Portuguese Continental waters.

2.2.7 Summary

The working group considered survey series within six geographic areas, from south to north: Portuguese shelf, central North Sea, Celtic Sea, North-West North Sea, North-Norway coast, and West-Spitsbergen shelf. Table 2.2.7.1. summarises the results in terms of identified trends in the three different ecosystem metrics considered. The trends were generally negative for the longer time series while absent in the shorter series. The only exception was an increase in the percentage of large fish on the Portuguese shelf.

Table 2.2.7.1. Summary of trends in percentage of large fish, mean individual weight and mean L_{max} within the different survey series considered. +/-: Data suggest an increase/decrease of the metric for all species combined within at least some parts of the survey area; *: The trend is statistically significant with $p < 0.01$; 0: Data do not suggest any clear linear trend in the metric for all species combined.

<i>Area</i>	<i>Time span</i>	<i>Trend in % large</i>	<i>Trend in mean W</i>	<i>Trend in mean L_{max}</i>
Portuguese shelf	1990-2002	+	0	0
North Sea	1970-2002	n.a.	—*	—*
Celtic Sea	1987-2002	—	—	—
North-West North Sea	1925-1996	—	—*	—*
North-Norway coast	1995-2002	0	0	n.a.
West-Spitsbergen shelf	1995-2002	0	0	n.a.

It is not easy to derive a consistent interpretation of the temporal trends in average weight and average maximum length presented by the various data sets, primarily because a suitable metric of the exploitation rate of each community is lacking for most areas. For the North Sea, enough information appears to be available to estimate some overall trend for the last 30 years as well as to classify rectangles according to average recent effort. Globally, the data presented appear to confirm that higher effort is related to a lower average weight and a lower average maximum length. For the other areas, where trends in effort data are lacking, we really have no clue as to what kind of trends in metrics might be expected. Thus, these data tell us largely something about the interannual variability in each metric.

Nevertheless, a few observations can be made. First of all, the emerging pattern in any metric strongly depends on the taxonomic groups included in the analysis. While guilds may be defined at whatever level seems practical and suitable, this obviously brings in some arbitrary choices that are not easy to defend if they are to represent “the fish community” in terms of ecological quality. In other words, if we specify that the metric should be typically applied to demersal species excluding elasmobranchs, additional EcoQO elements may be required for pelagics and elasmobranchs to ensure that total ecological quality of the fish community will be accounted for in management.

The apparent close association between average weight of pelagics and the NAO index brings out another problem. At this stage, we are by no means certain that the major factor influencing these metrics is always exploitation rate and environmentally driven changes may well occur. For example, there was great similarity in the 2-year running means of NAO and average weight of pelagic fishes in the Celtic Sea data set (Figure 2.2.7.1.). Further work on the comparative influences of fishing effort and other environmental factors is definitely required.

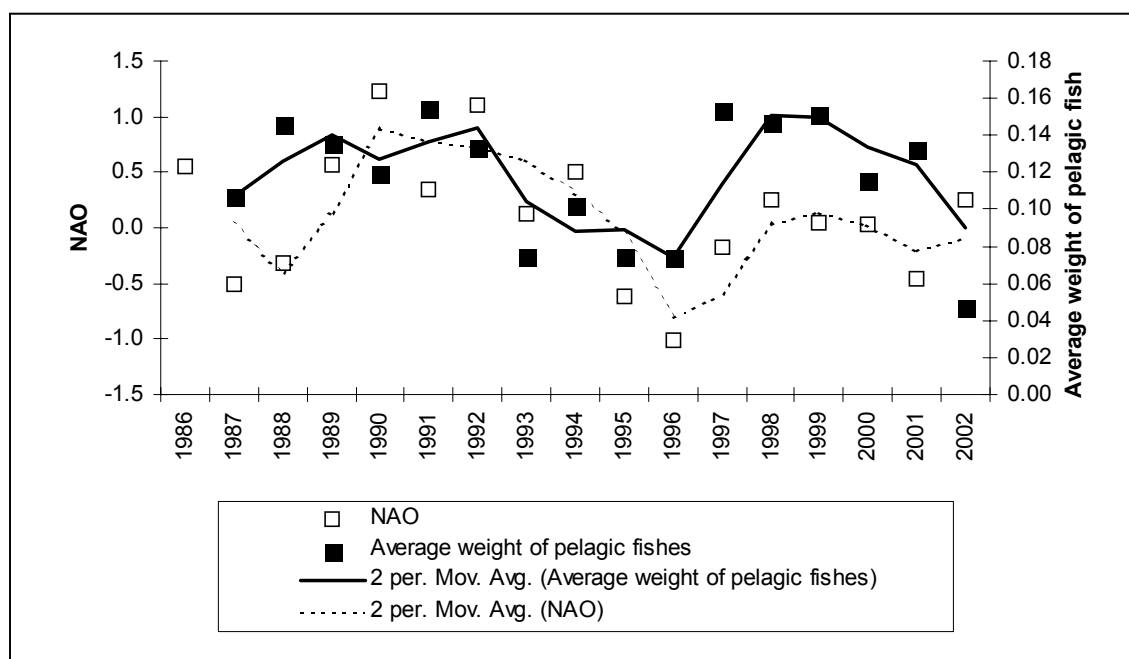


Figure 2.2.7.1. Temporal change in the average weight of pelagic fishes caught in the Celtic Sea during and the NAO index. The trend lines represent the 2-year running mean.

In many time series the metrics show signs of considerable autocorrelation. While this might be expected for metrics that are essentially conservative in the sense that information of last year is carried over to the new one, we have to be careful when drawing firm conclusions from simple significance statistics.

The analyses have revealed shortcomings in the quality and quantity of sampling and recording in the early years of some of the survey time series. Therefore some of the series had to be truncated in order to make the different measures comparable throughout the period analysed. This may not be surprising since the type of analyses applied are to some extent new compared to what was conceived relevant at that time of sampling. However, since long-term data-series are crucial in determining ecosystem quality it is recommended that these and other survey series be further evaluated and validated with respect to consistency in sampling protocol relevant to the different EcoQ metrics. Such evaluation is also important for future improvements to the sampling protocols. Before the next meeting of the working group, the longest time series possible should be established and validated for a few more geographic areas together with data on fishing effort. This will facilitate inter-regional comparisons of different candidate metrics of ecosystem quality.

2.3 Evaluation of data and trends in other size and maximum-length characteristics.

The two metrics proposed by OSPAR for fish-community EcoQO's (average weight, average maximum length) have the advantage that their meaning is easily understood by non-scientists and that they are clearly related to exploitation rate. However, representing the integrated effects of exploitation on a community by just one or two overall metrics also has severe disadvantages, because similar variations in each metric may reflect totally different effects. For instance, a reduction in average weight may be equally caused by a reduction in absolute abundance of large fish or by an increase in absolute abundance of small fish. Without additional information, the metric itself provides no clue, while more information would be warranted when interpreting observed changes in terms of ecosystem quality. More generally, the two metrics refer only to relative changes rather than to absolute changes. Thus, it would be entirely possible that total abundance in numbers and biomass declines without a marked effect on either metric. Therefore, it would seem at least doubtful whether sufficient information is contained in the agreed EcoQO element to ensure that overall ecological quality at the fish community level is maintained.

A working paper presented at the meeting (Daan *et al.* 2003) illustrated some of these problems for the North Sea. This initiated a range of new analyses aimed at disentangling the detailed changes underlying the observed changes in the metrics proposed.

2.3.1 North Sea

This section is based on work by Daan et al. (in prep).

2.3.1.1 Methods

Surveys

The analyses are based on three data sets: IBTS (International Bottom Trawl Survey; February surveys only; Heessen *et al.*, 1997; ICES, 1999), BTS (Beam Trawl Survey; van Beek, 1997) and DYFS (Demersal Young Fish Survey; Boddeke *et al.*, 1970; van Beek, 1997). These surveys differ in geographical coverage and gears used, but catches are always sorted to species and samples of each species are measured.

Based on the swept area estimates and to get roughly comparable cpue estimates in respect of swept area, data collected during the different surveys were raised to match the IBTS (150 000 m²). In the averaging process, hauls made within the same ICES rectangle in a particular year were averaged first before an average was taken over the entire survey. Primary data selections and analyses were made in SAS and followed up in EXCEL.

Size spectra

To get a detailed view of changes in absolute abundance for fish of different sizes, fish were grouped in 10 cm length classes. All species were included but a species group code (pelagics, gadoids, flatfish, other demersal and elasmobranchs) and for the IBTS a roundfish area code (1-8) was kept during the data extraction process to investigate the generality of some of the findings. The North Sea covers a large area with different communities inhabiting different regions (Daan *et al.*, 1990) that do not necessarily exhibit similar spectra. Also different fleets operate in different areas and spatially different responses of the communities may be expected. For the other two surveys such analyses were not warranted because of their restricted coverage.

All information from hauls with incomplete coverage of length measurements for all species was rejected *a priori*.

L_{max} spectra

When dealing with species composition of survey catches, some imminent problems emerge. First of all, the definition of community becomes important, because each species contributes in a singular way to the parameterization. In this case, it seemed more appropriate to focus on the community typically exploited by demersal gear and to exclude both pelagic species and vagrants to get rid of the high variability caused by these. In addition, preliminary inspection of the data revealed that elasmobranchs just created random noise in the data and these were also excluded.

After thus having defined the 'typical' North Sea demersal fish community, the problem of proper species identification remained. Daan (2001b) provides considerable evidence that identification of several species has been mixed up by different crews at different times, while species were not always identified to species level. We resolved this by assigning the appropriate genus (or family) name to apparently mis-identified taxa and then split the associated catch among the relevant species according to the fractions observed in hauls with trusted species information made by other countries in the same square and in the same year, or if this cell was empty, the average fraction in that square over the entire survey period. Such correction procedures were applied to species belonging to the genera *Callionymus*, *Liparis* and *Sebastes* and to the families Soleidae, Bothidae and Cottidae.

For the remaining species in the data set, L_{max} values were derived from Daan (2001a). Each record was then assigned a ln-L_{max} class (<3, 3-3.5, ..., 4.5-5, >5). Before summing over the length compositions within each class, the corresponding weight within each length class (midpoint L) was calculated as $W=0.01 \cdot L^3$. No effort was undertaken to assign individual condition factors to species, because this was considered a minor source of variation in survey data. Annual spectra were derived in terms of cpue in both numbers and weights by ln L_{max} class. Because communities vary considerably among areas, the comparisons among surveys were restricted to information from roundfish area 6 only. However, IBTS data were analysed also by roundfish area to investigate whether similar trends have occurred over larger areas.

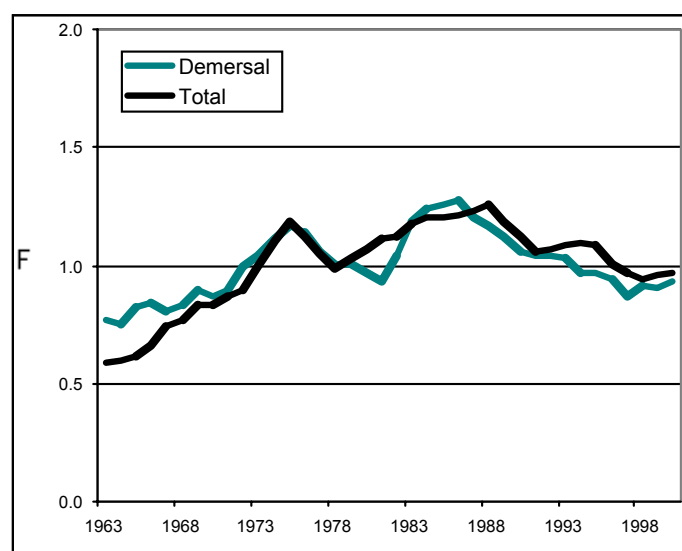


Figure 2.3.1.1.1. Average community exploitation index based on the corresponding normalized indices for the 10 major commercially important North Sea species derived from multispecies virtual population analysis.

Trends in exploitation

An integrated measure of exploitation of the North Sea fish community based on all fleets does not exist and an exploitation index of the entire community has to be based on estimates of fishing mortality available for the major exploited stocks. Although we could have used information from routine single-species assessments, we chose to use the most recent run of Multispecies Virtual Population Analysis (ICES, 2002c), because (1) it provides a coherent data set for cod, haddock, whiting, saithe, plaice, sole, herring, sandeel, sprat and Norway pout, (2) the tuning process guarantees approximately similar results for recent years and (3) takes historic changes in predation rates fully into account. There is no straightforward way to combine these estimates into a single index, because a large part of the estimated mortalities on individual species is exercised simultaneously by mixed fisheries and therefore might be counted twice. We therefore chose to normalize the F-values for individual species and to calculate arithmetic average values for different components (roundfish, flatfish, pelagics=herring and industrial). We also used a weighting procedure by the average biomass of each species over the entire period. Because the resulting patterns were essentially similar, we chose to work with straightforward averages, but smoothed them in threes for ease of interpretation of trends. Since size spectra were based on all species and L_{\max} spectra only on demersal fish, overall indices of exploitation rate (F), spawning stock biomass (SSB) and recruitment (R) were calculated for all groups combined and for flatfish and roundfish only (Figure 2.3.1.1.1.). However, they broadly convey similar trends and suggest that exploitation has consistently increased up to approximately 1985 and has been stable or has slightly decreased afterwards.

2.3.1.2 Results

Size spectra

Figure 2.3.1.1.2. presents a detailed view of the temporal change in \ln cpue by size class in the three surveys. The IBTS shows significant increases in the three smallest size classes, whereas abundance of all larger size classes have significantly declined over the entire period. The trend for the smallest size class in the BTS was not significant, but up to 50 cm all trends were significantly negative. Only the DYFS showed significant declines in abundance of both small and large (up to 50-60) size classes. A comparison with the trend in exploitation rate (Figure 2.3.1.1.1.) suggests that most of the increase in abundance of small fish occurred before 1985, while their abundance remained fairly stable afterwards. In contrast, large fish continued to decline until recent years.

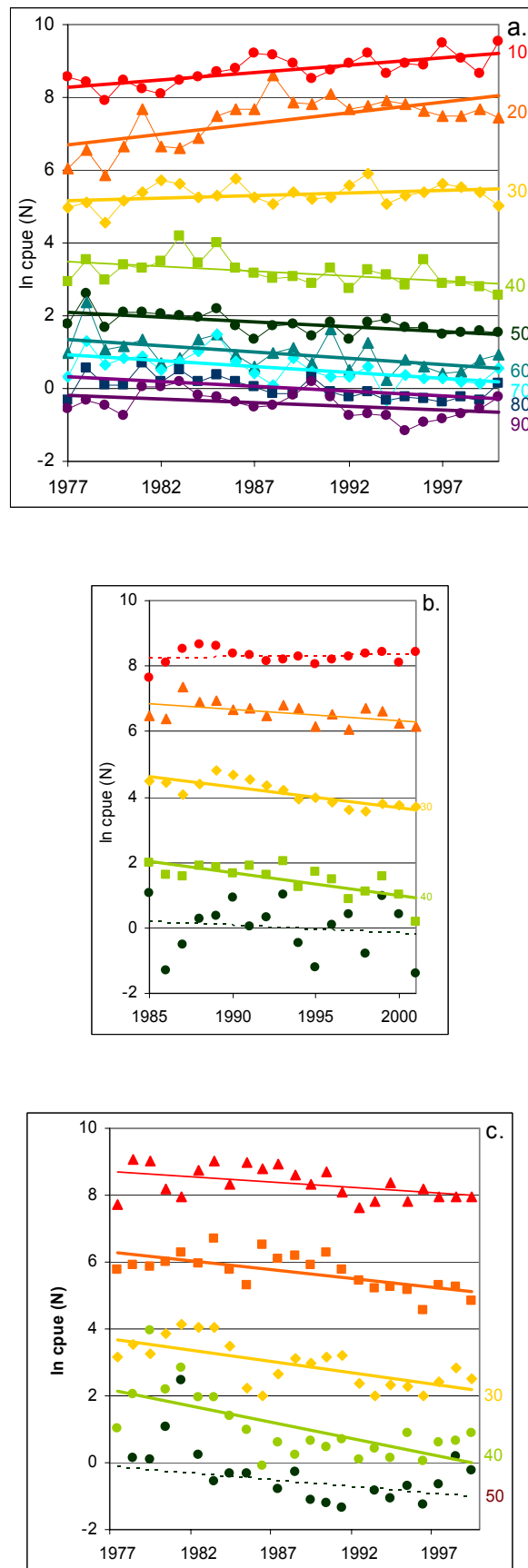


Figure 2.3.1.1.2. Ln cpue by size class (all species included) in (a) IBTS, (b) BTS and (c) DYFS.

Figure 2.3.1.1.3. compares trends in different size classes defined as small (<30 cm), medium (30-50 cm) and large (>50 cm) fish among the eight roundfish areas (IBTS data). The increase in abundance of small fish was significant in five areas, while the non-significant trends were also positive. Also medium sized fish increased significantly in four areas. In contrast, large fish declined significantly in five areas. The overall picture emerging is that increases in small fish were most pronounced in the central areas of the North Sea and in the Skagerrak/Kattegat, while large fish declined in the central-southern areas and Skagerrak/Kattegat. These results clearly indicate a large spatial coherence in the observed phenomena.

Figure 2.3.1.1.4. compares the abundance values by size class obtained for different groupings. With the exception of pelagic species, where medium fish have increased (probably reflecting the revival of the herring stock after its collapse), the same tendency is reflected by all groupings, with abundance of small fish increasing and of large fish declining. Thus, the response is neither restricted to specific areas nor to a limited number of species.

L_{max} spectra

The ln-abundance of successive ln- L_{max} classes (demarcation points: 3, 3.5, 4, 4.5, 5 corresponding to 20.1, 33.1, 54.6, 90.0 and 148.4, respectively) in roundfish area 6 for the three surveys (Figure 2.3.1.1.5.) reveal a consistent pattern with a marked increase in species with low L_{max} , stable values for species with an intermediate L_{max} and a decline in species with a high L_{max} . These changes do not stop around 1985, but appear to continue until the most recent years, even though exploitation rate appears to have declined. Ln cpue in weight essentially follow the same relative patterns, but at different levels because of the third power involved (not shown). However, the time series of relative weight fractions by L_{max} class, although indicating similar trends, clearly show more stability at least in the IBTS and BTS data, because both small and large L_{max} classes contribute relatively little to the total abundance (Figure 2.3.1.1.6.). The figure also shows clearly that the assemblages sampled differ markedly among surveys. For instance the signal for species with a low L_{max} is hardly visible in the IBTS data and species with a high L_{max} are underrepresented in the BTS.

Figure 2.3.1.1.7. shows the estimated trends for selected L_{max} by roundfish areas. With the exception of area 2, the abundance in the smallest L_{max} class (<3 ~ <20.1 cm) has increased significantly ($p < 0.01$) throughout the North Sea. For subsequently higher size classes, the results are more variable. With the exception of roundfish area 6, large fish do not exhibit a significant decline anywhere.

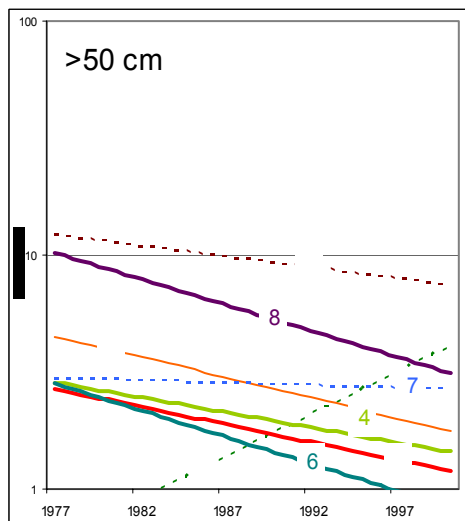
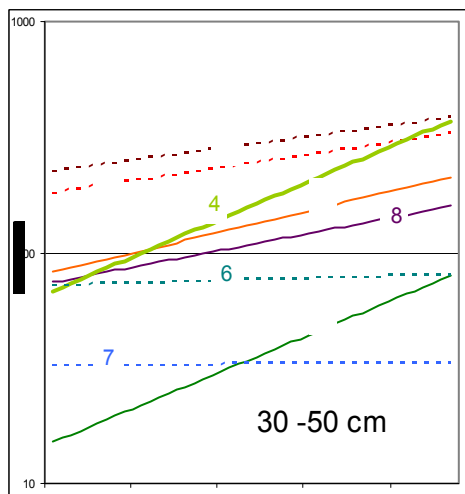
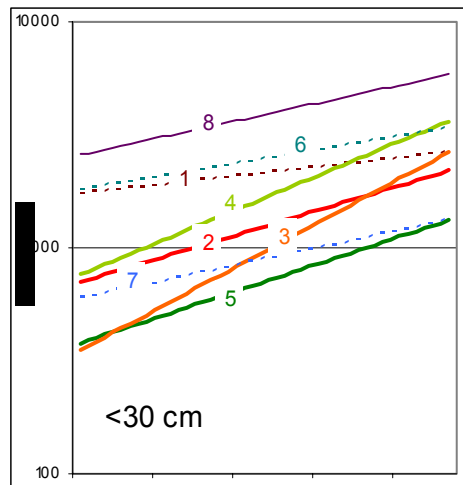


Figure 2.3.1.1.3. Temporal trends (heavy lines: $p < 0.01$; thin lines: $p < 0.05$; dashed lines: not significant) in cpue in N per hr fishing by length class (a: < 30 ; b: $30 - 50$; and c: > 50 cm) and roundfish area (1-8).

