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1 OPENING OF THE MEETING

The Working Group on Ecosystem Effects of Fishing Activities (WGECO) met at ICES Headquarters, Copenhagen, from 14–22 April 2004. The list of participants and contact details are given in Annex 1.

The Working Group members were welcomed by ICES Fisheries Adviser, Hans Lassen and the General Secretary, David Griffith. The Terms of Reference for the WGECO meeting were discussed on the first morning, and a plan of work was adopted for the meeting. Special efforts were made to make the fullest progress possible on the Terms of Reference directly supporting the advisory tasks of ICES, and to provide appropriate recommendations for the further development of work in support of those Terms of Reference where the process of arriving at a solution was begun.

Terms of Reference for the meeting were:

2ACEEC The **Working Group on Ecosystem Effects of Fishing Activities** [WGECO] (Chair: C. Frid, UK) will meet at ICES Headquarters from 14–21 April 2004 to:

a) for the EcoQO relating to spawning stock biomass of North Sea commercial fish species, and taking account of current reference points used in ICES advice and the outcome of the work of the Study Group on the Further Development of the Precautionary Approach to Fishery Management, to be used as baselines against which progress can be measured [OSPAR 2004/1]:

   i) reconsider the formulation of the EcoQO, determine whether a more specific EcoQO is needed in terms of its specification to the metric, time and geographical area, and as necessary propose (a) more specific EcoQO(s);

b) continue development, on the basis of the criteria for sound EcoQOs established by ICES in 2001, of related metrics, objectives and reference levels for the EcoQOs relating to the local availability in the North Sea of sandeels for black-legged kittiwakes, based on the output of WGSE, and reconsider the formulation of the EcoQO, determine whether a more specific EcoQO is needed in terms of its specification to the metric, time and geographical area, and as necessary propose (a) more specific EcoQO(s) [OSPAR 2004/1];

c) continue the development, on the basis of the criteria for sound EcoQOs established by ICES in 2001, of related metrics, objectives and reference levels for the EcoQOs relating to (l) changes in the proportion of large fish and hence the average weight and average maximum length of the fish community, based on input from WGFE and Assessment Working Groups; (o) density of sensitive (e.g., fragile) species, and (p) density of opportunistic species, based on input from SGSOBS; and (b) presence and extent of threatened and declining species in the North Sea [OSPAR 2004/1]. In this respect,

   i) for EcoQ element (l), taking into account all potential sources of relevant information, determine what information it will be possible to collect in future to assess whether the EcoQO is being met (taking into account practicability and costs), and develop draft guidelines, including monitoring protocols and assessment methods, for evaluating the status of, and compliance with, those EcoQOs,

   ii) for EcoQ elements (o) and (p), identify possible species in the respective categories, consider further the spatial scale requirements of sampling and the adequacy of existing monitoring activities to determine their status and trends, and provide further basis for advice based on scenario considerations on the applications of possible EcoQOs,

   iii) for EcoQ element (b), consider the invertebrate and fish species and the habitats on the Draft OSPAR list of threatened and declining species for their relevance and usefulness as a basis for EcoQOs for the North Sea,

   iv) where possible and appropriate, reconstruct the historic trajectory of the metrics and determine their historic performance (hit, miss or false alarm) relative to the objective being measured, as a basis for evaluating their relationship to management;

d) begin consideration of the means by which ecosystem considerations can be incrementally added to the ICES advisory framework with specific consideration of the approaches adopted by the existing advisory committees;

e) commence development of the scientific components of the framework and guidelines for the consideration of multiple EcoQO's as integrated sets for use in applied contexts;

f) complete the work started in 2003 in response to the EC request on ecosystem impacts of industrial fishing:

i) summarise information from relevant Expert Groups (Assessment Working Groups, SGDBI, WGFE) and prepare a compilation of the scientific information in response to this request,

ii) consider which aspects of this request require further work and propose plans to take forward such work;

g) consider a framework for the monitoring of the status of ecosystem components in the ICES area that makes use of both "descriptive surveillance metrics" and "performance metrics". The developed framework should include a consideration of how data routinely collected as part of ICES activities can be most effectively utilised for the purpose of reporting on ecosystem status, and what measures might ultimately be added to the current incomplete suite of EcoQOs (performance metrics) currently being developed;

h) review data on ecosystem responses to spatial reductions in fishing activities in temperate freshwater and marine areas, and describe similarities and differences in the biological development in these areas. Particular considerations should be given to differences in the ecosystem development in response to the geographical position/scale of the studied areas and our understanding of meta-population dynamics. Review published guidelines for the establishment of marine protected areas and recommend revisions;

i) consider the existing frameworks for assessing the role of habitats in support of biological diversity and the provision of "essential" habitat elements for key life history stages and review any existing measures of "habitat quality". Based on these analyses consider how this EcoQO element can be advanced;

j) start preparations to summarise the effects of fishing on North Sea biota for the period 2000-2004, and any trends in these effects over the recent decades.

WGECO will report by 3 May 2004 for the attention of ACE and the Marine Habitat, Living Resources, and Resource Management Committees.

Acknowledgements
The Working Group gratefully acknowledges the support given to us by Bengt Sjöstrand, of the Baltic Fisheries Assessment Working Group, who kindly provided information for Section 8, and Steve Murawski who provided invaluable information and expertise to Section 10.2. We also thank regular WGECO member Ellen Kenchington, who while unable to attend in person, contributed support and text by e-mail. The Working Group would also like to thank Marianne Neldeberg and Wim Panhorst for assistance, patience, and good humour in supporting our computing, system networking, and data requirements and Bodil Chemnitz for general logistical support and untiring assistance in a diversity of areas. On a personal level, this is my third meeting as Chair of WGECO and the tradition is that I now hand on to another. As this is my last report as Chair, I would like to express my sincere and deepest appreciation and thanks to all the members of WGECO over the last three meetings. Their hard work, often while being bombarded with multi-demands, and intellectual efforts have ensured that WGECO continues to produce material that is responsive to the needs of ICES advisory customers and contributes to the advancement of the discipline. I would also acknowledge the help and support I have had from the other members of the ICES 'community'.

2 EXECUTIVE SUMMARY

The terms of reference contained a mix of requests for work in support of advice, intellectual development of work-plans to underpin future requests for advice and original research. The Chair would like to record his sincere thanks to all the members of WGECO for their hard work and commitment both during the meeting and in preparing for it.

In 2001, 2002, and 2003 WGECO invested considerable effort in the development of the EcoQO framework and consideration of how various aspects of the approach adopted by the Bergen Conference could be made operational. In **Section 3** we return to this issue and consider the EcoQO relating to the spawning stock biomass of North Sea commercial stocks. Based on this consideration we propose a revision to the proposed EcoQO element such that "…above precautionary reference points for spawning stock biomass and below precautionary reference points for fishing mortality, for commercial fish species where these have been agreed to by the competent authority for fisheries management". In addition we begin a consideration of the possible effects of multi-species interactions on these EcoQOs and how environmental fluctuations might be incorporated in due course.

Section 4 continues the consideration of the appropriateness and potential effectiveness of the EcoQ element relating to the local availability of sandeels based on black-legged kittiwake breeding success. This work is a development of work started in WGSE and by WGECO in 2003. Based on a number of considerations and the results of our analyses WGECO proposed that this EcoQO be reformulated as "…black-legged kittiwake breeding success should exceed (as a three-year running mean) 0.6 chicks per nest per year in each of the following coastal segments: Shetland, north Scotland, east Scotland, east England".

Ecological Quality Objectives were also the focus of **Section 5**. Here we considered the EcoQ elements proposed but not forming part of the North Sea pilot project. We considered metrics covering the size of fish, the density of sensitive and opportunistic benthos, habitat quality and "threatened and declining species". In respect of the fish community metric, we have considered and extended the work of WGFE and WGECO2003 on this topic. The concerns expressed in 2003 about our present ability to determine reference levels remain, but further analysis has confirmed the existence of a relationship between weight and mean maximum length of fish. However, we conclude that these metrics will be poor performance metrics and should only be used as surveillance metrics. Consideration of the work of SGSOBS on selecting appropriate species, further analysis of spatial data of benthic communities, and an examination of the ability of a time series of benthic community data to inform decision control rules led us to conclude that the proposed EcoQ element concerned with opportunistic benthos (p) should be dropped as all the candidate species are too ubiquitous and respond to such a wide range of natural and anthropogenic perturbations. With respect to the EcoQ element (o) density of sensitive (fragile) species, we reconsidered our scenarios from 2003 and concluded that there are candidate species that should allow this element to be made operational using a limited number of sentinel species. We would encourage that other, specialist, groups be tasked with considering some of the statistical and practical considerations of this. In considering the EcoQ element concerned with Threatened and Declining Species and Habitats we remain very concerned at the difficulties of obtaining data or designing sampling programmes with sufficient statistical power. After careful consideration, we believe that this Ecological Objective would be best served by a metric formulated along the lines of the proportion of listed species/habitats for which a recovery plan had been prepared and implemented, with the EcoQO being 100% adoption. Such a formulation would alter the focus from single-species conservation to the more strategic consideration of conservation action.