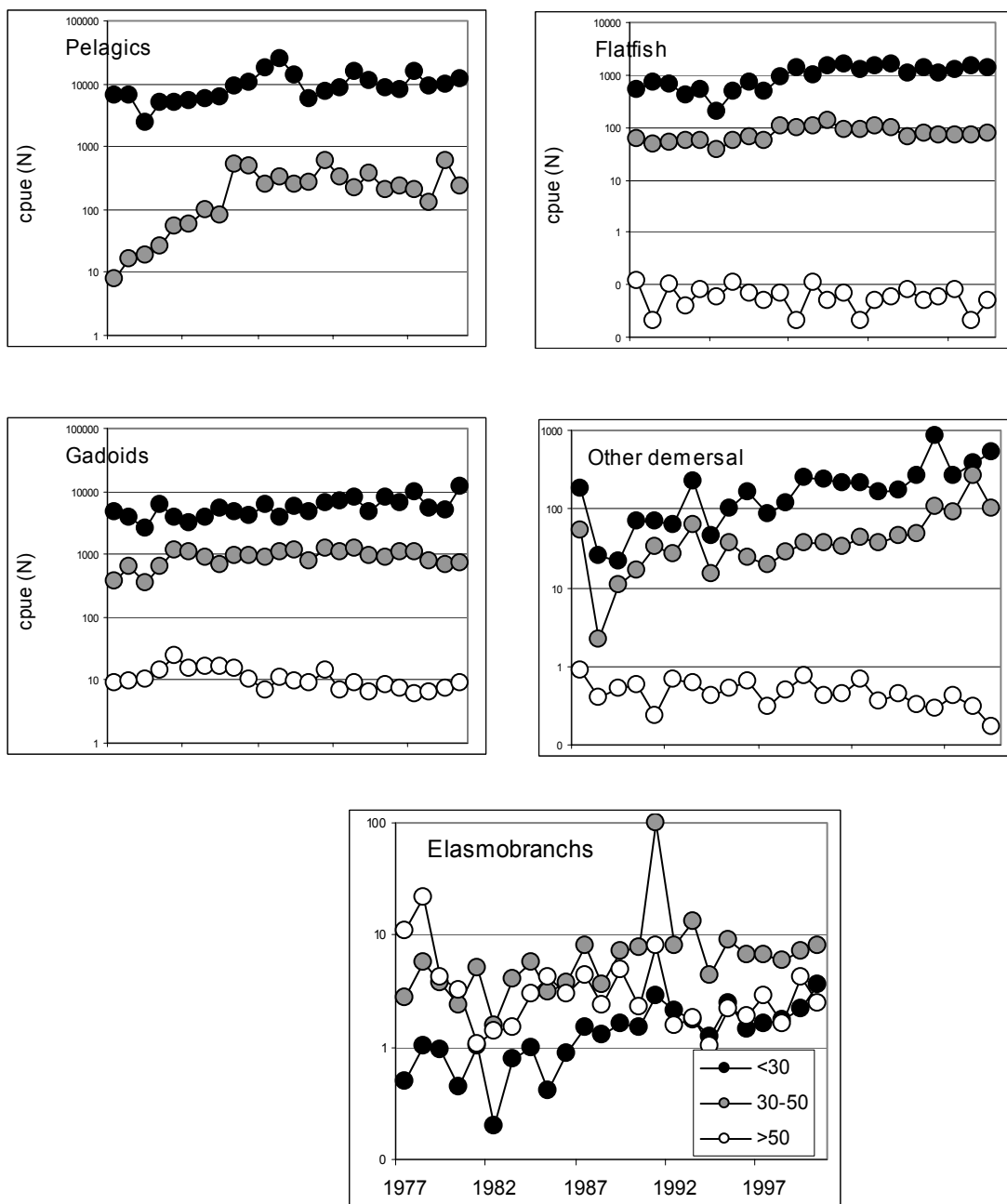


Figure 2.3.1.1.4. Time series of cpue in N per hr fishing by length class (<30; 30-50; and >50 cm) for major groups of species.

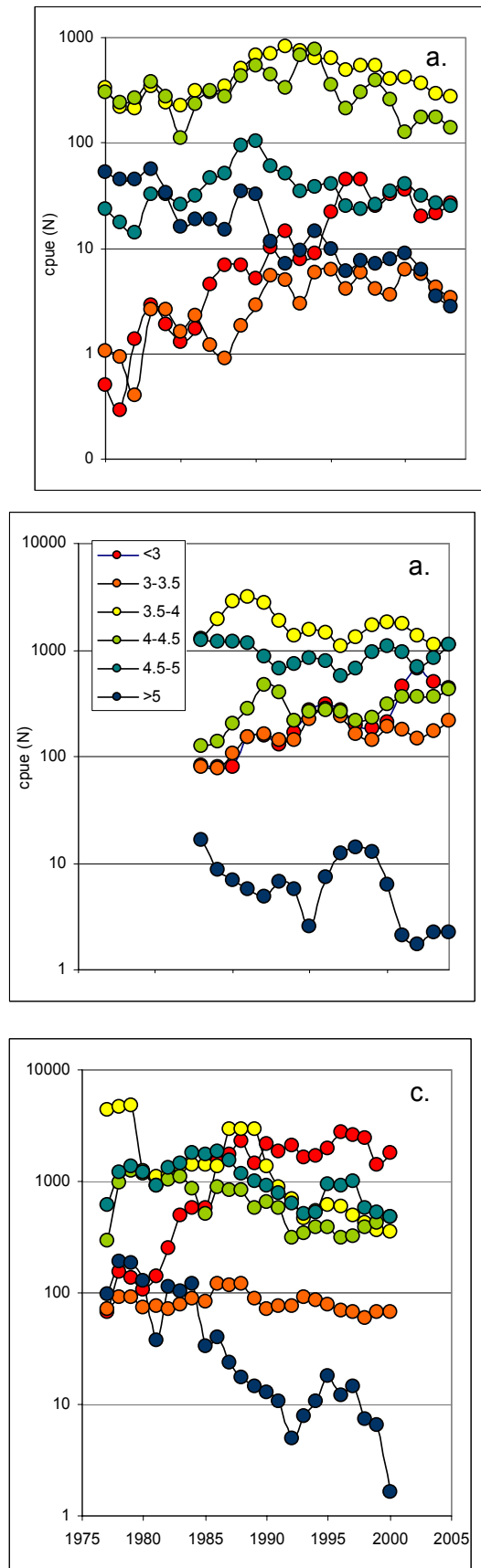


Figure 2.3.1.1.5. Cpu (N) by ln maximum length class (common demersal species only) in (a) IBTS, (b) BTS and (c) DYFS for roundfish area 6.

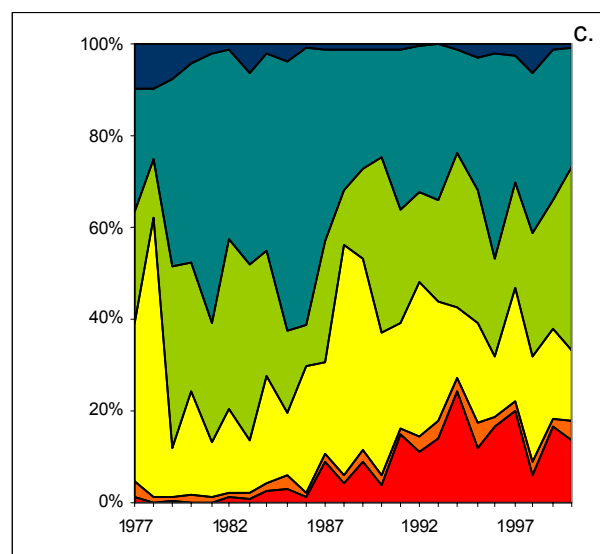
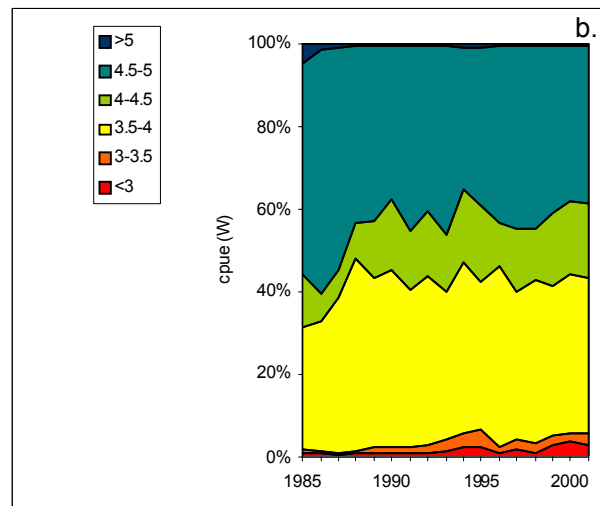
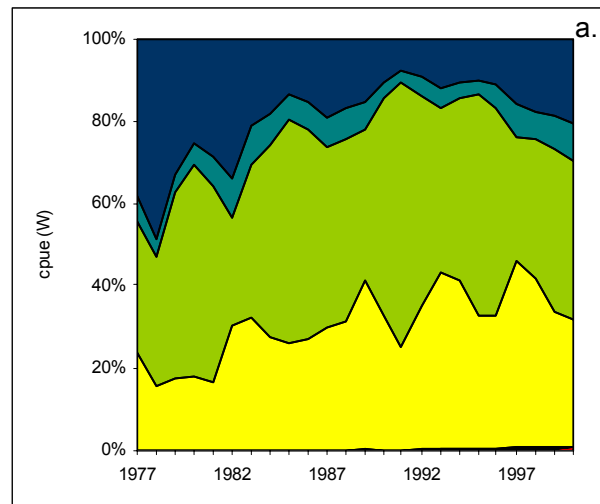


Figure 2.3.1.1.6. Percentage composition of cpue in W by ln maximum length class (common demersal species only) in (a) IBTS, (b) BTS and (c) DYFS for roundfish area 6.

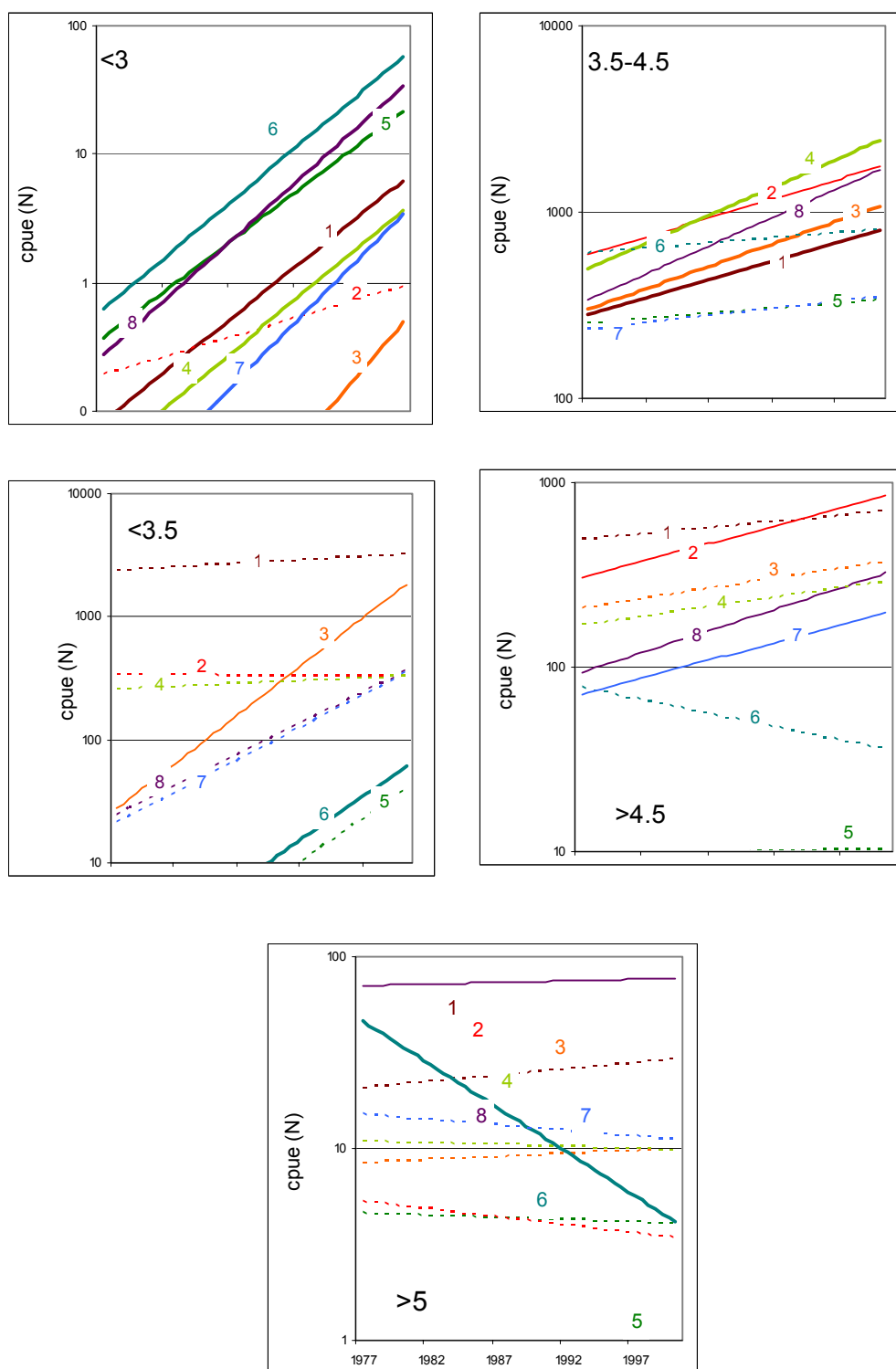


Figure 2.3.1.1.7. Temporal trends (heavy lines: $p < 0.01$; thin lines: $p < 0.05$; dashed lines: not significant) in cpue in N per hr fishing by ln maximum length class (a: < 3 ; b: < 3.5 ; c: $3.5-4.5$; d: > 4.5 ; and e: > 5) and roundfish area (1-8).

2.3.1.3 Discussion

While declines in proportion of average weight and average maximum length as observed by ICES (2001, 2002b) were confirmed in all three survey data sets, a detailed analysis of absolute rather than relative cpue showed that a widespread, long-term increase has occurred in both the abundance of small fish and of small species (low maximum length). In addition, absolute abundance of large fish has declined significantly, but the reduction in large species (high maximum length) was not significant, with the exception of one particular area (south-eastern North Sea).

The reduced abundance of large fish is undoubtedly caused by the observed increase in fishing mortality over the same period in many of the routinely assessed species. However, it does seem likely that the increase in absolute abundance of small fish is also induced by fishing. MSVPA has shown that a reduction in large predators owing to an increase in exploitation rate releases predation pressure on the juveniles of other species, leading to higher recruitment. This explanation as an indirect of fishing apparently fits theoretical expectations, whereas as other explanations such as environmental change might cause all kind of changes, but there is no theory why exactly this pattern would emerge.

The increased abundance of small species can be equally well explained by indirect effects of fishing, because releasing predation pressure on them should lead to increased abundance. The absence of a significant decline in large species was somewhat unexpected. Maximum length was assumed to be a proxy of natural mortality, and therefore species with a low natural mortality were expected to be more sensitive to the same level of exploitation than species with a high natural mortality. In practice, this does not seem presently the case. The exceptional significant decline in the south-eastern North Sea appeared to be entirely related to the collapse of a single species (cod), which can hardly be called a community effect. Still, it is quite possible that a clear signal is lacking because some stocks of the larger species (e.g. skates) had collapsed before the surveys started.

The two metrics proposed clearly integrate direct and indirect effects of fishing, but as such do not provide information on which of the two effects (or both) are responsible. In terms of effects of fishing on the fish community the reduced abundance of large fish means simply the accumulated direct effects of fishing on all species. From an ecological perspective, the compensatory response in small fish and small species seems of more interest, because an overrepresentation of these categories in respect of absolute abundance might well signify a more important aspect of reduced ecological quality than is represented by the average maximum length.

While similar trends were reflected consistently in the different surveys, the actual levels of the metrics varied considerably, even if the survey area was strictly comparable. This is not surprising because it is inherent to the use of specific survey gear. Each gear samples a specific assemblage within the total fish community present, with the bias dependent on the relative catchabilities for individual species. While the absolute bias is unknown, the relative bias among different gears might be evaluated, but this would be a major exercise. For the time being, we have to accept that different surveys reveal different patterns and the choice for a particular survey as the basis for an EcoQO for a broader sea area would be completely arbitrary and involve a specific bias.

Although broad temporal trends in exploitation rate matched those in various metrics, we are not well equipped to estimate their causal relations. First of all, exploitation rate at the community level is ill defined compared to fishing mortality at the single-species level. The best measure would probably be the ratio of total biomass removed per year by all fisheries (including discards) to average biomass present. Regrettably, these parameters are not available. Jennings *et al.* (1999) provide international hours fishing for North Sea otter and beam trawl fleets, but data for a few countries were missing as well as data for other gears. While these specific gears would be appropriate to estimate impacts on benthic communities, the fish community exploitation rate must reflect all gears.

2.3.2 Celtic Sea

Further analyses were undertaken to investigate changes in the size structure of the Celtic Sea fish community. The aim was to determine whether the temporal trends would reveal similar results using a community size-spectra approach as the results shown for the proportion of large fish, average weight and changes in L_{\max} presented in section 2.2.5.

2.3.2.1 Methods

The annual index of abundance was calculated as the number of fish caught per hour standardised by the number of stations sampled in each year of the Celtic Sea groundfish survey (1987-2002). The fish community size spectrum was estimated annually as the natural logarithm of fish abundance for each 5 cm length class. Linear regressions of $\ln(\text{abundance})$ on the midpoint of the length classes were performed for each year in the series. The midpoints of the length classes were rescaled to correct for correlation between slopes and intercepts by ensuring the data along the x-axis were centred on zero. The corrected intercept is henceforth referred to as the 'height'. To enable comparison, all analyses were carried out on the fish community as a whole and then for the following separate categories: pelagic teleosts, demersal teleosts, elasmobranchs and boarfish.

2.3.2.2 Results and Discussion

The abundance of size classes (by 10 cm) for all fish in the community (irrespective of category) show two opposite trends in the data series over 1987-2002 (Figure 2.3.2.2.1). There was a notable increase of smaller fish over time (<10 cm, $R^2=0.26$, $p=0.04$). An overall decline in larger fish was also apparent, though only significant for very large fish above 90cm (R^2 , 0.25, $p=0.05$). There is a noticeable residual pattern in the time series suggesting autocorrelation or environmental signalling. Therefore, relying on the significance tests from simple least squares regression analyses to determine whether there is a significant linear trend may be somewhat biased and the significance levels should be interpreted bearing this in mind. In general, the aim here is to discuss the general apparent trends.

When the abundance time series was separated into the different categories (pelagic teleosts, demersal teleosts, elasmobranchs and boarfish) different results were apparent for the respective categories. There were no significant linear trends for pelagic teleosts (Figure 2.3.2.2.2.a) although two peaks in the series occurred in 1991 and 1997 for larger pelagic fish (Figure 2.3.2.2.2.b). An even more pronounced pattern exhibiting peaks in 1992, 1995 and 1998 was evident for boarfish, considered simply as one size class due to their limited size range. Underlying these fluctuations is a significant positive trend in boarfish over time ($R^2=0.29$, $p=0.03$). There were no significant linear trends in the abundance of demersal fish when considered in different size classes over time. However there does appear to be a slight increase in the smallest size class and very slight declining trend in larger length classes (<60 cm) (Figure 2.3.2.2.3.). Intermediate size classes of elasmobranchs have declined over the time series (41-50cm and 51-60cm, $p=0.04$ and 0.03, Figure 2.3.2.2.4.). However, largest size classes were constant over the period though the levels of abundance of extremely large elasmobranchs (> 100 cm) is comparatively low throughout the time series considered.

The slopes and adjusted intercepts ('heights') from annual size spectra are shown in Figure 2.3.2.2.5. and were fitted with a localized smoothing function (loess, using Splus 6.1). The slopes and intercepts for 'all' species exhibited an overall significant decline over 1987-2002 in both (ANCOVA, $p=0.04$, Figure 2.3.2.2.5.). Explicit year-to-year differences were not significant at the 5% level.

The annual size spectra constructed for the separate categories (excluding boarfish) revealed somewhat different results (Figure 2.3.2.2.5.). In particular, there was a characteristic, though highly variable, 'w' pattern in the slopes of the pelagic fish over time. The slopes for demersal teleosts demonstrated an overall decline (steeper negative slopes). The slopes of the spectra for elasmobranchs were relatively constant although highly variable possibly due to the absence of particular size classes in certain years of the survey data. The intercepts or 'heights' of size spectra of all species, pelagics and demersal fish categories did not exhibit any particular trend (Figure 2.3.2.2.5.). However, a highly variable declining trend was somewhat apparent for the 'heights' of the demersal teleosts and elasmobranch size spectra.

The increasingly negative slope of the fish community as a whole is indicative of the decline in the size structure of community and this is consistent with the indicators presented in section 2.2.5. When size spectra are constructed for separate fish categories the patterns are less clear and highly variable but also consistent with results in some of the other analyses presented. For example, the fluctuating pattern of the slopes of the pelagic fish size spectra is notable. As previously discussed this maybe heavily influenced by the NAO (see section 2.2.7.).