ICES is currently undergoing a major restructuring and is taking the opportunity to also revise its advisory procedures; our ToR (d) gave us the opportunity to consider how ecosystem considerations might be most effectively introduced to this framework. In **Section 6** we make a number of suggestions as to how the advice might be best produced and then presented. We also consider how WGECO can continue to play a role in this process and reviewed possible tasks we might be called upon to do.

Section 7 WGECO has been considering the ecosystem impacts of fisheries for over ten years and has been involved from the outset in the development of what is now referred to as the "ecosystem-based approach to management". The Ecological Quality Objective framework is a major initiative aimed at making this operational. As such, the EcoQOs are seen not individually but as a suite and while we, and many others, have in recent years dedicated considerable intellectual effort to the development of individual EcoQOs, this was our first consideration of the need to make multiple EcoQOs work together. We have commenced this process by developing a framework, based on the DPSIR framework and our own views of the criteria for a good EcoQO, for assessing EcoQOs and mapping on their possible metrics. We believe that this will be a fruitful way of formally assessing where a suite of metrics will, or will not, function together. We were unable to complete this process in the time available but have progressed to the stage where others can see how we envisage it operating and we would welcome feedback from other parties before we take it further.

Industrial fisheries in European waters take very large tonnage of biomass from the sea. Approximately half of the fish biomass that is harvested and landed in the North Sea is of sandeels, and the blue whiting fishery in deeper waters off north-western Europe has an even larger harvest. In **Section 8** we complete the consideration of the impact of industrial fisheries on the ecosystem that we began in 2003. In particular, we revisit our consideration of the blue whiting fishery in the light of new information, document the scale of the industrial fishery in the Baltic and highlight the paucity of information on the ecosystem effects of this large fishery. We also consider the vexed question of whether it is ecologically more efficient to harvest small fish, such as sandeels, process them to meal and oil and feed this to aquacultured stock, or to leave them in the sea and harvest their natural predators. Our results suggest that there is surprising little difference in the ecological efficiency of the two routes. There are a number of other concerns both ecological and social that impinge on this issue. What is clear is that if we did not harvest the sandeels, the forgone harvest would not all be turned into fish flesh that we might harvest directly.

In **Section 9** we develop the framework for the use of 'descriptive surveillance metrics' and 'diagnostic performance metrics' for the monitoring of the status of ecosystem components. The framework developed comprised a three-stage monitoring framework for ecosystem status: surveillance monitoring, diagnosis, and post-intervention monitoring. WGECO also developed a list of eight ecosystem components and assessed the availability of data for assessing these within this framework. We conclude that there is an excess of data with which to evaluate current status for most components, but there is a shortfall in the availability of reliable performance metrics to aid diagnosis of environmental problems and to assess the management regime's effectiveness.

In previous years, we have referred to the value of marine protected areas as part of a management regime controlling the ecosystem effects of fishing. **Section 10** provides a review of the ecosystem response to spatial reductions in fishing (closed areas) in temperate waters drawing upon six case studies. These show that in two of the six cases examined, poor incorporation of science at the planning stage seriously undermined the effectiveness of the management measure. We went on to review some of the available guidelines for MPA establishment. While we recognise the value of a consistent standard to the science base of such guidelines, the need for social, cultural, and economic considerations to be included means that no single framework is ever likely to be universally applicable. Furthermore, these same constraints mean that while ICES can, and should, provide the science framework for a North Atlantic MPA framework, this must be developed in partnership with other groups having the necessary social, cultural, and economic expertise.