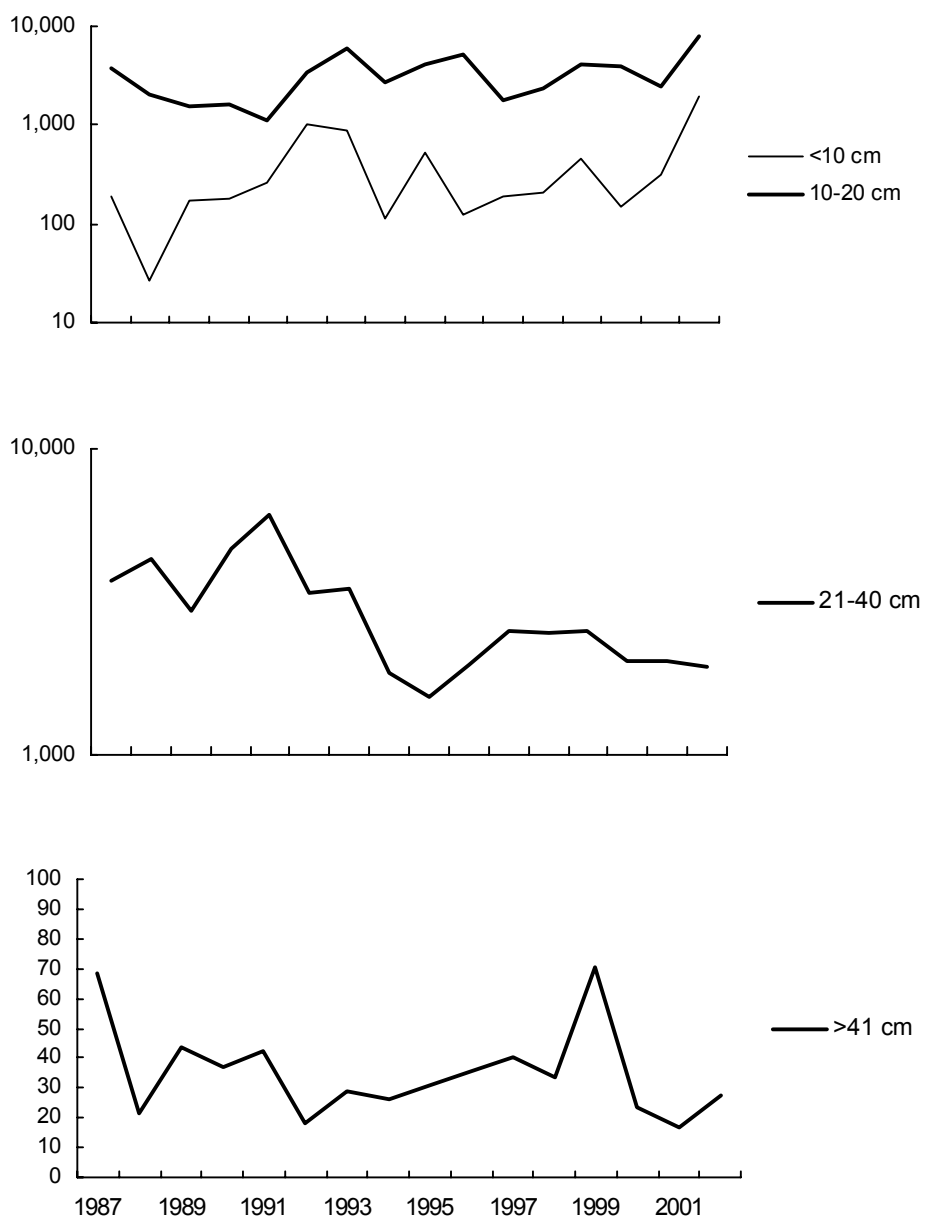


Figure 2.3.2.2.1. Trends in the abundance of all fishes separated by 10 cm size class in the Celtic Sea between 1987-2002

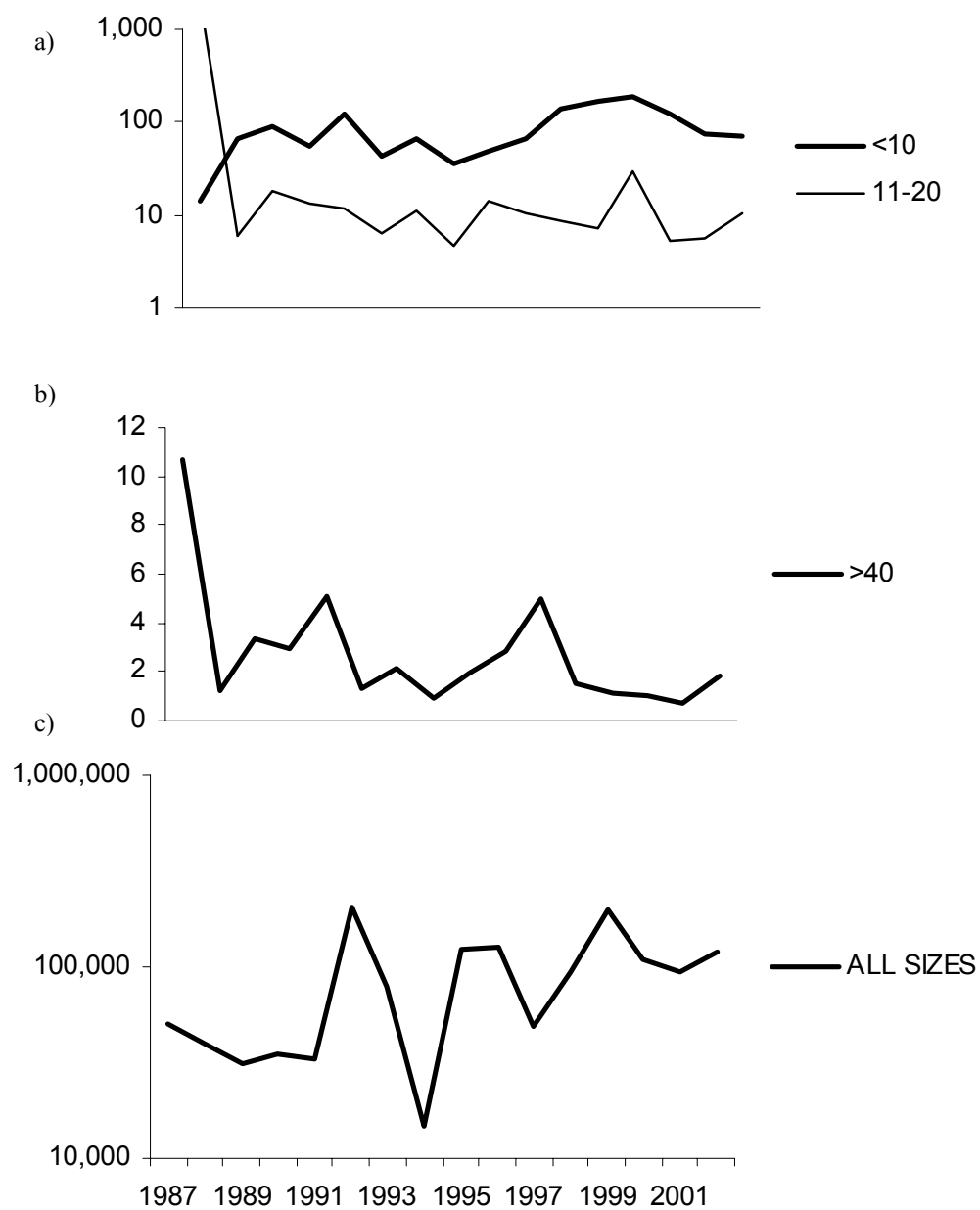


Figure 2.3.2.2.2: Trends in the log abundance of specific size classes of pelagic fish (a and b) and boarfish (c) in the Celtic Sea over 1987-2002.

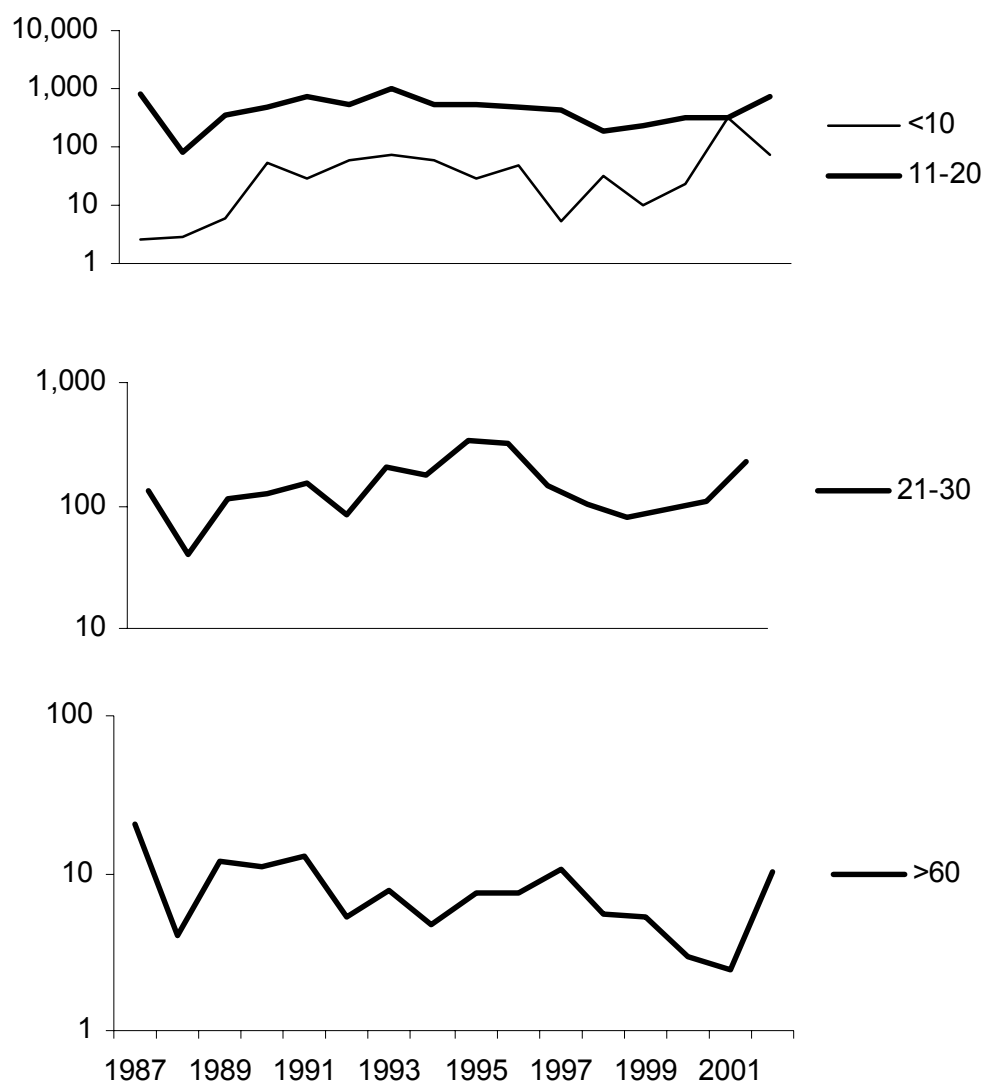


Figure 2.3.2.2.3: Trends in the log abundance of separate size classes of demersal fish in the Celtic Sea over 1987-2001.

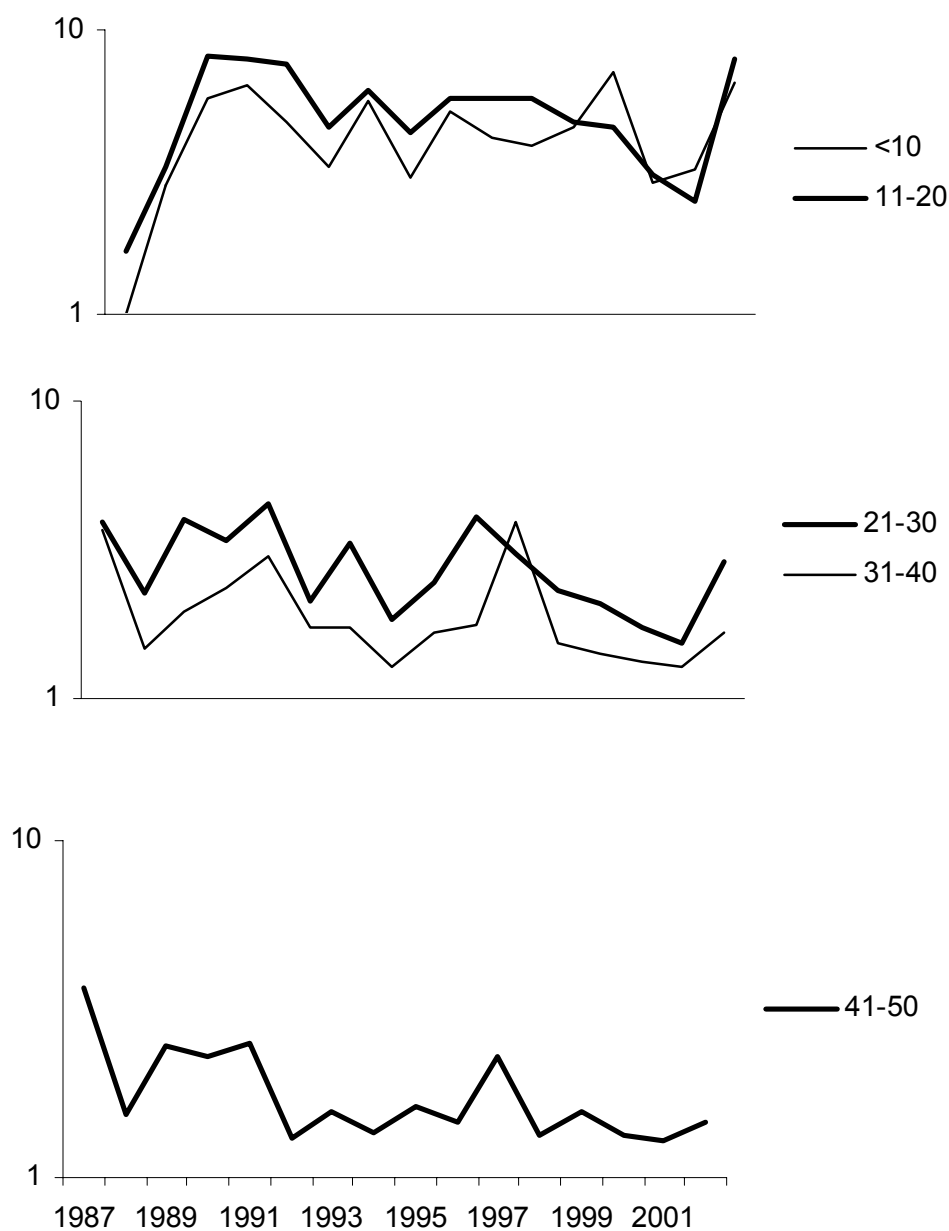
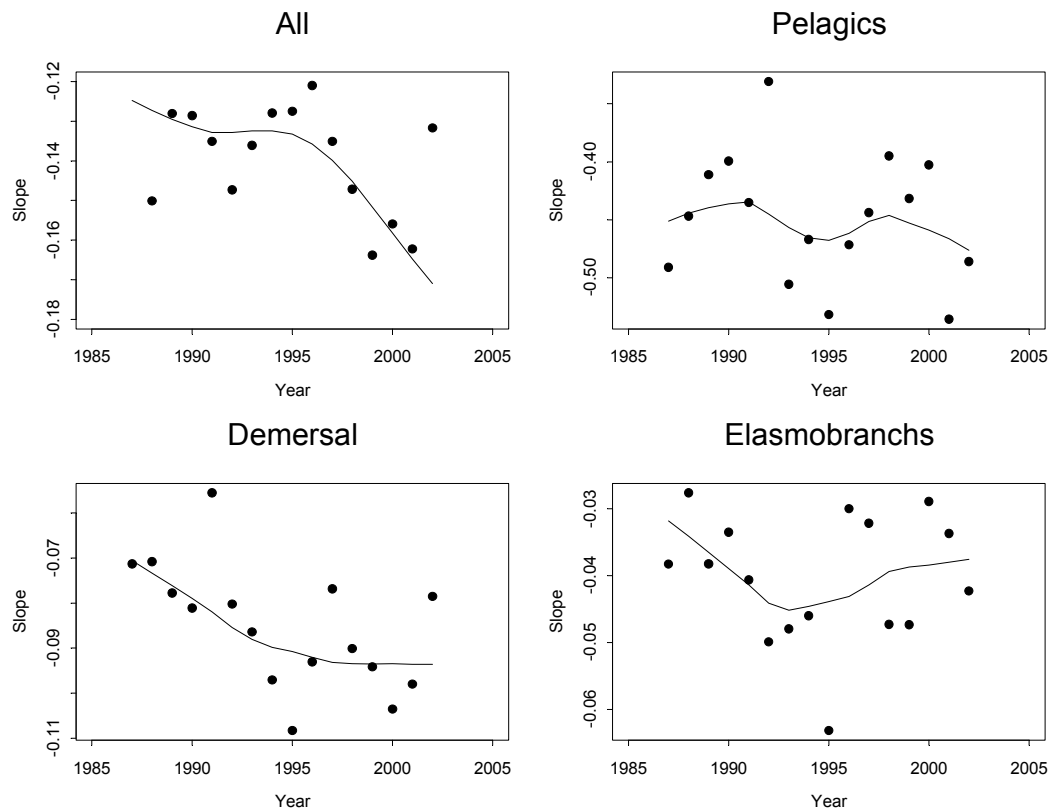


Figure 2.3.2.2.4: Trends in the log abundance of separate size groups of elasmobranchs in the Celtic Sea over 1987-2002.

A.



B.

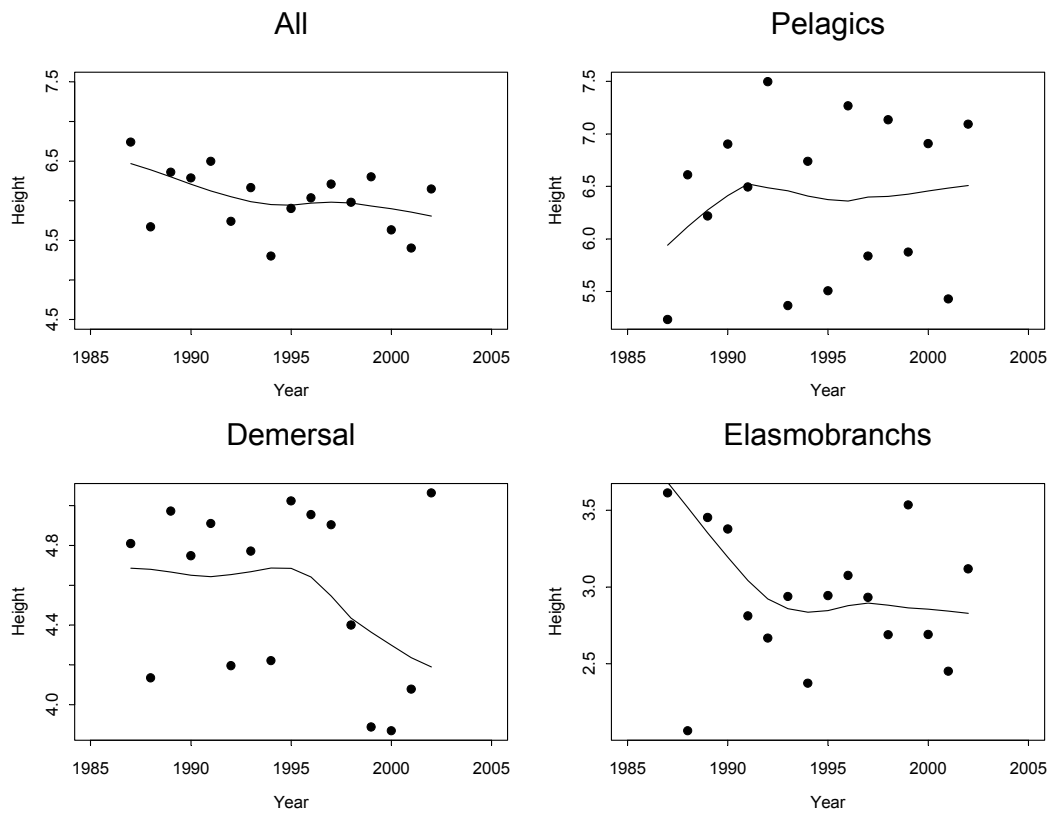


Figure 2.3.2.2.5: Slopes (A.) and intercepts (B.) of size spectra over 1987-2002 for Celtic Sea fish assemblages fitted with a LOWESS smooth function.

2.4 Identification of the levels of the metrics that can be taken as indicative of various states of the marine ecosystem

2.4.1 A case study for the North Western North Sea

The analysis carried out in section 2.2.2. of long-term trends in the three assemblage size composition metrics in areas of varying fishing disturbance, provides a possible mechanism with which to set ecological quality reference points and target levels with respect to these parameters. For each metric the fitted regression lines intercept within the time span of the time series. Thus the earliest intersections with the low effort trend line were in 1938 with respect to the percentage of large fish, 1926 with respect to average weight and 1922 with respect to Length_{Infinity} (Figure 2.4.1.1). This suggests that the assumption, that demersal fish assemblages present in rectangles subjected to varying levels of otter trawling have diverged from the same start point (see section 2.2.2.), does indeed hold true. If so then the points of interception of the regression lines provide an indication as to when the perturbed assemblages diverged from the “pristine” state. They also therefore provide an indication as to what the ecological quality reference points for each size composition metric should be.

In each of the long-term trend analyses, the regression lines fitted to the low effort treatment data were not statistically significant (Figures 2.2.2.4.2. to 2.2.2.4.4.). Being unable to disprove the null-hypothesis, a slope of zero over time should be assumed. Under these circumstances reference points for each size composition metric are the mean values of the low effort data for each metric; ie 10.22% for the percentage of large fish in the assemblage, an average weight of fish in the assemblage of 124.2g and an average Length_{Infinity} of fish in the assemblage of 49.3cm (Figure 2.4.1.2.). In rectangles of high otter trawl effort, the demersal fish assemblage departed from these values around 1948 with respect to the percentage of large fish, around 1949 with respect to average fish weight, and around 1931 with respect to Length_{Infinity}. Figure 2.4.1.2. shows the 95% confidence around the low effort mean level. These indicate that confidence limits (between 15 and 20 years) around these point-of-disturbance dates are considerable. Data falling below the lower 95% confidence limit might be considered to represent a perturbed assemblage as indicated by the particular metric being applied to the groundfish survey data. With respect to the fish assemblage in rectangles of high fishing effort, the choice of size composition metric makes little difference to the conclusions drawn. Since 1960, the assemblage size composition has been below the lower 95% confidence limit of the proposed reference point between 82% and 91% of the time (Figure 2.4.1.3.).