WGECO have previously identified that the biggest impediment to the development of the EcoQO to maintain or restore habitat quality was the lack of a definition of habitat quality. In **Section 11** we consider the existing legal frameworks and scientific protocols of assessing the need to protect habitats and determine their quality. From this we develop a possible framework for advancing this EcoQO. This involves firstly applying a classification regime to the habitat types and maps of their location. Following this the statutory instruments and the IUCN guidelines will ensure protection of all of the area for the designated habitat types. We believe that in dealing with the remainder parallels will emerge with the WFD's need to improve ecological status.

WGECO have previously discussed the merits of regional ecosystem management and are therefore keen to contribute to the development of this framework for advice with ICES. In **Section 12** we begin the task of preparing to make a formal assessment of the impacts of fishing on the North Sea biota for 2000–2004. We formally reviewed the aspects of the environment that need to be considered and then list the information required to make the assessment; we do this in a manner that will hopefully assist our sister expert groups in providing us with the appropriate data when the time comes.

During the course of our work, a number of other issues have emerged that we feel warrant noting in the report; these are described in our *Food for Thought* chapter (**Section 13**). We have also extracted from the body of the text our specific recommendations for future work and development of procedures or data acquisition. These are documented in **Section 14** *Recommendation for Future Activities*. In addition to these broad recommendations, many of our Sections also include specific recommendations for advancing those particular areas of work and these are cross referenced here.

3 TOR A) FOR THE ECOQO RELATING TO SPAWNING STOCK BIOMASS OF NORTH SEA COMMERCIAL FISH SPECIES, AND TAKING ACCOUNT OF CURRENT REFERENCE POINTS USED IN ICES ADVICE AND THE OUTCOME OF THE WORK OF THE STUDY GROUP ON THE FURTHER DEVELOPMENT OF THE PRECAUTIONARY APPROACH TO FISHERY MANAGEMENT, TO BE USED AS BASELINES AGAINST WHICH PROGRESS CAN BE MEASURED [OSPAR 2004/1]

i) reconsider the formulation of the EcoQO, determine whether a more specific EcoQO is needed in terms of its specification to the metric, time and geographical area, and

ii) as necessary propose (a) more specific EcoQO(s).

3.1 Examination of the EcoQO

Last year WGECO conducted an analysis of the performance of $\mathbf{B}_{pa}$ and $\mathbf{F}_{pa}$ as guides to setting TACs, using the approach of signal detection theory (ICES, 2003a). We found that, aside from industrial fisheries, error rates in performance of the metric were between 40% and 50% for all stocks tested, and that Misses and False Alarms were equally frequent (ICES 2003a, Section 3). We concluded that the results suggested that advice based on SSB and F will not recommend catch reductions when in fact they are needed for about one stock in five. However, advice based on SSB and F relative to their reference points would recommend unnecessary catch reductions about equally often. Such performance by a metric used to support management would be seen if estimates of SSB and F have some uncertainty, and managers were using the PA reference points as targets, successfully keeping stocks, on average, at $\mathbf{B}_{pa}$ and fishing mortality, on average, at $\mathbf{F}_{pa}$. ICES advice clearly labels the PA reference points as boundary conditions to be avoided with high probability, but the stock dynamics suggest that is not the case in practice.

This symmetry in error rate *de facto* treats both types of errors (Misses and False alarms) as equally undesirable. In this particular EcoQO, False Alarms are more ecologically precautionary than Misses, so from the perspective of conservation, it would be desirable to reduce the Miss rate. This could be done by choosing different positions for the reference points (higher $\mathbf{B}_{pa}$, lower $\mathbf{F}_{pa}$), but with present knowledge this is likely to increase the False Alarm rate, and advice for unnecessary catch reductions will be given more often. This may not please all customers of the advice. The alternative of management keeping SSB well above $\mathbf{B}_{pa}$ and F well below would also reduce the error rate, through having many more Hits (stocks *estimated* to be above $\mathbf{B}_{pa}$ and actually *being* above $\mathbf{B}_{pa}$). This alternative would also be consistent with the uses intended for the precautionary reference points when ICES proposed them.

Last year's analysis treated each annual assessment and stock advice on fisheries management actions as an independent event. In practice, the assessment and advisory process has some self-correction built into its recurrent pattern. Given normal stock dynamics, a Miss in one year is going to result in a greater discrepancy between the *estimated* SSB and F in the following year, and the corresponding values of $\mathbf{B}_{pa}$ and $\mathbf{F}_{pa}$. With larger discrepancies, the likelihood of a Hit is greater in the second year (a reduction in F and an increase in SSB is needed, and the advice recommends lower harvests), so errors will not compound over time. The same correction is built into the system response to False Alarms. Compared to simply following ICES advice and avoiding the PA reference points with high probability, our results show that current assessment and management are expending a great deal of effort chasing noise in the assessment, but the reference points do seem to be a sound basis for management advice.