Although the overall trends for the low effort rectangles shown in Figure 2.4.1.1. were not statistically significant, they serve to highlight an important precautionary note. The overall trends of the size composition metrics in the low effort rectangles may actually reflect a response to some other causal factor. For example, in this case, changing environmental conditions, such as increasing water temperature, may have favoured smaller fish and species with a lower Length_{Infinity} in recent decades. Under these circumstances setting ecological quality reference points at the level where the high effort regression lines intercept the mean low effort metric values (Figure 2.4.1.2.) may be unrealistic and not take account of current environmental conditions. In this situation, a more realistic approach might be to set reference points at contemporary values of the regression lines for low fishing effort, ie 10.4% for the percentage of large fish in the assemblage, an average weight of fish in the assemblage of 107.3g and an average Length_{Infinity} of fish in the assemblage of 47.3cm. Given the extent of the confidence limits at this point in the relationships (Figures 2.2.2.4.2. to 2.2.2.4.4.) and the fact that this is extrapolating beyond the time-span data, there is little point in assigning confidence limits to these values. However, it is worth noting that in all cases, these values fall inside the 95% confidence limits around the potential reference points proposed in the previous paragraph.

The analyses presented here must be viewed only as a case study for illustration purposes. Some further, and important, precautionary points regarding this approach to assessing potential ecological quality reference points need stating.

- This method is gear specific. These are reference points set for the demersal fish assemblage as sampled by the Aberdeen 48ft Trawl. Had the assemblage been sampled using a GOV or Beam trawl, the reference points would almost certainly have been very different.
- This method is area specific. These data were all collected from 75 ICES statistical rectangles in the north-western North Sea. Had the data been collected in the southern North Sea, or in another marine area altogether, then the reference point values almost certainly have differed.
- This method is guild specific. The data refer only to those species deemed to be sampled adequately by the Aberdeen 48ft Trawl. Thus many species, such as sandeels, herring and sprats were excluded from the analysis. Had these species been included, then different reference point values would have been obtained.

- The method relies on the availability of adequate data. The divergence points between the high and low effort fish assemblages occurred between 1931 and 1949 depending on the size composition metric used. Few groundfish survey data sets extend back this far in time. Even fewer effort data are available from this period to test some of the assumptions underpinning this method.

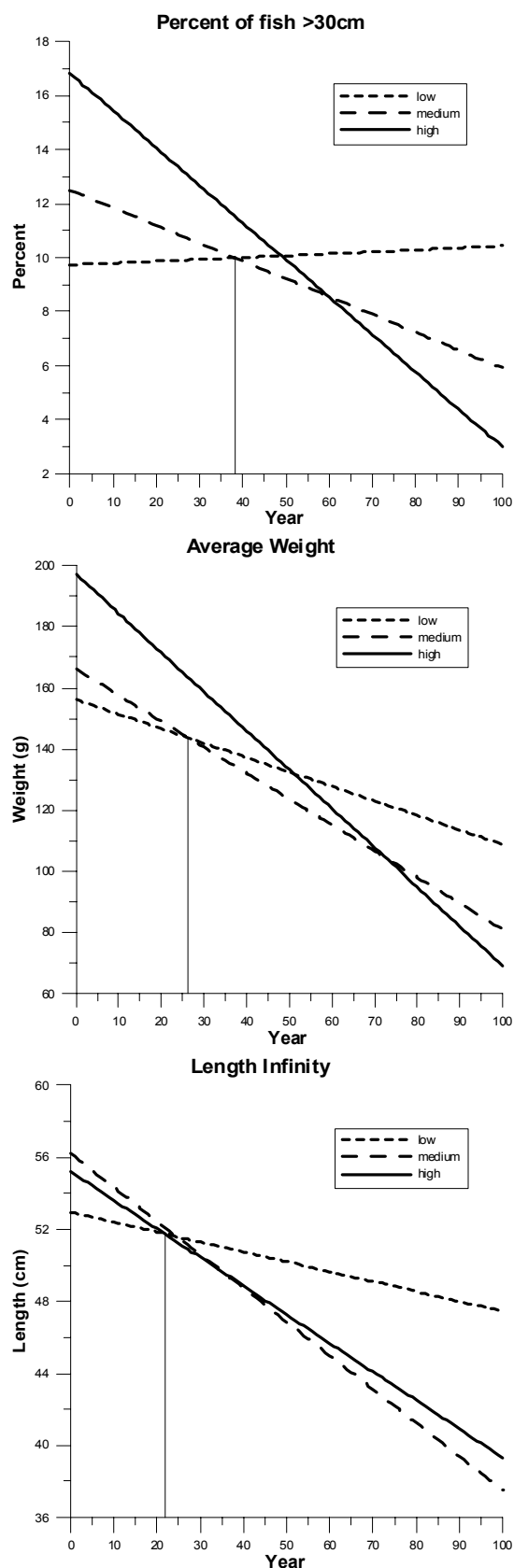


Figure 2.4.1.1. Overall long-term linear regression trends for three demersal fish assemblages under three fishing effort treatments. Earliest departure points from the “pristine” state and potential ecological quality reference points are indicated for each metric.

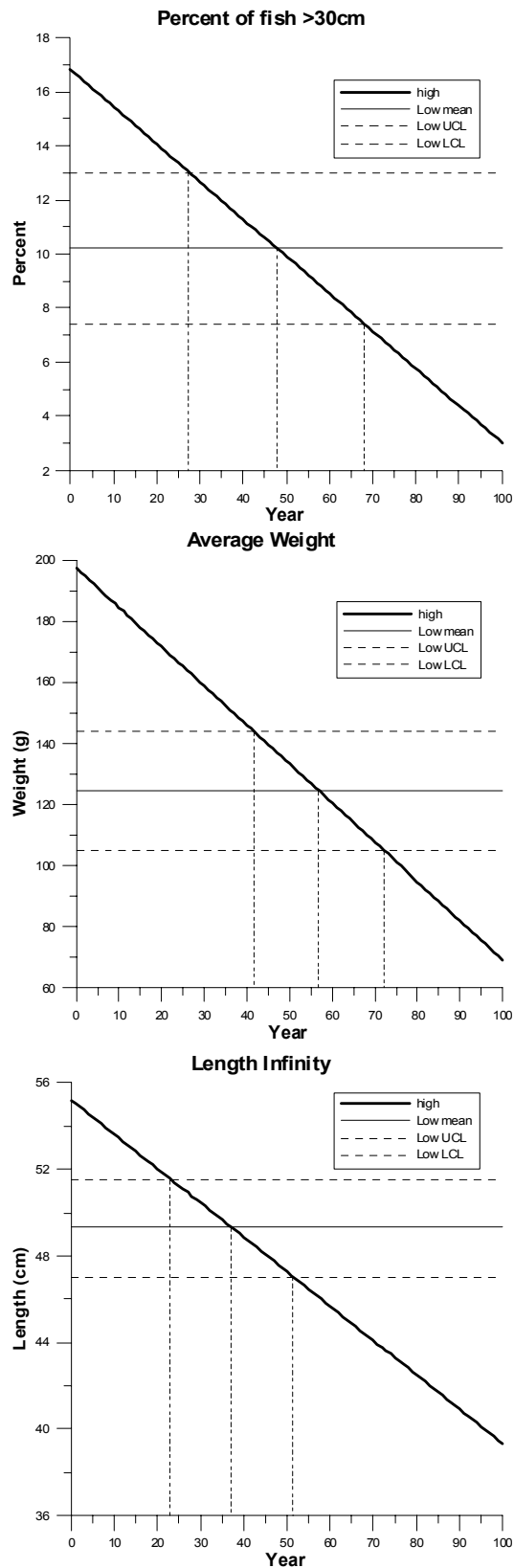


Figure 2.4.1.2. Overall long-term linear regression trends for the three demersal fish assemblage size composition metrics in rectangles of high fishing effort. The mean values (with 95% confidence limits) for each metric under a low fishing effort regime are indicated. Departure points from the “pristine” (= low effort) state and potential ecological quality reference points are indicated for each metric.

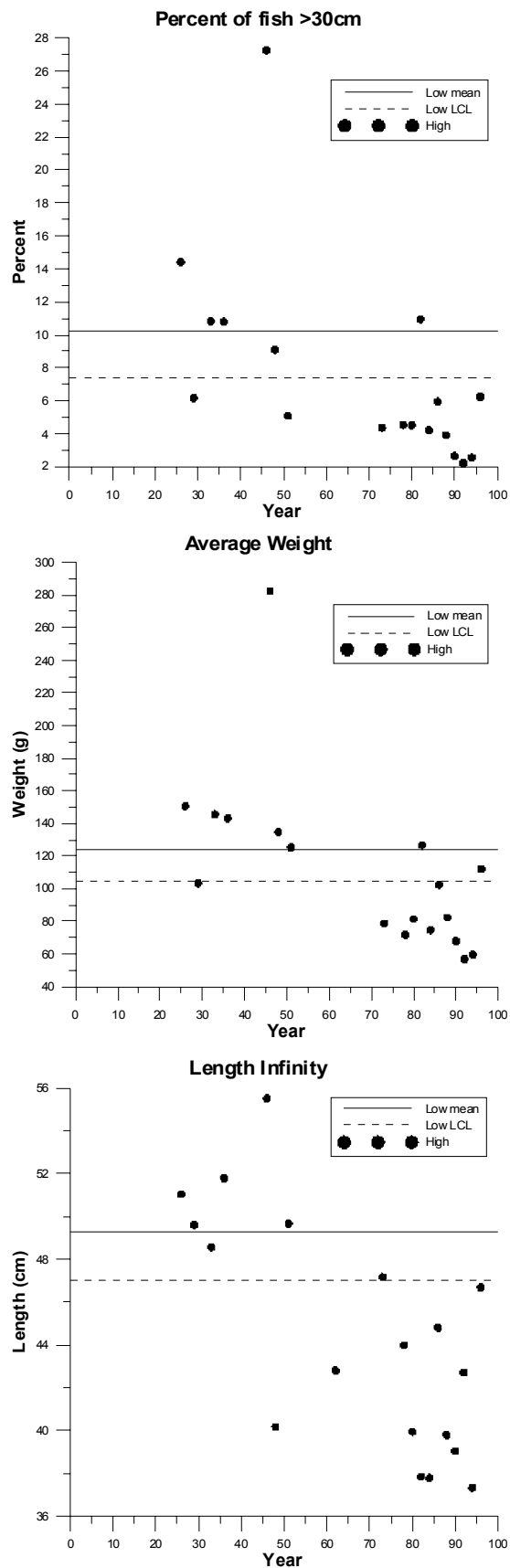


Figure 2.4.1.3. Long-term variation in the value of each metric calculated for the fish assemblage on rectangles of high fishing effort. The suggested reference point (mean of the data calculated of the entire time-series for rectangles of low fishing effort) and the lower 95% confidence limit are shown.

2.4.2 Discussion

There is no theory that could predict what kind of average weight or average maximum length might be obtained in a specific survey for a specific reduction in exploitation rate of the fish community, let alone what kind of values might be expected in a non-exploited system. The only relevant information is the empirical relationship between any metric and available estimates of community exploitation during the period a survey has been carried out systematically. Even if the correlation is statistically significant, the relationship may reflect delayed responses of the fish community, because community metrics integrate effects over several years of change in exploitation superimposed on annual (random) variations in recruitment to all species in the assemblage sampled in the survey gear. For these reasons, the predictive value of any empirical relationship is very limited, while extrapolations outside the observed range of values are not warranted. Thus any sensible reference level should be within the observed range. Given that none of the available surveys extends into periods when communities can be considered as unexploited, the reference level could only indicate the state of an exploited ecosystem and therefore, should be used as a limit reference level.

Suppose that these two metrics were selected to formulate EcoQOs and that the reference level was chosen to correspond to the historic period of minimum exploitation in the survey time series, what management advice might be given to ensure that these objectives might be reached? There are various answers to this question:

- An overall reduction in exploitation in all fisheries should ultimately lead to some improvement, but probably with a delay of several years, because that amount of time is required for a small fish of a large species to become a large predator and exert its predation pressure on small fish.
- Protect those large fish that are still around by closing fisheries that target relatively large fish (e.g. gillnets) in particular areas favoured by large fish (e.g. wrecks) or where spawning aggregations might occur. While this would seem effective in the short term, in the long term this is probably less effective than overall effort reductions that enhance the chance of reaching age at maturity.
- Reduce minimum mesh size. This would increase fishing mortality on small fish and thus compensate for reduced predation by large fish. Thus, it might be expected that a more evenly distributed fishing mortality on size groups would actually increase the average weight in the survey and possibly also the average maximum length.
- As a variation of the former, increase effort in industrial fisheries.

This 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 ones, because advice to increase exploitation to meet quality objectives is probably not intended by the framework.

Overall, WGFE feels that as yet it is neither appropriate to implement these fish community metrics as part of an EcoQO, nor to define a global North Sea reference level for management.

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3 EVALUATION OF THE QUALITY AND SUITABILITY OF DATA FOR THE LISTING OF THREATENED AND DECLINING FISH SPECIES BY OSPAR

3.1 Introduction

The OSPAR Biodiversity Committee are nominating a variety of marine fish and invertebrate species that are considered threatened or declining and ICES was asked to contribute to the peer-review process for these nominations. Whilst the better-known species were evaluated by other ICES working groups (e.g. ICES, 2002a), ICES was not able to formulate advice for seven fish species (ICES, 2002b). Hence, 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 oxyrinchus*)
- Short-snouted seahorse (*Hippocampus hippocampus*)
- Seahorse (*Hippocampus guttulatus*)
- Couch's goby (*Gobius couchi*)

The Texel-Faial criteria for identifying species in need of protection are given in Table 3.1.1, and the criteria identified for the case species summarised in Table 3.1.2. The OSPAR areas are I-Arctic, II-North Sea, III-Celtic Seas, IV, Bay of Biscay and Iberian waters and 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 3.1.3).

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 favorable 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 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 co-operation of several States, and to promote such co-operation. Particular emphasis is given to endangered and vulnerable species, including endangered and vulnerable migratory species.

CITES: CITES (the Convention on International Trade in Endangered Species of Wild Fauna and Flora) is an international agreement between Governments. Its aim is to ensure that international trade in listed species of wild animals and plants does not threaten the survival of the population.

Table 3.1.1: 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 3.1.2: Potential threatened fish species, indicating the OSPAR areas that were suggested and the rationale supporting the original nomination.

	Area	Global importance	Local Importance	Rarity	Sensitivity	Keystone species	Decline
Sea lamprey (<i>Petromyzon marinus</i>)	I,II,III,IV			✓	✓		✓
Sturgeon (<i>Acipenser sturio</i>)	II IV	✓			✓		✓
Allis shad (<i>Alosa alosa</i>)	II,III,IV			✓			✓
Houting (<i>Coregonus lavaretus oxyrinchus</i>)	II			✓			✓
Short-snouted seahorse (<i>Hippocampus hippocampus</i>)	II,III,IV,V		✓		✓		✓
Seahorse (<i>Hippocampus ramulosus</i>)	II,III,IV,V		✓		✓		✓
Couch's goby (<i>Gobius couchi</i>)	All	✓		✓	✓		✓

Table 3.1.3: Conservation categories for the nominated species

	Habitats Directive	CITES	Bern Convention	IUCN
<i>Acipenser sturio</i> , Common Sturgeon	Annex II & IV	Appendix I	Annex II	classified as Critically Endangered
<i>Alosa alosa</i> , Allis Shad	Annexes II & V		Annex III	
<i>Coregonus lavaretus oxyrinchus</i> , Houting	Annexes II & V		Annex III	
<i>Gobius couchi</i> , Couch's Goby				
<i>Hippocampus hippocampus</i> , Short snouted Seahorse		Appendix II		classified as Vulnerable
<i>Hippocampus guttulatus</i> , Long snouted Seahorse		Appendix II		classified as Vulnerable

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3.2 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 midwater 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 3.2.1) the main fisheries for sea lamprey are 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 caution and are likely to be a significant underestimation, as catches in the Gironde system alone have been between 40 and 155 tonnes per year (Girardin *et al.*, 2002), and the CPUE index of abundance suggests an increase in the size of the population (Figure 3.2.2).

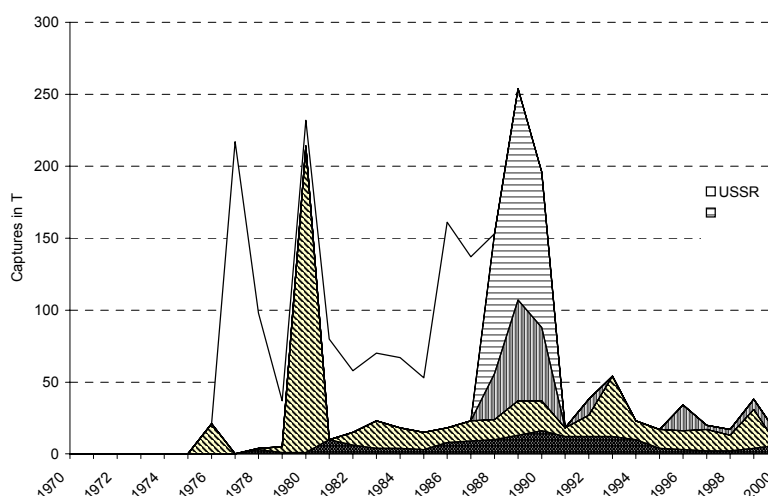


Figure 3.2.1: Reported total catch in tonnes of sea lamprey in OSPAR maritime area (<http://www.fishbase.org/report/FAO/FAOCatchList.cfm?scientific=Petromyzon%20marinus>)

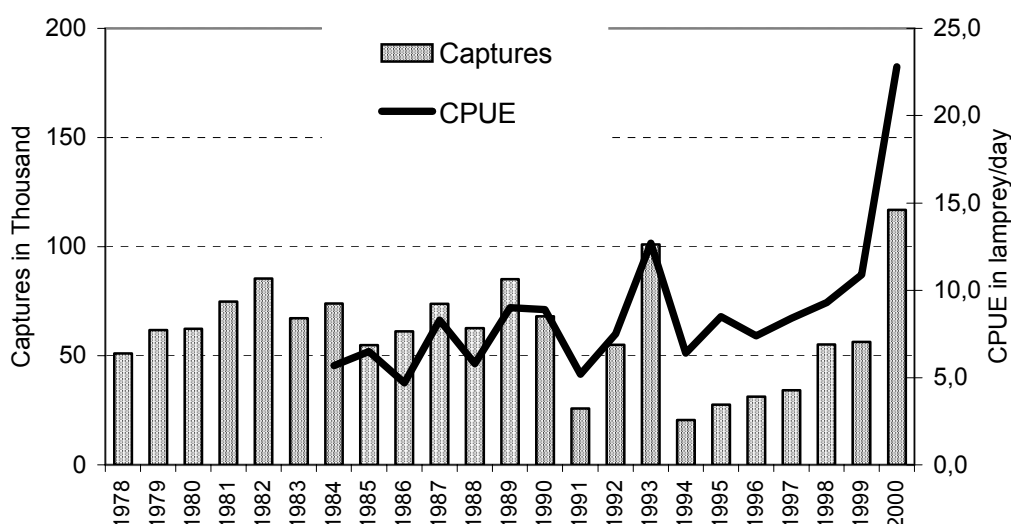


Figure 3.2.2: Catch and CPUE for sea lamprey in the Gironde system between 1978 and 2000. Captures in thousand of fish and CPUE in fish/trammel net/fisherman (From Girardin *et al.*, 2002).

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. com.)

In Finland reports from coastal water lamprey 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 UK (Brown *et al.*, 1997) they have been reported to be in decline. In the British Isles it 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 ones due to pollution and engineering barriers (Maitland 1980a, 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 part of the country with the River Minho being the major fishery (Almeida pers. comm). There have been numerous reports of a 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; Almeida *et al.*, 2002), 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 3.2.3) (Almeida *et al.*, 2002). Alteration of the discharge pattern associated with dams has also been implicated, 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.*, 2002), while at low flows the cues and stimulus to migrate may be reduced (Almeida *et al.* 2000).

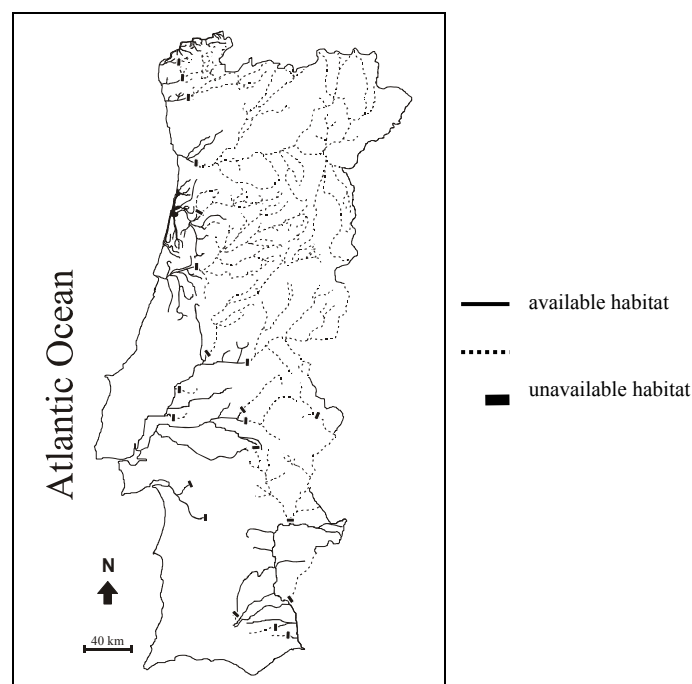


Figure 3.2.3: Habitat available to sea lamprey populations in Portuguese river basins where the species is known to occur (Almeida *et al.* 2002).

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 dependent on life stage (size); ranging from silt-sand for fish of 20-60 mm to coarse grained sediment for those between 140 and 200mm. As a result channelisation, 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 lamprey (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.) larva 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 birds (e.g. herons). There are a number of records of birds and mammals attacking adult sea lamprey, especially at spawning time, but it 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% of mortality could happen at this moment. 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 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.* (2002) mentions that stocks are over fished 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. com). Adult sea lamprey, being long and thin fish are vulnerable to passage through pumps.

Conclusion in relation to 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, erection of dams and dredging of the channel (Ducasse and Leprince, 1980) and the blocking off 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, 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 3.2.2)

Because of 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 was 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.

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3.3 Common Sturgeon (*Acipenser sturio*)

Status and distribution

The marine geographical distribution of *Acipenser sturio* in the OSPAR area is now restricted from south Bay of Biscay to North sea including the British Isles (Castelnaud, 1988; Rochard *et al.*, 1990; Lepage and Rochard, 1995; Rochard *et al.*, 1997) (Figure 3.3.1). The last recorded information about the former population of the Rioni river, a tributary of the Black Sea in Georgia, was more than 20 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) we should consider that *A. sturio* is now extinct in Rioni. The status of *A. sturio* in the Danube is uncertain.

The common sturgeon reproduces only in the Garonne and Dordogne River in south west France (Castelnaud *et al.*, 1991 ; Rochard, 1992 ; Williot *et al.*, 1997) and juveniles stay several years in the Gironde estuary on feeding zones and this region is of particular importance for the species (Rochard *et al.*, 2001).



Figure 3.3.1: 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 homogenous in abundance.

Sturgeon were common and widespread in NE European waters in the 8-11th centuries, however they have 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, whereupon exploitation and the renewed damming of rivers initiated further widespread declines (Hoffmann 1996). The common sturgeon was historically present in most large west European rivers (Magnin, 1959; Hoffmann, 1996) from the Black Sea (Ninua, 1976) to Baltic Sea including Iberian peninsula (Classen, 1944, Almaca 1988; Almaca and Elvira, 2000) and British isles. Populations of 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 Barents sea were *A. sturio*, it could also have been *Acipenser oxyrinchus*.
- the historical presence of *A. sturio* in the Danube is also questionable (I. Navodaru, Danube Delta Institute pers. com.).
- 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, b; Garrido Ramos *et al.*, 1997).

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.* 1991 a and b) became extinct in the 1950's. However, as it is a long-lived species, vagrant large individual 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 3.3.2) (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 et IV). It is also listed in the IUCN red list of threatened animals as critically endangered.

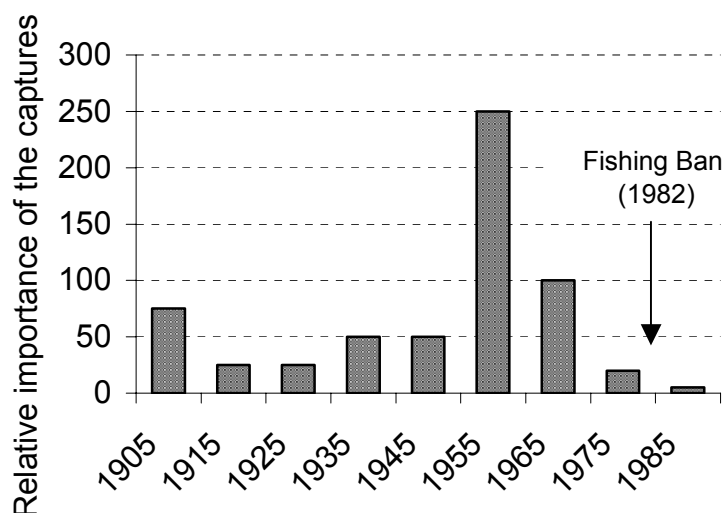


Figure 3.3.2: Relative importance of the captures of common sturgeon in the Gironde basin (from Rochard, 2002)

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 the fishermen show a drastic decline (Figure 3.3.3)(Williot *et al.*, 1997 ; 2002 ; Rochard. coord.2002).

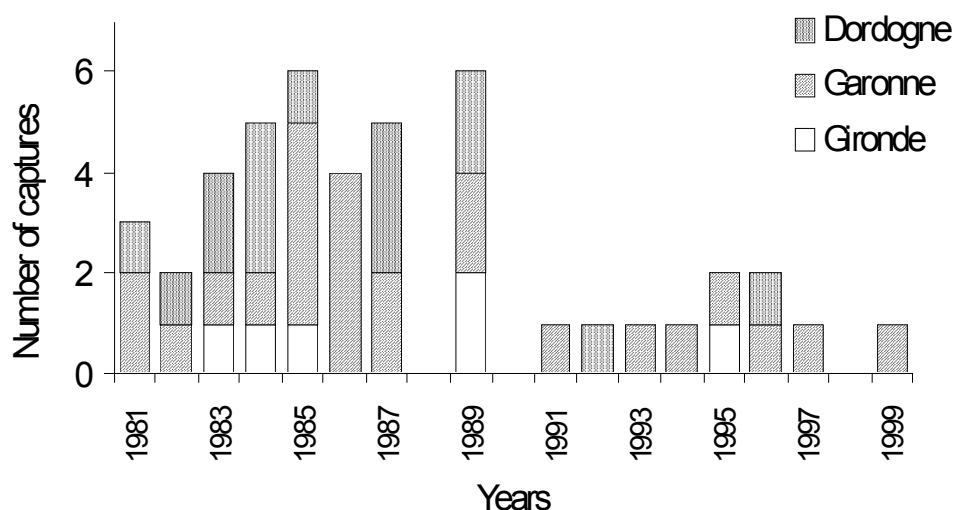


Figure 3.3.3: Number of incidental captures of adult common sturgeon in the Gironde system (from Rochard, 2002)

From the monitoring of the population in the estuary it has been concluded that reproduction does not occur annually (Figure 3.3.4, Rochard *et al.*, 2001). The last natural reproduction occur in 1994 (Elie, 1997). In 1995, 9000 artificially reared juvenile common sturgeon were stocked (Williot *et al.*, 2000). The effectiveness of the stocking is currently being evaluated (Lochet, 2002).

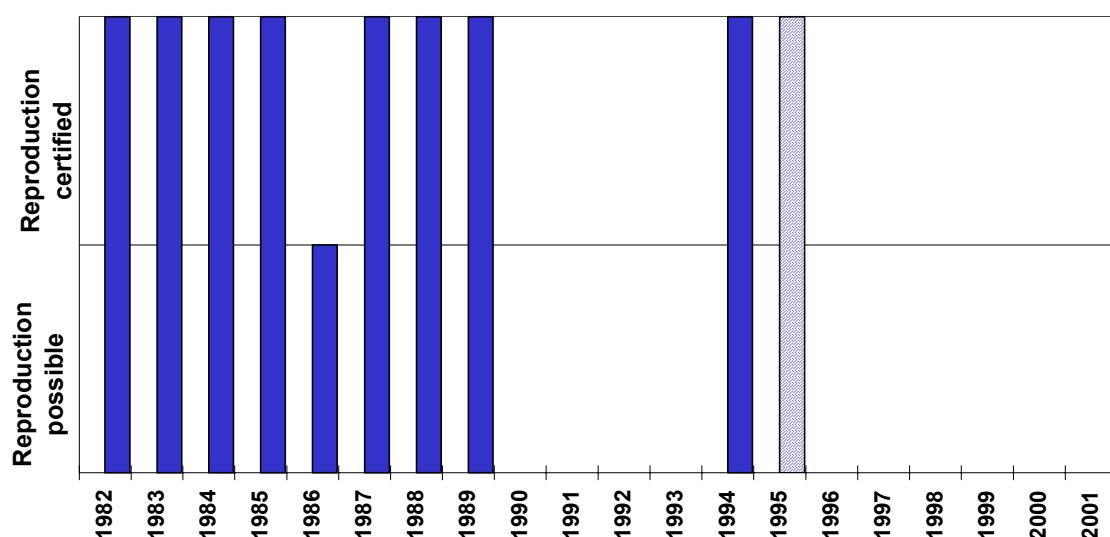


Figure 3.3.4: Deduction of the evidence of common sturgeon reproduction in the Gironde system from the monitoring of the population in estuary. The 1995 cohort comes from artificial reproduction (Williot *et al.* 2000) and stocking.

By comparing the number of fish within the cohorts which we estimate should return for spawning (i.e. fish born 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 ($M \gg 0.25$).

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, 1994; 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 20 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.* 2000 a and b) 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 has lead to the local extinction of a lot of common sturgeon populations. Presently, a major threat is their occurrence in the bycatch in coastal fisheries operating along the European sea board (Rochard *et al.*, 1997; Lepage *et al.*, 1998 a; Mayer and Lepage 2001). Poaching activities also occurred at sea at the entrance of the Gironde estuary (Mayer and Lepage 2001 ; Lepage, 2002).

Habitats in rivers are legally protected in 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.*, 1998 b; 2000). The incidental introduction of an alien sturgeon *Acipenser baerii* in the Gironde system in 2000 constitutes a new threat: these species can hybridise and probably compete for food. More over fishermen have difficulties protecting one species, as the non native species must be eradicated (Lepage, 2002).

Conclusion in relation to 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 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 recolonisation if it were to occur.

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3.4 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, Holland, Belgium, and France, and then down to Spain, Portugal and Morocco (Blanc *et al.* 1971; Lelek 1980) (Figure 3.4.1). 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).

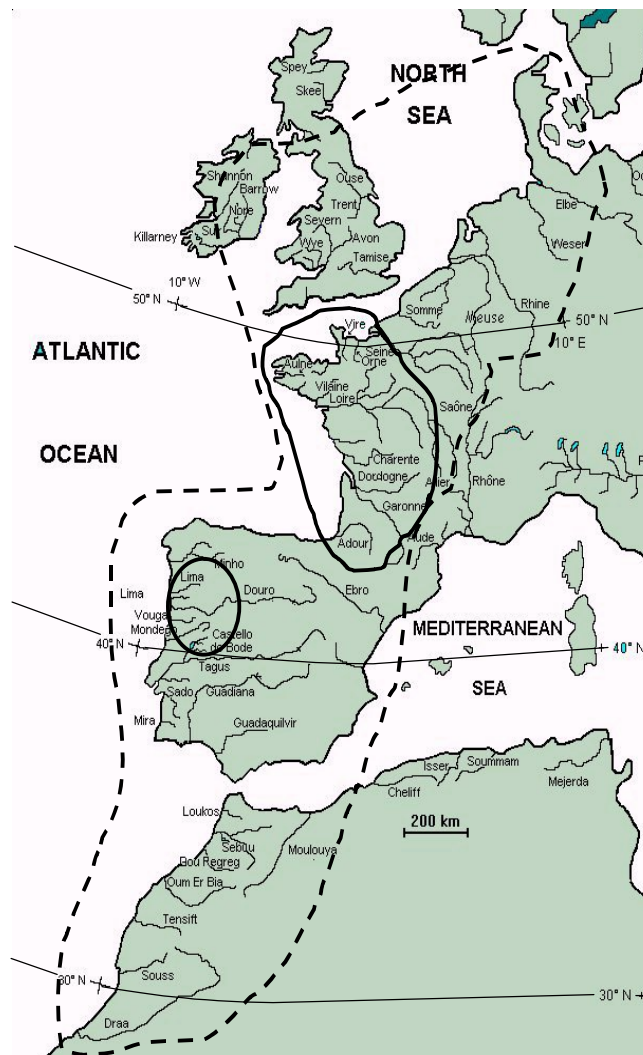


Figure 3.4.1: Historic (dashed line) and current (solid line) distribution areas of allis shad in the Eastern Atlantic. Shown are the main rivers colonized at the end of the 19th / beginning of the 20th century, and at the beginning of the 21st century (from Baglinière *et al.*, in press).

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 the species have been included in Appendix III of the Bern Convention and into Annexes II and V of the EC Habitats Directive. Within their distribution range *Alosa alosa* are considered extinct in 3 countries, critically endangered in 1, endangered in 6, vulnerable in 2 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, channelisation 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 c.1900 (Figure 3.4.2). The decline of *A. alosa* was due to over fishing, 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)

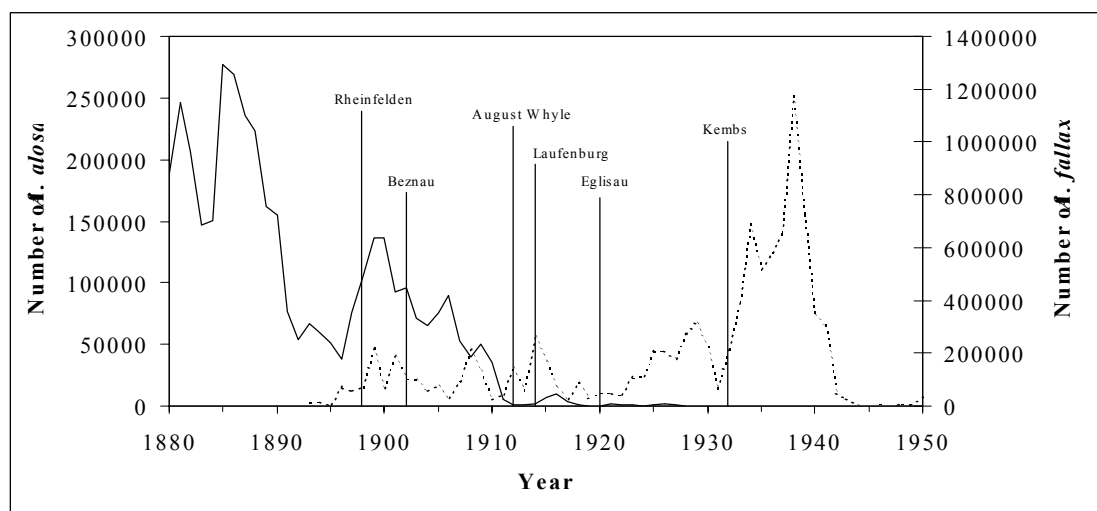


Figure 3.4.2 The catch of *Alosa alosa* (solid line) between 1880 and 1934, and of *A. fallax* (dotted line) between 1893 and 1950, from the lower Rhine (Data from de Groot, 1989). Vertical lines show the dates when dams were built.

At the turn of the nineteenth century both species of shad were present in Belgium (Anonymous, 1901a) and a spawning populations of *Alosa alosa* was present in the River Meuse (Anonymous, 1888). in the River Meuse. However, by 1925 *A. alosa* were 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 c. 1842 (Day, 1890). Along the North 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 (LeClerc, 1941).

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.*, 1994a; 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 have 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 increase in the CPUE in the Gironde fishery (Figure 3.4.3).

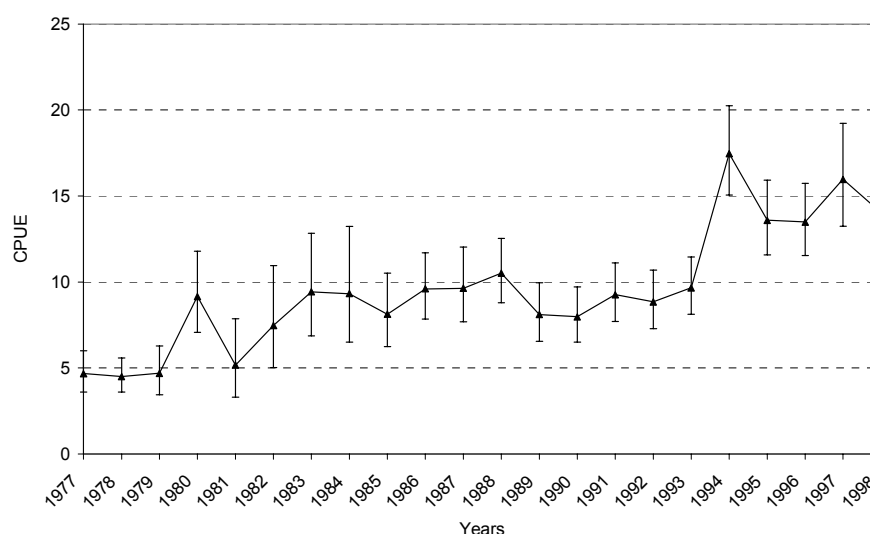


Figure 3.4.3: Annual variation in the number of *Alosa alosa* caught in the Gironde per net per day between 1977 to 1999 (Castelnaud *et al.*, 2001).

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 1981b, Alexandrino, 1996aandb; 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). While in the rivers Tagus and Minho the populations of *A. alosa* have declined dramatically (Figure 3.4.4) 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). Though 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 has 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 barrage Idriss ler and from pollution derived from the processing of sugar (Figure 3.4.5, 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 1968-69, 1929 and 1973 respectively.

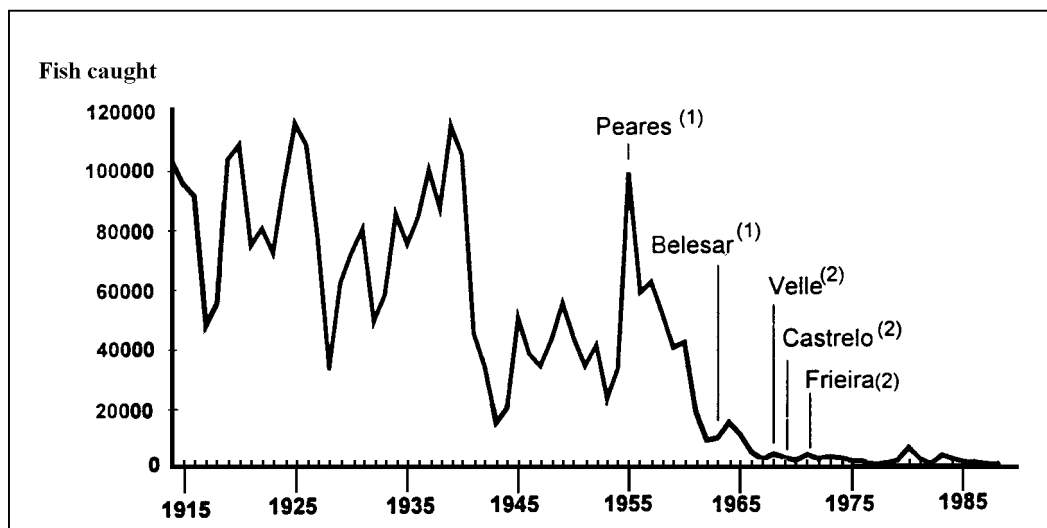


Figure 3.4.4 : The catch of *Alosa alosa* between 1914-1988 from the River Minho, and the dates when barrages were constructed; 1 = Lugo province, 2 = Orense province (Alexandrino, 1996b).

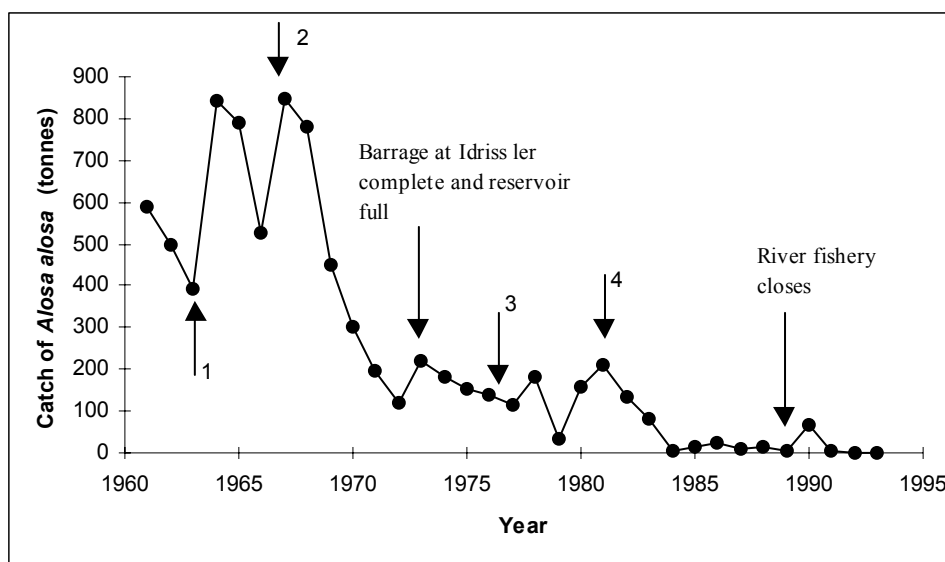


Figure 3.4.5. The combined marine and river catch of *Alosa alosa* from the Oued Sebou (Morocco) between 1961 and 1993. Numbers refer to when different sugar refineries were constructed (Sabatié, 1993).

Conclusion in relation to 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, 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 evidence 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 prevent the fish migrating freely (Taverny, *et al.*, 2000) and can lead to hybridisation 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 3.4.3. 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 extraction of gravel and stones from the river bed, and 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 (eg. 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 gill nets appears low.

At present, except for the Gironde, there is little information on the impact of a fishery on a population. On 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 included as protection measures for *A. alosa* will also afford protection to the twaite shad *A. fallax*.

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3.5 Houting (*Coregonus lavaretus oxyrinchus*)

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.*, 2002; Figure 3.5.1). 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 ever known to have occurred as a vagrant in coastal waters off the south east 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).



Figure 3.5.1. 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)).

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 North Europe (Hansen, 1997). The houting is now classified as *C. l. oxyrhynchus*.

In the late eighties 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 stream with a stable and self-producing population of houting. Between 1987-1992, 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 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 and Kongeåen.

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, from illegal fishing during the spawning migration and from 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 Texel-Faial criteria for the identification of threatened species and habitats

There is evidence of a decrease in both the area 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 reestablished.

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 estuary have been characterized, however pollution and bycatch poses a continual threat.

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3.6 Sea horses (*Hippocampus* spp.)

Status and distribution

Short-snouted seahorse (*Hippocampus hippocampus*) and long-snouted seahorse (*Hippocampus guttulatus* (formerly *H. ramulosus*)) were each nominated by one party (Portugal). The nominations for both species cited regional importance, decline and sensitivity.

Both these species are distributed from the Mediterranean and North-west Africa to the English Channel, with *H. guttulatus* extending further north along the western coasts of the British Isles and to 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 overwinter 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 (Fig 3.6.1) 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 Wadden Sea. The size and distribution of seahorse populations in the OSPAR area is not known.