ACFM has not yet changed reference points for stocks on which it provides harvest advice, so there was no reason to repeat the analyses conducted last year. In the context of good EcoQOs, we concluded last year that "the wording of the EcoQO be modified slightly. Rather than 'spawning stock biomass *also taking into account fishing mortality, …*' [italics ours], the EcoQO should explicitly include both properties. The EcoQO should be based on the proportion of stocks where SSB>$\mathbf{B}_{pa}$ **and** F < $\mathbf{F}_{pa.}$ both are fulfilled.

> The revised EcoQO would state **'Above precautionary reference points for spawning biomass and below precautionary reference points for fishing mortality, for commercial fish species where these have been agreed to by the competent authority for fisheries management.'**

The existing management approaches for individual stocks are all based on an assumption that the objective of management is to move SSB above $\mathbf{B}_{pa}$ and to keep fishing mortality sustainable. The EcoQO would simply condense this information into a form that gives an appropriate overview of the overall status of North Sea fish stocks." We also included a reminder that good fisheries management consists of much more than sustainable use of the target species.

Last year's results were examined intersessionally by working group members and by both ACFM and ACE. Nothing was found to change our views on the results of the analysis, and our conclusions on the utility of SSB as a foundation for EcoQOs remains. We have identified some additional considerations, however. One consideration is whether there is one EcoQO on spawning stock biomass of commercial fish species, or if there are as many EcoQOs as there are assessed stocks. This is addressed in Sections 7 and 9 of this report. Another consideration is that at present for the large majority of stocks, $\mathbf{B}_{pa}$ and $\mathbf{F}_{pa}$ are both set on the basis of single-species stock dynamics assuming that all population dynamics parameters have constant values over time. Although our analysis suggests that this assumption has not caused serious problems yet, the time frame of our analysis extends back only to the late 1980s. An ecosystem approach to management encourages taking a long-term view of conservation and sustainable use, and in this context it may be important to consider species interactions and environmental forcing in selecting reference points. A preliminary discussion of some of these considerations is included in Sections 3.2 and 3.3, respectively.

3.2 Species interactions

The WGECO evaluation of $\mathbf{B}_{pa}$ and $\mathbf{F}_{pa}$ as EcoQOs has been performed on the single-species stock assessment reported by ACFM. As mentioned earlier, these assessments are prone to errors, some of which are analysed and reported. In the proposed framework for revising ICES precautionary reference points (by the Study Group on the Further Development of the Precautionary Approach to Fishery Management (SGPA) (ICES, 2003b)), the key element is the $\mathbf{B}_{lim}$, from which other reference points are derived. $\mathbf{F}_{lim}$ is derived as the F which leads to an equilibrium SSB at $\mathbf{B}_{lim}$. The $\mathbf{B}_{pa}$ and $\mathbf{F}_{pa}$ are derived taking assessment error into account. The problem with retrospective bias and its possible influence on PA reference points needs careful analysis however, and the revision of reference points is still in progress in various fisheries assessment working groups.

Species interactions may affect the estimates and dynamics of the stocks in a complex manner. These interactions are at least a major source of uncertainty and may possibly result in inaccurate estimates of reference points. In the fisheries context there are only a limited number of operational models that account for important trophic links among fish species in a given ecosystem. To evaluate the degree to which the formulation of fisheries EcoQOs would depend on the multispecies interactions of the commercial fish stocks, WGECO revisited recent reporting on single-species reference points treated in multispecies contexts.

The Study Group on Multispecies Assessments in the North Sea (SGMSNS) (ICES, 2003c) considered to what extent reference points derived within a single-species framework are valid when multispecies interactions are taken into account. In the North Sea MSVPA, the only multispecies interaction is predation mortality. The group argued that was far beyond what could currently be achieved to propose a full set of reference points based on multispecies assessment, and it therefore focused on limit points as being the basis for other reference points.