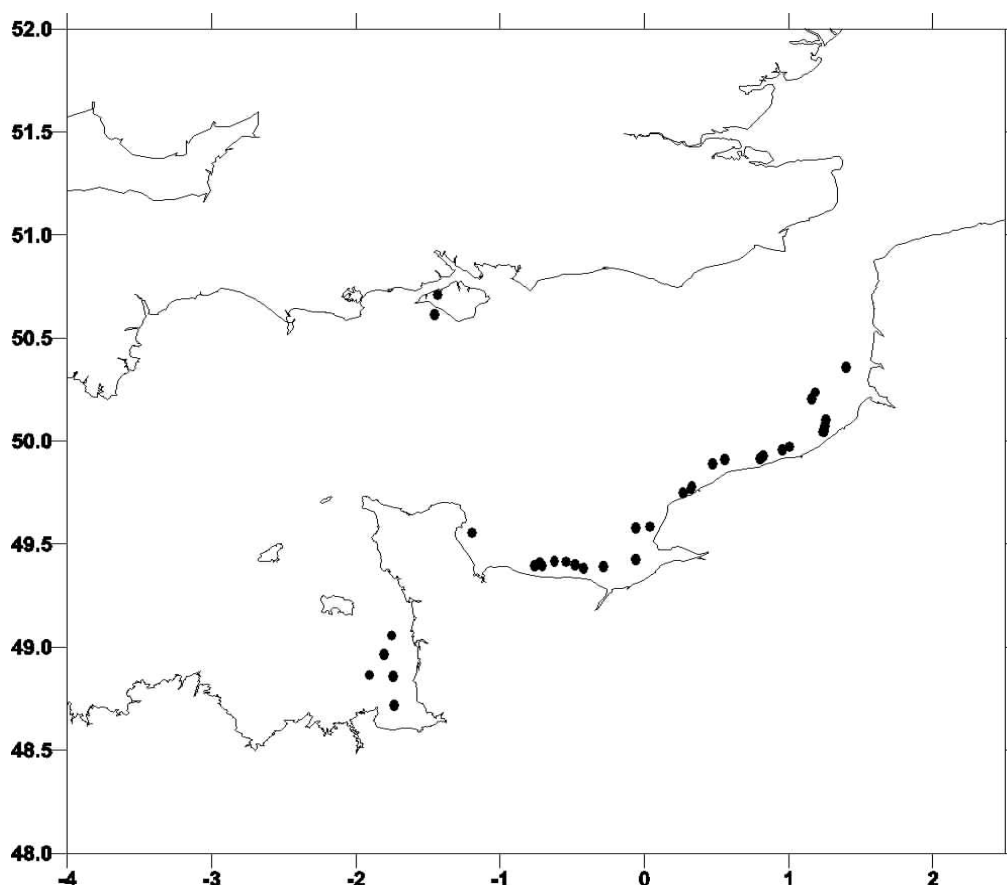


Figure 3.6.1: Occurrence of sea horses (*Hippocampus* spp.) in the eastern English Channel, as observed from CEFAS beam trawl surveys (1991-2002).

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 over-exploitation 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, 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 re-colonisation 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.

Short-snouted seahorse lives in mixed habitats of macro algae and rocky areas during the spring, summer and early autumn. The long-snouted seahorse occupies predominately eel grass beds during the spring, summer and early autumn, and migrate to deeper waters in winter. Thus conservation of sea grass 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 population of seahorses in the OSPAR area is 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 comprises 160 signatory nations (Parties), 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, and 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 number and in size of seahorses. Seahorse biology is such that populations will be particularly susceptible to over-fishing: (a) pregnant seahorses must survive if the young are to survive; (b) lengthy parental care combined with small brood size limits reproductive rate; (c) low mobility and small home ranges restrict recolonisation 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 means 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 the damage of 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 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. Sea grass 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 are considering to list both species to Schedule 5 on 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.

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3.7 Couch's goby (*Gobius couchi*)

Status and distribution

Only 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 in three locations in the British Isles: Helford in south Cornwall; Lough Hyne, Co. Cork, Ireland; and Mulroy Bay, Co. Donegal, Ireland. More recently, it has been recorded from the Mediterranean (Ischia 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 thought.

Potts and Swaby (1991) indicated that at the Helford site the population has decreased over the last ten years. We have no information on the situation for any of the other sites.

No fishery is known to target this species. Couch's goby uses coastal shallow habitats (3-1 m depth) and is often associated with substrate made of stones, boulders or shell debris or in 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 to pollution, but no particular threat has been identified. It's an omnivorous species feeding on polychaetes, algae, crustaceans and bivalves.

Conclusion in relation to Texel-Faial criteria for the identification of threatened species and habitats.

Couch's goby is a little-known species that was nominated by one party (Germany), due to its global importance, rarity, decline and sensitivity. However this review revealed that the status and distribution of this species is 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 appropriate 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.

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3.8 Sea turtles

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, ignoring or coming out of the ocean to go towards a light source, 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 of 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 species nests in the OSPAR region, although 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 bycatch 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 rope, particularly those used in pot fisheries targeting crustaceans and whelk. Rope entanglement occurs predominantly between July and October, on the north, west and south-west coasts of the UK and the south and west coasts of Ireland. Leatherbacks have also been caught and drowned in pelagic driftnets and bycatch has been documented in NW Atlantic pelagic longline fisheries (e.g. Witzell 1984, 1996). However, the significance of marine turtle bycatch in the OSPAR area is not known. Leatherback turtles are globally endangered and Spotila *et al* (1996) suggest that the impact of bycatch on Atlantic leatherback populations may be unsustainable. The threat of bycatch 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 of 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 catch were observed in the Greek monitoring program (22 loggerhead *Caretta caretta* and one leatherback *Dermochelys coriacea*), 220 turtle in the Italian program (218 loggerheads and two green turtle *Chelonia mydas*) and 2127 turtle in the Spanish one (2125 loggerheads and two leatherbacks). Turtle catch rates were highly variable depending on the fishing areas and fishing seasons, being higher during quarter 2 and 3 in Spain. Set operations as well as gear characteristics also affected the outcome (Laurent *et al.*, 2001).

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 by turtles. Furthermore by understanding the relative importance of the OSPAR area to marine turtles implementation of international conservation measures can be facilitated.

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 South Florida loggerhead 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 individual from the Mediterranean and the western population forage in OSPAR waters outside the breeding season (June to August).

Loggerhead 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 populations are coastal development and tourism, commercial fisheries (particularly shrimp trawling in nearshore areas), and various forms of pollution.

In the OSPAR area the greatest threat appears to come from bycatch in fishing gear, most often in fishing lines, including those used in trap fisheries. Loggerheads 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 byproducts. The impact on the population of the effects of these contaminants has not been determined. In areas where recreational boating and ship traffic is 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 biphenols (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 to develop 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 by turtles. Furthermore by understanding the relative importance of the OSPAR area to marine turtles implementation of international conservation measures can be facilitated.

Conclusion in relation to Texel-Faial criteria for the identification of threatened species and habitats

The data provided in the nomination for loggerhead meets the Texel-Faial criteria for declining and threatened species, although some available data on bycatch should be added for the OSPAR area.

In UK waters there has been a small number of records of loggerhead turtles *Caretta caretta*, although most specimens are thought to have been carried north by adverse currents from their usual habitats (Carr 1987; Penhallurick 1991; Mallinson 1991).

Leatherback Turtle *Dermochelys coriacea*

Current status

Leatherback turtles breed circumglobally within latitudes 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 adaptations 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 ($\pm 8,000$ 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 US 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 12 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 Texel-Faial criteria for the identification of threatened species and habitats.

The data provided in the nomination for Leatherback meets the Texel-Faial criteria for declining and threatened species, although some available data on bycatch should be added for the OSPAR area.

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3.9 Conclusions

WGFF considered that the nominations for sturgeon, houting, and both species of turtle and seahorse met the Texel-Faial criteria and that threats in the maritime area were relevant. Sturgeon is globally and locally important in the OSPAR region, rare, very sensitive to certain human activities and declines are marked. Houting is of local importance, occurring in only parts of OSPAR Region II, although there are populations in the Baltic. 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 is no accurate data to illustrate a decline, both species are known to utilise sea grass habitats, and such habitats have declined.

The group 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 improve local populations, the overall populations and their spatial extent has 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 (see section 6).

The group considered that the original classification according to the Texel-Faial criteria was inconsistent and a revised version is presented in Table 3.8.1.

Table 3.8.1: Revised classification of proposed fish species under the Texel-Faial criteria.

	Global importance	Local Importance	Rarity	Sensitivity	Keystone species	Decline
Sturgeon (<i>Acipenser sturio</i>)	✓	✓	✓	✓		✓
Houting (<i>Coregonus lavaretus oxyrinchus</i>)	?	✓	✓	✓		✓
Short-snouted seahorse (<i>Hippocampus hippocampus</i>)	?	?	✓	✓		?
Seahorse (<i>Hippocampus ramulosus</i>)	?	?	✓	✓		?
Allis shad (<i>Alosa alosa</i>)	✓		✓	✓		✓
Sea lamprey (<i>Petromyzon marinus</i>)	✓			✓		✓
Couch's goby (<i>Gobius couchi</i>)	?	?	?			?

4 REVIEW OF ESSENTIAL FISH HABITAT

4.1 Introduction

That anthropogenic activities have detrimentally affected aquatic habitats is widely reported (e.g. Schmitten, 1999; Rosenberg *et al.*, 2000; Collins *et al.*, 2000), and examples include:

- Physical alteration of rivers and estuaries, including obstructions, which may restrict the spawning migrations of anadromous fish species.
- Poor water quality (pollution, dissolved oxygen, eutrophication) and physical alterations in estuarine and inshore environments that may affect diadromous species and the juvenile stages of fishes that have inshore nursery grounds, including areas of aquatic vegetation (e.g. macroalgae and seagrass beds).
- Trawling disturbance that impacts on fragile habitats (e.g. *Oculina* and *Lophelia*, sponge communities) that are thought to support fish productivity.

Essential Fish Habitat (EFH) is an issue of increasing awareness to fisheries scientists, and a major policy driver in the USA and Canada (Benaka, 1999). In 1996, the USA Congress added habitat conservation measures to the Magnuson-Stevens Fishery Conservation and Management Act, which states “*One of the greatest long-term threats to the viability of commercial and recreational fisheries is the continuing loss of marine, estuarine, and other aquatic habitats. Habitat considerations should receive increased attention for the conservation and management of fishery resources of the United States*” (16 U.S.C. 1801 (A)(9)).

The USA Congress defined EFH as “*those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity*”, where **waters** are “aquatic areas and their associated physical, chemical, and biological properties that are used by fish and may include aquatic areas historically used by fish where appropriate”; **substrate** “includes sediment, hard bottom, structures underlying the waters, and associated biological communities”; **necessary** means “the habitat required to support a sustainable fishery and the managed species' contribution to a healthy ecosystem”; and “spawning, breeding, feeding, or growth to maturity” covers a species' full life cycle.

To this end, the National Marine Fisheries Service (NMFS) and the regional Fishery Management Councils were required to describe and identify EFH for each federal Fishery Management Plan, to minimize (where practicable) the adverse effects of fishing on EFH, and to develop other measures to promote the conservation and enhancement of EFH. NMFS have subsequently produced preliminary accounts of EFH designations for various species and stocks (e.g. Steimle *et al.*, 1999) for which there are Fishery Management Plans.

Habitat Areas of Particular Concern (HAPCs) are the most critical or vulnerable types of EFH (Rosenberg *et al.*, 2000), and such sites are identified by the importance of the ecological function, the extent to which the habitat is sensitive to anthropogenic degradation, and the rarity of the habitat type. Examples of HAPCs identified in US waters include various estuaries and coastal inlets, areas of hard bottoms/reefs, mangroves, seagrass beds and oyster/shell habitats (e.g. Minello, 1999; Packer and Hoff, 1999; Dobrzynski and Johnson, 2001). Hence, it may be that only a proportion of a fish species' biogeographical range can be considered as EFH, and in turn only a proportion of its EFH considered critical (e.g. Habitat Areas of Particular Concern). This is illustrated theoretically in Figure 4.1.1 and an example of the relationship between the distribution, EFH and Habitat Area of Particular Concern for Atlantic cod *Gadus morhua* on the George's Bank is illustrated in Figure 4.1.2.

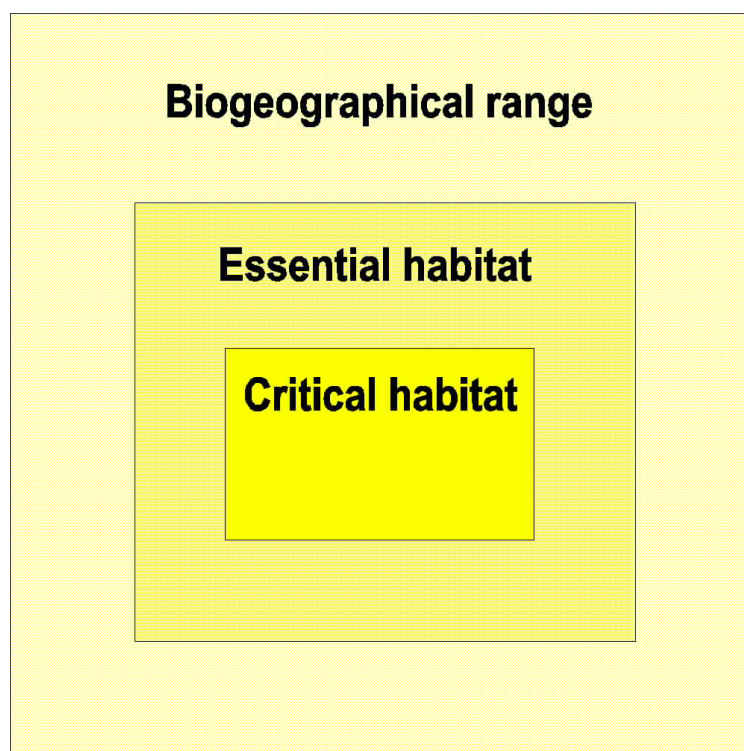


Figure 4.1.1: Theoretical relation of critical habitat and essential fish habitat with the biogeographical distribution of the species. Adapted from Cross *et al.* (1997).

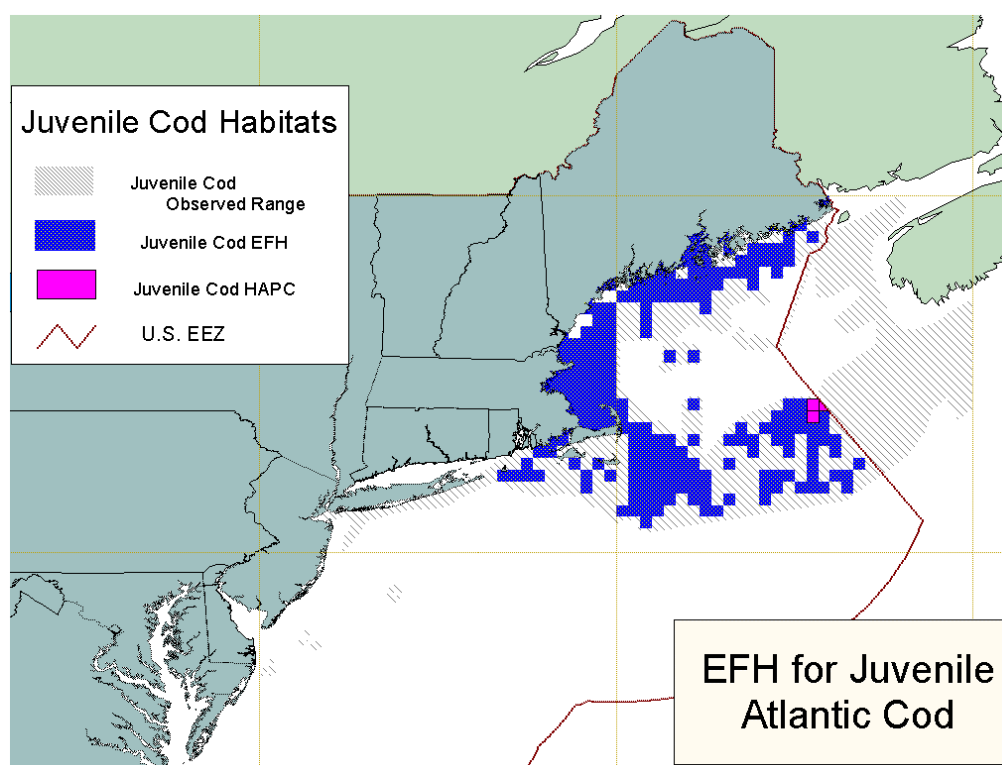


Figure 4.1.2: Juvenile cod habitats in the George's Bank area, illustrating overall range, EFH and HAPC (<http://www.nmfs.noaa.gov/habitat/habitatprotection/factsheets/areas.pdf>).

4.2 Review of the concept of Essential Fish Habitat

Although it is widely acknowledged that habitat is important, traditional fishery assessment and management rarely consider the role of habitat in the survivorship of juvenile fishes (Thrush *et al.*, 1998).

Fishes may exploit a variety of habitats throughout their various life-history stages (e.g. Cross *et al.*, 1997), and whereas some sites may be spatially defined, others may vary on a range of temporal and spatial scales. The types of site that may be regarded as EFH for particular species would include:

- Breeding, spawning and parturition grounds
- Nursery grounds (for post-larvae, neonates and juveniles)
- Shelter and natural refuges
- Feeding grounds
- Migratory corridors

Furthermore, the grounds utilised by those species that exhibit high habitat specificity or are endemic to restricted locations may also be regarded as EFH. Indeed, many fish species that are widely viewed as threatened are endemic to certain areas, for example the totoaba *Totoaba macdonaldi* in the northern Gulf of California (Barrera-Guevara, 1990; Cisneros-Mata *et al.*, 1995). The following sections provide a few examples of case studies where habitats may be considered essential or critical.

4.2.1 Breeding, spawning and parturition grounds

- Some fish may have specific grounds where adults mate. For example the nurse shark *Ginglymostoma cirratum* is known to aggregate at a specific site on the Florida Keys (Carrier and Pratt, 1998), and this area was recognised as “unique and critical to the breeding success of the sharks”, as well as serving as a nursery ground for juveniles.
- There are some areas used by many species of fish for spawning (i.e. broadcast spawners). An example of this would be Trevoze Head (Bristol Channel), where several species of teleost are known to spawn (Horwood *et al.*, 1998; Horwood, 2000).
- Spawning of those species having specific substrate requirements for their eggs (e.g. herring) may be spatially restricted by suitable spawning grounds. Additionally, if particular spawning grounds afford the greatest chances of successful development and survivorship of eggs and larvae, then these could be considered essential.

4.2.2 Nursery grounds

- Nursery areas are often characterised by high production, abundant and suitable food and habitat resources and reduced predation (e.g. Castro, 1993). Nursery grounds for many commercially important shelf species occur in estuarine and coastal environments (Figure 4.2.1).
- Several important nursery grounds in north-west Europe have technical restrictions to afford some protection to juvenile fishes, examples including the Plaice Box (established in 1989), Mackerel Box and various UK estuarine sites that are important for bass (Rogers, 1997, Horwood, 2000).
- The New England Fishery Management Council has recently designated an area of gravel sediment on Georges Bank as a HAPC, as it was considered vital for juvenile cod (Rosenberg *et al.*, 2000; Rieser, 2000).
- Many species of elasmobranch have nursery grounds in shallow bays and estuaries (Castro, 1993; Simpfendorfer and Milward, 1993; Ellis *et al.*, 2002).

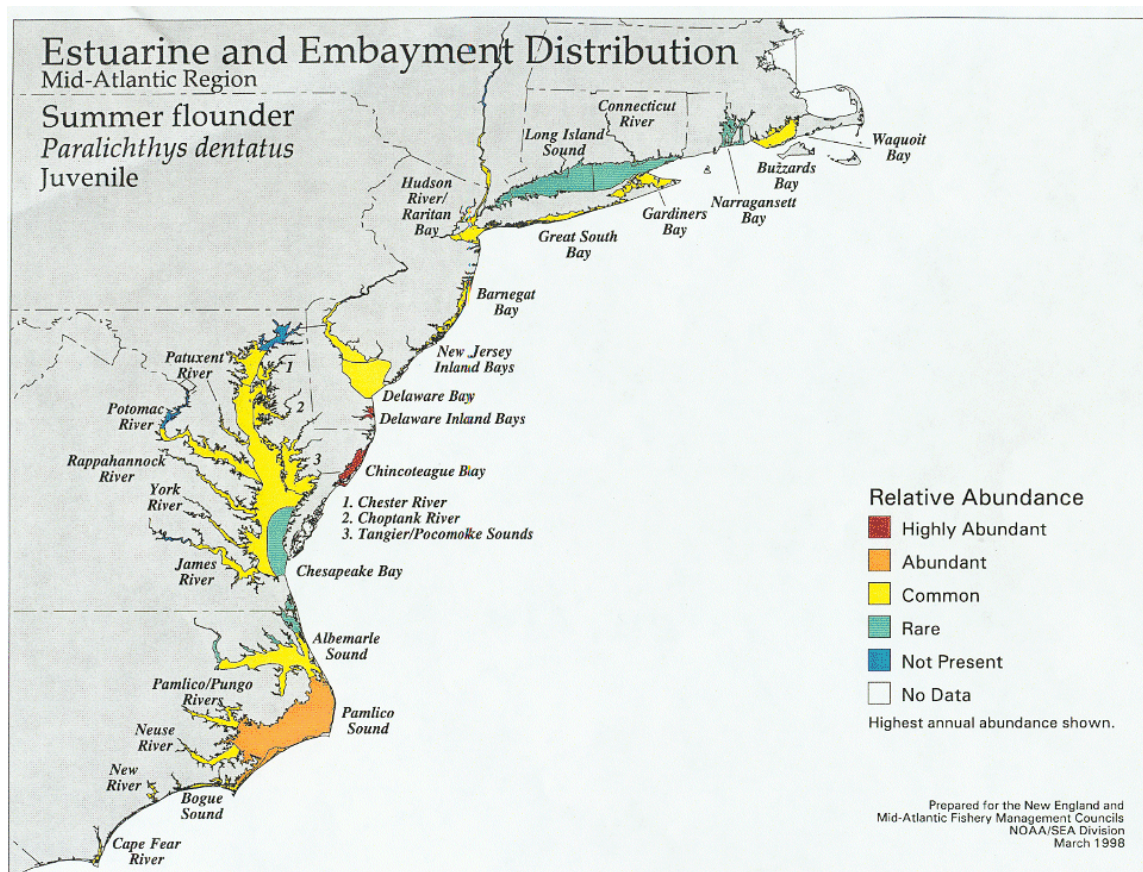


Figure 4.2.1: Distribution and relative importance of estuarine and inshore sites for juvenile summer flounder *Paralichthys dentatus* (Source: National Marine Fisheries Service).

4.2.3 Shelter, and natural and artificial refuges

- Several studies have shown that survivorship of post-larval and juvenile fishes are enhanced in structurally complex habitats (Gotceitas and Brown, 1993; Auster *et al.*, 1997; Lindholm *et al.*, 1999; Norse and Watling, 1999). These habitats may be of geological and/or biological origin. Biogenic habitats are most vulnerable to impacts from towed fishing gear and have the slowest post-impact recovery time (Collie *et al.*, 2000).
- Yoklavich *et al.* (2000) reported that certain large-bodied species of rockfish (e.g. *Sebastes chlorostictus*, *S. levis*, *S. rosenblatti*, and *S. ruberrimus*) were locally abundant on specific habitats (e.g. rock ledges, caves and overhangs) and that such deep-water rocky outcrops that were less accessible to fishing activities could provide an important natural refuge for commercial fishes.
- Complex geological complex habitats are thought to be productive areas and the North Pacific Fishery Management Council has recently protected the Edgecumbe pinnacle (Alaska) and associated boulder field through gear restrictions (Rosenberg *et al.*, 2000). Similarly, the Charleston Bump of South Carolina is regarded as ecologically important to a variety of commercial fish species (Sedberry *et al.*, 2001).
- Biogenic habitats with high habitat complexity may support a high faunal diversity, provide shelter for fish (including juveniles) and, in some areas, may also act as important spawning habitats, for example the commercially important reef-fishes *Mycteroperca microlepis* and *M. phenax* aggregate to spawn on *Oculina* reefs in the north-eastern Gulf of Mexico (Koenig *et al.*, 2000).
- Oyster reef habitats are also suggested to act as an important habitat for several fish and shellfish species, as living oysters and dead shells increase the structural complexity, thereby supporting diverse communities of crustaceans and demersal fish (Peterson *et al.*, 2000).
- Artificial structures, including reefs, wrecks and oil rigs may also act as locally important areas for fish aggregations (e.g. Kasprzak, 1998; Wantiez and Thollot, 2001)

4.2.4 Feeding grounds

- Some fish species may aggregate at specific feeding grounds (e.g. Chase, 2002), although these are generally less well defined than spawning and nursery grounds. For example, whale sharks *Rhincodon typus* are known to aggregate at certain sites (e.g. Ningaloo Reef in Australia and Gladden Spit in Belize) where they feed during peak spawning activities of reef-fish (Heyman *et al.*, 2001; Wilson *et al.*, 2001).
- In oceanic ecosystems, fishes may aggregate at frontal and upwelling systems, although such oceanographic features will vary spatio-temporally.

4.2.5 Migratory corridors

- Species of fish that have restricted migratory corridors (bottlenecks) may also be negatively and seriously impacted by anthropogenic activities. Examples would include diadromous species (e.g. sturgeon and lampreys), for which freshwater spawning grounds and freshwater and estuarine juvenile habitats may be considered essential (Collins *et al.*, 2000).

4.3 Data requirements for identifying Essential Fish Habitat

The primary data requirements for the preliminary identification of EFH for any given species will be descriptive or quantitative information on the geographic range of all life-history stages at an appropriate spatial and temporal resolution (Schmitt, 1999). Specifically, the types of information that are required are the:

- Current and historical population (or stock) size
- Biogeographical distribution
- Spatio-temporal distribution and relative abundance of the various life-history stages (an example of which is illustrated in Fig. 4.3.1)
- Biological and habitat requirements for the various life-history stages
- Ecological and environmental characteristics, and spatial distribution of habitats
- Population density, survivorship, growth rate, breeding success and production rates for habitats used

Data that are required to assist in the interpretation of the relative importance of habitats include long-term data sets of habitat use, an improved knowledge of the distribution and ecological stability of aquatic habitats, and improved knowledge of fish behaviour in specific habitats (Able, 1999).

Although the quality of available data will vary among species, habitats and geographical areas, studies that would assist in the interpretation of the potential role of EFH will include post-larval studies, understanding of spawning behaviour, the role of offshore reefs (geological and biological) and other hard bottoms as spawning grounds and refuges, the role of habitat in survivorship of various life-history stages, site fidelity, and the locations of high abundance of threatened species.

For management purposes, it may also be important to be able to delineate and map habitats deemed of ecological importance. Rubec *et al.* (1999) and Brown *et al.* (2000) have recently illustrated that modelling indices of habitat suitability, in conjunction with geographic information systems (GIS) could be used as a method of analysing and delineating the overall range of a species, areas of abundance and EFH.

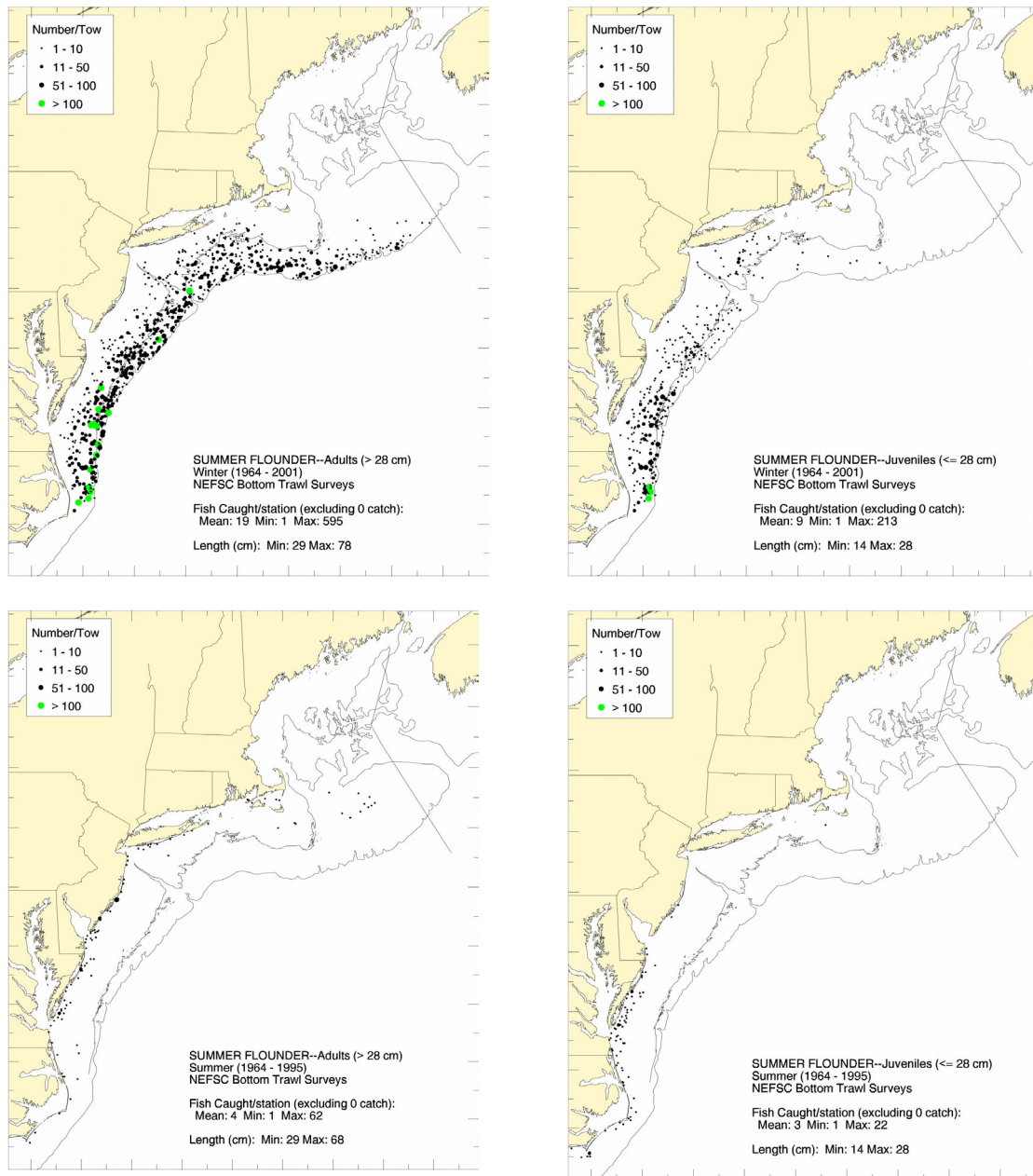


Figure 4.3.1: Distribution of adult and juvenile summer flounder *Paralichthys dentatus* during winter and summer groundfish surveys (Source: National Marine Fisheries Service).

4.4 Conclusions

The main rationale behind identifying EFH and critical habitats is that certain sites may be of significant importance for, for example, increased survivorship and growth of various life-history stages, and reproductive success of fish species, particularly those of commercial or conservation importance. Whereas EFH is defined as "*those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity*", in practical terms habitats should be viewed in terms of their spatial extent and sensitivity, with priority given to those critical fish habitats that are spatially restricted and/or sensitive to disturbance.

Obviously, the identification of critical fish habitats is only one process in the concept of EFH. If anthropogenic activities can detrimentally affect the functional role of such habitats, then either improved management of human activities or protection may be required for the fish stocks and habitats in question.

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5 GASTRIC EVACUATION, TOTAL STOMACH CONTENT, PREY COMPOSITION AND FOOD RATION

5.1 Introduction

The main objective of this section was to prepare a thorough revision of the estimates of food rations of MSVPA predators in the North Sea based on the latest years improvements in gastric evacuation modeling. Specifically, the aims were to initiate in 2003;

- i) the compilation of ICES data on individual stomach contents as well as other relevant stomach data to obtain estimates of the correction factor k ;
- ii) the estimation of food rations of whiting and saithe (feeding primarily on fish) using a prey energy density dependent evacuation model to quarterly mean stomach data;
- iii) the compilation of the ICES stomach data to acquire possible information about prey composition of individual stomachs. The focus will be on major prey types that deviate significantly in energy density;

and to initiate for completion in 2004;

- iv) the estimation of food rations of cod and haddock (preying to a large extent on invertebrates with a robust exoskeleton) using a gastric evacuation model which accounts for the effects of a robust exoskeleton on evacuation of total stomach content as well as individual prey types. Such a model is expected to be in operation around mid-2003.

5.2 Correction Factor k

Recent research has shown that mean consumption rates, through time and across the population, of predatory fishes may be estimated by

$$\hat{C} = \overline{\rho S^{0.5}}$$

where S and ρ are the total mass and evacuation rate constants, respectively of the stomach content of each individual sampled in the field (Andersen, 2001). The ICES stomach contents data are generally pooled by predator size. Therefore only mean values \bar{S} of stomach contents are available. However, the values of $\bar{S}^{0.5}$ and $\bar{S}^{0.5}$ differ unless all values of S are equal. Consumption rates as estimated by

$$\hat{C} = \bar{\rho} \bar{S}^{0.5}$$

are therefore likely to be biased by the frequency distribution of S . This bias may be remedied by introduction of a correction factor $k = \bar{S}^{0.5} \bar{S}^{-0.5}$ obtained from data on individual stomachs (Andersen, 2001):

$$\hat{C} = \bar{\rho} k \bar{S}^{0.5}$$

The correction factor may be further refined by splitting the influence of the S variation into the frequency of empty stomachs ($S=0$) and the frequency distribution of S in stomachs with food. This way, the information about the frequency of empty stomachs that are available for the pooled ICES data set can be used to modulate the value of k as follows:

N_F is the number of stomachs with food (including the regurgitated ones)

N_E is the number of empty stomachs

N_T is the total number of stomachs

S is the total food mass in a stomach

The correction factor including all stomachs in a sample of individual stomachs:

$$k_T = \frac{\frac{\sum S^{0.5}}{N_T}}{\left(\frac{\sum S}{N_T}\right)^{0.5}} = \frac{\sum S^{0.5}}{(\sum S)^{0.5}} N_T^{-0.5}$$

The correction factor obtained exclusively from stomachs with food:

$$k_F = \frac{\frac{\sum S^{0.5}}{N_F}}{\left(\frac{\sum S}{N_F}\right)^{0.5}} = \frac{\sum S^{0.5}}{(\sum S)^{0.5}} N_F^{-0.5}$$

$\Rightarrow \sum S^{0.5} = k_F N_F^{-0.5} (\sum S)^{0.5}$ inserted into the above expression for k_T gives:

$$k_T = \frac{\sum S^{0.5}}{(\sum S)^{0.5}} N_T^{-0.5} = \frac{k_F N_F^{-0.5} (\sum S)^{0.5}}{(\sum S)^{0.5}} N_T^{-0.5} = k_F \left(\frac{N_T}{N_F}\right)^{-0.5}$$

Thus, applied on a pooled set of ICES data:

$$k_T = k_F \left(\frac{N_{T,ICES}}{N_{F,ICES}}\right)^{-0.5}$$

where k_F is obtained from a representative set of individual stomachs. $N_{F,ICES} = N_{T,ICES} (1 - 0.01p)$ if p is the percentage of empty stomachs within the pooled set of ICES data as given by Hislop (1997).

Calculations from various North Sea predators showed that the values of k_F obtained from different data-sets of a predator species are relatively similar whereas the number of empty stomachs may heavily influence the k_T values as exemplified by whiting sampled near Devil's hole in 1992 (Table 5.1). k_F values for most of the major predatory fish entering the MSVPA has been calculated and compiled in Table 5.1. Substantial amounts of data on individual stomachs that were not available for the present meeting have been identified. No comparisons and conclusions about values of k_F will therefore be done until these additional sets have been analysed.

5.3 Food rations of saithe and whiting estimated from quarterly mean stomach data

Quarterly food rations of whiting and saithe (Tables 5.2.1 and 5.2.2) were estimated using quarterly mean stomach data from Hislop (1997). Temperature data were obtained by the procedure described in Andersen (2001), prey energy data were compiled using Pedersen and Hislop (2001), and the gastric evacuation model parameterized for whiting and saithe was taken from Andersen (2001). These estimates of the quarterly food rations of whiting and saithe will subsequently be compared to estimates obtained from the approach described below.

5.4 Prey composition of individual stomachs

Prey composition of individual stomachs sampled in the field is not necessarily similar to the mean value obtained from the pooled stomach contents. The rate constant ρ may be heavily influenced by prey composition, e.g. by energy density of fish prey and exoskeleton of invertebrates (Andersen, 1999). Therefore, estimated consumption rates of individual prey types, and in consequence the prey composition of ingested food, may be biased by application of the mean value of ρ which is based on mean prey composition of pooled stomach samples (Andersen, 2001). The energy density is the most important characteristic of fish prey influencing the gastric evacuation rate. The large potential for bias that may be introduced by application of quarterly mean values of total mass and prey composition of pooled stomach contents to estimate food rations was demonstrated by Andersen (2001) for North Sea saithe of age 5-8 years feeding on prey of highly different energy densities.

The intention of this TOR sub-section was to scrutinize the ICES sub sets on individual stomachs to acquire general information about prey composition of individual stomachs for each predator. These sub-sets cannot, however, be regarded as being representatives of the entire data set for each predator. Therefore, another approach has been adopted: Food rations will be estimated from pooled ICES stomach data on haul basis taking advantage of the information about prey composition at this low aggregation level. These rations will be compared with those obtained from quarterly mean stomach data (like the ones for whiting and saithe in Tables 5.2.1 and 5.2.2).

5.5 Estimation of food rations of fish feeding on invertebrates with a robust exoskeleton

A variety of invertebrates with robust exoskeletons constitute a significant element in the diet of haddock, cod and grey gurnard. The description of evacuation of mixed meals including prey with a robust exoskeleton is complicated because more prey characteristics - i.e. both prey energy density as well as exoskeletal barrier - influence the evacuation process. However, a model based on a geometric, mechanistic interpretation of the square root principle ($dS/dt = -\rho S^{0.5}$) and with inherent rules for the effects of the above prey characteristics as well as the prey size has been established and subsequently verified from data on whiting fed mixed meals of sandeel and brown shrimp (Andersen and Beyer in prep.). Gastric evacuation experiments on cod fed a variety of invertebrates are presently being performed to enable a general parameterization of the model regarding the barrier effect of exoskeletons.

In addition to a suitable gastric evacuation model for these predators we need information about the temperature regime. Data on the geographical distribution of quarterly mean temperature by depth stratum and abundance of each of the predators by age class were combined to estimate the mean temperature experienced by the predator (compiled in Table 5.3 for four predators – more data will soon be available).

5.6 Conclusions

The data used in the present study to estimate values of the correction factor k do not cover all MSVPA predators and originate from limited geographic areas. There is therefore a need to include further existing data-sets on individual stomachs to acquire estimates for all predators as well as to achieve a better geographical coverage.

The mechanistic, gastric evacuation model taking into account the different prey characteristics will soon be fully parameterized. Food ration estimates of all MSVPA predators should then be revised using the two approaches (i.e. application of quarterly mean stomach data and stomach data on haul level), and the consequences of using each of the two sets of ration estimates as input to MSVPA/FOR runs should be examined.

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Table 5.1. Correction factors k_T and k_F obtained from data on individual stomachs of major North Sea predatory fishes.

Predator		Sampling		N_T	Empty (%)	k_T	k_F
Species	Size class (cm)	Locality	Season				
Saithe	30-34	Eigersund	August 1996	142	11.3	0.70	0.74
	35-39			500	11.2	0.73	0.77
	40-44			510	8.0	0.70	0.73
	45-49			355	11.3	0.72	0.76
Saithe	35-39	Eigersund	August 1997	213	2.8	0.86	0.87
	40-44			138	10.9	0.75	0.79
	45-49			111	8.1	0.80	0.84
	50-54			122	9.0	0.83	0.87
Grey gurnard	15-19	Central NS	August 1997	67	59.7	0.45	0.71
	20-24			365	26.8	0.67	0.79
	25-29			365	21.9	0.71	0.80
	30-34			241	32.0	0.73	0.88
	35-39			44	15.9	0.79	0.86
Whiting	10-14	Devil's Hole	June 1992	288	14.9	0.75	0.82
	15-19			299	12.0	0.71	0.76
	20-24			253	25.3	0.60	0.69
	25-29			261	33.3	0.58	0.72
	30-34			65	46.2	0.52	0.71
Whiting	10-14	Devil's Hole	June 1993	95	11.6	0.78	0.83
	15-19			401	12.0	0.79	0.84
	20-24			468	15.4	0.70	0.77
	25-29			479	25.3	0.62	0.72
	30-34			314	22.0	0.63	0.72
	35-39			44	22.7	0.69	0.78
Whiting	12-14	Firth of Forth	June 1997	44	2.3	0.70	0.70
	15-19			282	0.4	0.75	0.76
	20-24			332	0.3	0.73	0.73
	25-29			186	0.5	0.72	0.73
	30-39			22	0.5	0.62	0.63
Whiting	12-14	Firth of Forth	Sept 1997	89	1.1	0.73	0.73
	15-19			144	0.7	0.71	0.72
	20-24			254	0.4	0.64	0.64
	25-29			118	0.8	0.58	0.59
	30-39			27	0	0.53	0.53
Whiting	12-14	Firth of Forth	March 1998	166	0.6	0.56	0.57
	15-19			195	0.5	0.63	0.64
	20-24			121	0.8	0.68	0.68
	25-29			124	0.8	0.73	0.73
	30-39			31	3.2	0.67	0.68

Table 5.1. (continued). Correction factors k_T and k_F obtained from data on individual stomachs of major North Sea predatory fish.

Predator		Sampling		N_T	Empty (%)	k_T	k_F
Species	Size class (cm)	Locality	Season				
Whiting	12-14	Firth of Forth	June 1998	97	1.0	0.85	0.85
	15-19			149	0.7	0.82	0.82
	20-24			186	0.5	0.78	0.78
	25-29			106	0.9	0.78	0.78
	30-39			11	9.1	0.70	0.73
Whiting	12-14	Moray Firth	June 1997	29	3.4	0.93	0.94
	15-19			146	0.7	0.89	0.90
	20-24			137	0.7	0.81	0.81
Haddock	15-19	Firth of Forth	June 1997	38	0	0.78	0.78
	20-24			273	0.4	0.79	0.80
	25-29			299	0.3	0.80	0.80
	30-34			312	0.3	0.81	0.81
	35-39			213	0.5	0.78	0.78
	40-59			16	0	0.82	0.82
Haddock	12-14	Firth of Forth	Sept 1997	68	1.5	0.86	0.86
	15-19			88	0	0.90	0.90
	20-24			69	1.4	0.84	0.85
	25-29			152	0.7	0.75	0.75
	30-34			167	0.6	0.73	0.74
	35-39			120	0	0.73	0.73
	40-59			12	0	0.80	0.80
Haddock	15-19	Firth of Forth	March 1998	65	1.5	0.88	0.90
	20-24			63	1.6	0.80	0.81
	25-29			82	1.2	0.77	0.78
	30-34			65	1.5	0.74	0.75
	35-39			28	3.6	0.77	0.79
	40-59			12	0	0.72	0.72
Haddock	15-19	Firth of Forth	June 1998	65	1.5	0.92	0.92
	20-24			135	0.7	0.91	0.92
	25-29			141	0.7	0.87	0.87
	30-34			150	0.7	0.84	0.84
	35-39			150	0	0.85	0.85
	40-59			27	0	0.83	0.83
Haddock	15-19	Moray Firth	June 1997	36	0	0.87	0.87
	20-24			101	1.0	0.81	0.81
	25-29			72	1.4	0.87	0.87
	30-34			79	0	0.85	0.85
	35-39			14	7.1	0.78	0.81

Table 5.1. (continued). Correction factors k_T and k_F obtained from data on individual stomachs of major North Sea predatory fish.

Predator		Sampling		N_T	Empty (%)	k_T	k_F
Species	Size class (cm)	Locality	Season				
Cod	12-14	Firth of Forth/ Moray Firth	June 1997	4	0	0.99	0.99
	15-19		Sept 1997	67	3.0	0.76	0.79
	20-24		March 1998	157	1.3	0.78	0.78
	25-29		June 1998	110	3.6	0.77	0.85
	30-34			58	3.4	0.76	0.88
	35-39			26	3.8	0.84	0.89
	40-49			27	7.4	0.86	0.91
	50-79			15	13.3	0.83	0.90

Table 5.2.1. Quarterly food ration of a North Sea whiting by age and quarter of 1991.

QUARTER 1							
PREY	age 0	age 1	age 2	age 3	age 4	age 5	age 6+
<i>Gadus morhua</i>		0.00	0.00	0.00	0.00	0.00	0.00
<i>Melanogrammus aeglefinus</i>		0.01	0.23	0.54	1.01	1.14	1.50
<i>Merlangius merlangus</i>		0.22	4.24	7.80	13.29	14.06	19.99
<i>Trisopterus esmarki</i>		4.16	14.19	20.33	26.82	29.23	35.67
<i>Pollachius virens</i>		0.00	0.00	0.01	0.13	0.40	1.27
<i>Clupea harengus</i>		0.32	2.25	3.40	4.34	5.01	6.48
<i>Sprattus sprattus</i>		1.89	10.17	13.68	14.95	15.61	15.70
Ammodytidae		5.46	11.86	12.66	11.71	11.91	9.61
Microstomus kitt		0.00	0.00	0.00	0.00	0.00	0.00
<i>Solea solea</i>		0.00	0.00	0.00	0.00	0.00	0.00
<i>Limanda limanda</i>		0.00	0.00	0.00	0.00	0.00	0.00
Other fish		2.93	6.69	8.09	10.59	11.02	13.02
Annelida		2.97	2.98	2.69	2.73	2.83	2.78
Mollusca		2.71	2.20	3.58	4.27	4.55	4.50
Crustacea		7.16	6.36	5.88	6.73	7.01	8.23
Other prey		0.21	0.12	0.11	0.20	0.22	0.35
Total food ration (g)		28.03	62.02	78.78	96.76	103.0	119.1

Table 5.2.1 (continued). Quarterly food ration of a North Sea whiting by age and quarter of 1991.

QUARTER 2							
PREY	age 0	age 1	age 2	age 3	age 4	age 5	age 6+
<i>Gadus morhua</i>		0.89	1.30	1.08	0.87	0.76	0.72
<i>Melanogrammus aeglefinus</i>		0.10	0.35	0.67	0.72	0.63	0.43
<i>Merlangius merlangus</i>		0.14	0.51	1.16	2.35	3.17	4.13
<i>Trisopterus esmarki</i>		1.00	8.16	15.29	21.40	24.49	26.66
<i>Pollachius virens</i>		0.00	0.00	0.00	0.00	0.00	0.00
<i>Clupea harengus</i>		0.19	2.01	4.83	7.97	9.96	9.89
<i>Sprattus sprattus</i>		3.91	11.26	12.61	14.67	16.00	16.93
Ammodytidae		31.65	60.34	80.52	97.69	105.19	105.08
Microstomus kitt		0.00	0.00	0.00	0.00	0.00	0.00
<i>Solea solea</i>		0.00	0.00	0.00	0.00	0.00	0.00
<i>Limanda limanda</i>		0.22	0.00	0.00	0.00	0.00	0.00
Other fish		5.05	12.92	16.86	19.85	21.09	20.18
Annelida		6.28	5.84	6.47	7.12	7.37	7.34
Mollusca		2.01	1.40	1.23	1.09	1.03	1.02
Crustacea		17.65	25.86	28.74	31.14	32.08	31.61
Other prey		1.42	1.23	1.23	1.49	1.65	1.78
Total food ration (g)		70.51	131.2	170.7	206.4	223.4	225.8

Table 5.2.1 (continued). Quarterly food ration of a North Sea whiting by age and quarter of 1991.

QUARTER 3							
PREY	age 0	age 1	age 2	age 3	age 4	age 5	age 6+
<i>Gadus morhua</i>	0.000	0.22	1.37	1.65	1.75	1.88	1.70
<i>Melanogrammus aeglefinus</i>	0.094	3.39	9.44	10.63	12.15	13.72	14.55
<i>Merlangius merlangus</i>	0.316	4.56	8.64	10.11	12.81	14.34	28.26
<i>Trisopterus esmarki</i>	3.708	26.87	58.42	68.70	80.79	89.75	104.9
<i>Pollachius virens</i>	0.000	0.00	0.00	0.00	0.00	0.00	0.00
<i>Clupea harengus</i>	0.000	2.25	17.06	29.37	42.77	50.78	80.59
<i>Sprattus sprattus</i>	0.000	7.29	20.27	22.30	24.06	26.24	23.51
Ammodytidae	1.833	41.16	50.79	52.17	56.81	59.57	58.08
Microstomus kitt	0.000	0.00	0.00	0.00	0.00	0.00	0.00
<i>Solea solea</i>	0.000	0.00	0.00	0.00	0.00	0.00	0.00
<i>Limanda limanda</i>	0.000	1.21	0.48	0.00	0.00	0.00	0.00
Other fish	3.037	15.03	15.99	15.69	16.55	16.91	14.75
Annelida	0.173	5.72	7.56	7.65	7.84	7.99	6.61
Mollusca	0.094	0.46	0.29	0.24	0.27	0.27	0.24
Crustacea	3.972	22.68	19.28	16.75	16.30	15.74	12.29
Other prey	0.263	0.95	1.07	0.99	1.01	0.98	0.76
Total food ration (g)	13.49	131.8	210.7	236.3	273.1	298.2	346.3

Table 5.2.1 (continued). Quarterly food ration of a North Sea whiting by age and quarter of 1991.

QUARTER 4							
PREY	age 0	age 1	age 2	age 3	age 4	age 5	age 6+
<i>Gadus morhua</i>	0.029	0.70	0.41	0.12	0.16	0.11	0.11
<i>Melanogrammus aeglefinus</i>	1.484	2.24	4.18	6.86	7.64	10.56	9.78
<i>Merlangius merlangus</i>	0.201	2.29	5.12	8.67	8.84	10.39	10.40
<i>Trisopterus esmarki</i>	14.919	76.57	89.47	104.22	111.38	124.01	117.34
<i>Pollachius virens</i>	0.000	0.00	0.00	0.00	0.00	0.00	0.00
<i>Clupea harengus</i>	0.029	1.93	4.28	7.55	7.64	9.40	9.42
<i>Sprattus sprattus</i>	1.952	13.98	22.24	28.60	30.24	30.89	30.82
Ammodytidae	2.390	16.56	22.89	27.53	29.10	29.70	28.73
Microstomus kitt	0.000	0.00	0.00	0.00	0.00	0.00	0.00
<i>Solea solea</i>	0.057	0.05	0.23	0.46	0.47	0.60	0.62
<i>Limanda limanda</i>	0.107	0.00	0.61	0.88	0.88	1.14	1.14
Other fish	5.781	23.69	25.17	27.07	28.87	30.29	28.89
Annelida	0.791	4.91	5.84	7.08	7.77	9.62	8.45
Mollusca	1.726	0.49	0.57	0.75	0.80	0.85	0.90
Crustacea	11.373	23.92	23.49	22.98	24.72	25.52	24.17
Other prey	0.160	0.08	0.29	0.63	0.62	0.82	0.79
Total food ration (g)	41.00	167.4	204.8	243.4	259.1	283.9	271.6

Table 5.2.2. Quarterly food ration of a North Sea saithe by age and quarter of 1991.

QUARTER 1

PREY	age 3	age 4	age 5	age 6	age 7	age 8	age 9+
<i>Trisopterus esmarki</i>	140.1	164.1	149.6	235.8	298.6	353.3	66.3
<i>Melanogrammus aeglefinus</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Merlangius merlangus</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Clupea harengus</i>	3.1	111.8	480.6	593.4	724.0	869.3	3479.6
Ammodytidae	1.7	3.5	5.0	3.2	1.1	35.7	24.8
Other fish	58.0	98.8	127.5	160.4	158.6	159.4	1197.3
Euphausiacea	23.7	27.7	18.5	26.3	22.1	11.7	0.0
Other Crustaceans	14.2	15.8	18.1	31.4	35.1	31.9	3.8
Mollusca	45.1	9.0	2.6	1.5	0.5	0.0	0.0
Other prey	1.4	2.6	0.8	0.1	0.0	0.0	0.0
Total food ration (g)	287.3	433.3	802.8	1052.0	1239.9	1461.2	4771.8

QUARTER 2

PREY	age 3	age 4	age 5	age 6	age 7	age 8	age 9+
<i>Trisopterus esmarki</i>	116.6	247.2	673.6	855.1	837.0	720.8	614.9
<i>Melanogrammus aeglefinus</i>	0.1	0.7	2.6	1.0	175.0	530.0	779.2
<i>Merlangius merlangus</i>	0.0	0.0	0.2	1.7	134.6	401.8	1340.8
<i>Clupea harengus</i>	0.0	2.8	92.2	410.0	1031.4	1919.4	3232.1
Ammodytidae	97.4	144.9	300.6	238.3	179.8	340.4	413.7
Other fish	140.1	189.1	396.8	947.3	1347.2	1419.5	1604.0
Euphausiacea	407.9	474.6	508.5	910.2	1038.6	787.0	599.5
Other Crustaceans	68.2	50.3	30.1	16.2	11.9	15.3	16.3
Echinodermata	0.0	0.1	0.4	0.0	0.0	0.0	0.0
Other prey	0.0	0.0	0.2	0.0	0.0	0.0	0.0
Total food ration (g)	830.2	1109.6	2005.2	3379.7	4755.4	6134.2	8600.6

Table 5.2.2 (continued). Quarterly food ration of a North Sea saithe by age and quarter of 1991.

QUARTER 3

PREY	age 3	age 4	age 5	age 6	age 7	age 8	age 9+
<i>Trisopterus esmarki</i>	246.2	574.2	734.0	708.6	423.1	458.7	526.2
<i>Melanogrammus aeglefinus</i>	146.9	104.4	59.0	39.8	30.9	36.4	52.1
<i>Merlangius merlangus</i>	27.7	28.3	34.6	40.2	5.3	3.4	3.5
<i>Clupea harengus</i>	0.6	21.9	94.8	243.6	2217.3	2912.2	4663.0
Ammodytidae	51.4	41.7	19.4	0.0	2.1	3.0	4.7
Other fish	97.0	148.2	140.2	87.4	214.5	275.7	442.5
Euphausiacea	55.7	26.8	29.0	41.5	76.1	97.7	161.0
Other Crustaceans	3.1	19.6	27.4	21.7	1.5	0.0	0.0
Echinodermata	0.3	0.2	0.2	0.4	0.3	0.4	0.6
Other prey	0.1	0.1	0.0	0.0	0.0	0.0	0.0
Total food ration (g)	628.9	965.4	1138.6	1183.1	2971.1	3787.5	5853.6

QUARTER 4

PREY	age 3	age 4	age 5	age 6	age 7	age 8	age 9+
<i>Trisopterus esmarki</i>	538.8	815.5	841.0	582.2	330.4	267.8	403.9
<i>Melanogrammus aeglefinus</i>	58.2	59.7	52.9	105.9	98.3	43.6	76.8
<i>Merlangius merlangus</i>	9.4	12.2	7.9	20.7	20.0	6.7	21.1
<i>Clupea harengus</i>	2.9	54.5	645.1	1261.4	2140.0	2883.0	4272.6
Ammodytidae	0.1	0.2	0.0	0.0	0.0	0.0	0.0
Other fish	66.4	95.1	154.0	200.6	185.7	147.5	239.8
Euphausiacea	2.8	5.6	5.8	8.3	7.0	3.0	2.5
Other Crustaceans	1.2	1.4	1.9	0.9	0.0	0.0	1.0
Echinodermata	0.5	1.3	1.5	3.1	2.2	0.3	0.0
Other prey	0.7	3.0	1.0	0.0	0.0	0.0	0.0
Total food ration (g)	726.1	1048.4	1711.1	2183.1	2783.5	3351.9	5017.7

Table 5.3. Quarterly mean temperatures of major predatory fishes in the North Sea

Age/size class (y,cm)	Temperature (°C)			
	Q1	Q2	Q3	Q4
<i>Saithe</i>				
0	---	---	7.4	9.4
1	8.9	7.1	6.8	7.3
2	6.5	6.6	7.0	6.4
3	6.8	6.6	6.9	6.3
4	6.8	6.5	6.7	6.2
5	6.8	6.0	6.9	6.0
6	6.6	5.8	7.1	6.0
7+	6.8	6.6	6.9	6.3
<i>Whiting</i>				
0	---	---	8.2	8.9
1	5.8	7.7	11.5	9.8
2	6.7	7.3	10.1	9.4
3	6.7	7.3	9.5	8.9
4	6.2	7.5	10.2	9.5
5	6.3	7.4	10.2	9.5
6+	6.1	7.2	9.6	9.3
<i>Haddock</i>				
0	---	---	7.2	7.7
1	6.1	7.0	8.4	8.9
2	6.6	7.1	8.5	9.3
3	6.4	7.1	8.7	9.1
4	6.4	7.0	8.1	9.2
5	6.5	6.9	8.4	9.3
6+	6.5	6.8	7.7	9.3
<i>Grey gurnard</i>				
<10	6.7	6.8	6.1	10.2
10-14	6.8	7.6	10.3	9.7
15-19	6.8	7.4	10.1	9.1
20-24	6.8	7.5	8.9	8.8
25-29	6.8	7.7	7.8	8.6
30-34	6.6	7.5	8.2	8.4
35-39	6.4	7.1	8.4	7.9
40-44	6.6	6.9	8.7	8.0

6 REVIEW TOPICS OF GEOGRAPHIC AREAS WHERE ICES MAY CONTRIBUTE TO WORK ON RARE MARINE FISH SPECIES

Rarity must also be considered in a broader applied context where there are conservation and management implications. While it is important to note that rarity is a *natural* phenomenon, rarity can also be *apparent* or *anthropogenically-induced* and distinguishing between all three forms will become increasingly important in this context.

Natural rarity should be evaluated at the level of the population and has two major components: numerical abundance and spatial extent of distribution. The latter aspect is clearly related to the concept of essential fish habitat, because accumulation within specific areas during particular life history stages (nurseries, migration corridors, spawning areas) increases the vulnerability of a species compared to one that may be numerically equally abundant but spread over large and heterogeneous areas (ubiquists).

Apparent rarity may have different causes, but is generally related to inadequate sampling of the population of a species: it is rarely recorded because one sees only the vagrants from a population outside the defined area or, and potentially worse, because only the odd individual belonging to a population that lives within the defined area in a habitat that can not be adequately sampled. This can easily lead to the wrong conclusions about rarity but also about their essential fish habitat. Furthermore, our perception of rarity is typically based on the number of observations of a species, which is strongly affected by sampling protocol. If the appropriate sampling method is used on the appropriate spatial and temporal scale, then a species may no longer be perceived as rare.

Fishes have certain suites of life history and ecological attributes that make them more susceptible to anthropogenic activities, including fisheries, and they may therefore be more prone to anthropogenically-induced rarity. These would include long-lived species (e.g. sturgeons and elasmobranchs), species with comparatively low reproductive output (e.g. elasmobranchs, syngnathids, zoarcids), species with restricted geographical ranges (e.g. endemics), species with critical habitats that have been and/or are currently impacted by various anthropogenic activities and other factors (e.g. habitat destruction, poor water quality, non-native species). Additionally few species are abundant, indeed the superabundance historically typical of many commercially exploited species could be considered exceptional. Species that are intrinsically rare may well exhibit adaptations to enable their persistence at low densities or population sizes. There is no reason why 'superabundant' species should exhibit adaptations that would enable their persistence in the face of artificial anthropogenic-induced rarity. Such species may be particularly prone to compensatory mechanisms that may hinder their recovery from exploitation.

OSPAR define rarity as the following: "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".

Essentially, rarity is a relative concept that depends on both intrinsic characteristics related to the biology of the species involved in any comparison and external conditions during the collection of observations as well as on the spatial scale at which rarity is considered. Consequently, any index of rarity must take into account the perspective from which particular data sets have been collected. Furthermore in the marine environment where almost all comparisons of abundance among species are based on sampling gear, where species-specific catchabilities critically affect the results obtained. Moreover, use of sampling gear is often spatially restricted to specific physical conditions (such as "trawlable bottom"). Therefore, marine scientists are seriously hampered in evaluating absolute differences in rarity among species.

Rabinowitz (1981) described seven categories of rarity based on species abundance, geographical distribution and habitat specificity for plants. These categories were:

- Wide range, broad habitat specificity, local population size somewhere large.
- Wide range, broad habitat specificity, local population size small everywhere.
- Wide range, restricted habitat specificity, large populations in habitat.
- Small range, broad habitat specificity, populations somewhere large.
- Small range, restricted habitats, populations somewhere large.

- Small range, broad habitat, small populations.
- Small range, restricted habitats, small populations

As yet, the utility of such a categorisation scheme has not been explored for marine fishes.

Currently, ICES member states have sampling programmes throughout much of the OSPAR area, including the Norwegian Sea, North Sea, north-west Scotland, Celtic Sea, Bay of Biscay and Iberian coastal waters, in addition to surveys in the north-west Atlantic. Although many of these surveys operate in offshore waters (>10m depth), some member states also have surveys in estuarine and coastal waters.

The appropriate way forward with fish communities seems to be to integrate as many sources of survey information over as wide areas as possible (Dann, 2001), because this will reduce the chance of assigning the wrong rarity classification. However, this type of approach demands scrutinizing all individual databases against potential sources of error. Moreover, an analysis of relative catchabilities of a variety of species to different gears would greatly help standardization of information from different sources. In particular this would allow estimates of relative abundance be raised to absolute abundance estimates, which would facilitate greater comparability of abundance / rarity across the OSPAR region. This is a huge task that would require a major effort of the working group and could not be started at this first meeting.

It is likely that requests for advice on rare fish will increase in relation to the work of for a such as OSPAR, IUCN and a variety of environmental NGOs. Areas in which ICES could potentially contribute to work on rare fish:

- Improve taxonomic identification of non-target fishes on ICES groundfish surveys and ensure that all data are of high quality.
- Collate national data sets to improve our knowledge of the distribution, relative abundance and habitat requirements of rare and unusual marine fish species.
- Encourage member states to continue with surveys in a variety of habitats, from estuarine to deep water, in order to provide detailed spatial coverage of the OSPAR and ICES regions.
- Inventory of surveys by geographical area, time, gear, habitat.
- Analysis of relative catchabilities of a variety of species to different gears
- Review the methods of assessing rarity, including geographical range size, abundance and habitat specialisation, and evaluate which methods would be most suitable method (including spatial scale) with which currently available data could be utilised.
- Explore the consequences of anthropogenically-induced rarity for previously 'superabundant species', e.g. depensatory mechanisms.
- Review evidence for the existence of depensation in fishes, in light of increased data availability.
- Apply the Texel-Faial criteria (i.e. global importance, local importance, rarity, sensitivity, decline, keystone species) for fish species in order to prioritise those species for which more detailed assessments may be required.
- Compile an atlas of fish occurrence / abundance in the OSPAR region
- Geographical areas and habitats which would benefit from dedicated surveys include: Azores (possible endemics), Arctic / Northern waters (e.g. Spitzbergen), estuarine and coastal areas, rocky / coarse habitats, deep-sea / shelf edge.
- Explore the implications of climate change for OSPAR endemic / rare species, particularly for species in more northerly waters.

- Explore projected climatic influences on the coupling of pelagic and demersal fish assemblages, particularly with reference to rarity.

References

- Daan, N. (2001). A spatial and temporal diversity index taking into account species rarity, with an application to the North Sea fish community. ICES CM 2001/t:04, 15pp.
- Rabinowitz, D. (1981). Seven forms of rarity. In *The Biological Aspects of Rare Plant Conservation*. (H. Synge, ed.), John Wiley and Sons. New York, NY, 205-217.

7 RECOMMENDATIONS

Issues regarding threatened fishes, fish communities and many other aspects of fish ecology are increasingly asked of ICES. Hence, it was considered that WGFE should continue. Furthermore, the following potential work areas for WGFE were considered:

- Further studies to develop EcoQOs for fish communities are required. Such work could include (i) increased understanding of the theoretical basis of size-structured indicators, (ii) exploratory analyses of trophic level-size spectra, (iii) abundance-range size relationships; (iv) explore the utility and application of EcoQOs over a range of spatial scales.
- Completion of studies on the food rations, prey composition and gastric evacuation rates and gadoids.
- Explore the nature, types and consequences of rarity in the fishes of the OSPAR region.
- To assess our knowledge of habitat requirements of fish species. Whereas some species (e.g. whiting) appear to be ubiquitous on a variety of grounds, other species (e.g. small-eyed ray) have more restricted distributions.
- Studies focusing on the most critical habitats (e.g. the concept of *Habitat Areas of Particular Concern*) could be assessed for rare and threatened fish species, including diadromous species.

ANNEX

Working Group on Fish Ecology

ICES, Headquarters, 3-7 March 2003

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