SGMSNS highlighted that the link between $\mathbf{B}_{lim}$ and $\mathbf{F}_{lim}$ would need to be revisited in a multispecies context, because:

1) The equilibrium F corresponding to a given SSB is no longer unique, because it depends on the state of the other stocks in the system;
2) When F-values have been specified for all species, there is equilibrium with a unique set of SSB values. The opposite may not be true: several possible combinations of F values may lead to a specific set of SSB values.

Consequently, in a multispecies context, no unique determination of F corresponds to the derivation of $\mathbf{F}_{lim}$ from $\mathbf{B}_{lim}$, as currently used in the single-species based advice. Rather, $\mathbf{F}_{lim}$ depends on how other stocks are exploited. In the multispecies setting, the joined limits SSB>$\mathbf{B}_{lim}$ for all species translates into a multidimensional parameter-space for F. The estimation of the boundaries of this space is not straightforward for several reasons, and would be extremely difficult to communicate. Setting limits to potential exploitation scenarios was considered beyond the competence of the SGMSNS at their 2003 meeting.

Because the equilibrium biomass at a given fishing mortality for a prey species would depend on the exploitation and, therefore, the abundance assumed for the other species, both predators and prey, $\mathbf{F}_{lim}$ values derived as proposed by ICES SGPA are conditional on the exploitation regime in the system as a whole, and might need to be revised if that changed. The validity of the $\mathbf{F}_{lim}$ values derived from $\mathbf{B}_{lim}$ in a single-species framework depends on how well the assumed natural mortalities represent the actual state of the system.

Addressing a limited set of specific questions, and limiting the discussion to reference points, SGMSNS performed updated MSVPA runs for the North Sea in order to study the sensitivity of SSB to the introduction of predation mortality at selected F-values ($\mathbf{F}_{lim}$, $F_{status\ quo}$, F_{1960s}) in a long-term equilibrium. The calculations were made assuming that future recruitment would be constant, and at arithmetic mean of the past. The group argued that this assumption would be justified by the requirement that $\mathbf{B}_{\mathrm{lim}}$ should be such that “recruitment was not impaired”. Consequently, only the effect of changed natural mortalities on the equilibrium was analysed.

For F at currently adopted $\mathbf{F}_{\mathrm{lim}}$ (or substitutes for undefined values) for all stocks, the difference between single-species and multispecies results was not great, although SGMSNS noted that the equilibrium SSB was somewhat lower for most species in the multispecies framework. The other F-regimes simulated lead to clear differences between single- and multi-species scenarios, but SGMSNS concluded that these differences would not lead to drastically different conclusions in qualitative terms.

With the exception of one species (haddock), the overall conclusion of SGMSNS from the comparison of single- and multi-species predictions of three selected management scenarios was that currently effective $\mathbf{F}_{\mathrm{lim}}$ values were adequate to ensure $\mathbf{B}_{\mathrm{lim}}$ (assuming that recruitment was not impaired at $\mathbf{B}_{\mathrm{lim}}$), i.e., that the M values used in single-species assessment adequately represent predation mortalities for the current situation. For haddock, the multispecies model predicted a collapse in SSB, due to predation by saithe.

Similarly, Gislason (1999) found considerable differences of multispecies reference points (derived from MSVPA) compared to single-species reference points from the cod-sprat-herring subsystem in the central Baltic Sea. Like WGMSNS, he noted that multispecies reference limits for forage fish cannot be defined without considering changes in the biomass of their natural predators, and vice versa. He emphasised the necessity to explicitly include socio-economic considerations into management objectives, in order to address trade-offs in multispecies scenarios.

Walters *et al*. (2004) tested multispecies harvest rates of eleven marine Ecosim case models, including the North Sea, to predict equilibrium yields. Their results indicate that fishing all species at MSY would have severe ecosystem impacts. In their modelling results, changes in the harvest rate for any one species showed asymmetric effects on other trophic levels. They found that (a) top-down effects tended to be strong (i.e., the prey of the species in question were liable to become much more productive); and (b) bottom-up effects tended to be weak (predators of the species in question may find other food sources if it declines).

Specifically for the North Sea, the modelling by Walters *et al*. (2004) found that the ecosystem MSY was lower than the single-species MSY for most groups. The authors found that total ecosystem yield and landed value were generally not predictable from the sum of single-species assessment predictions. They remained particularly uncomfortable about predictions of “bottom-up” effects. They concluded that there is apparently no easy way to balance the three basic objectives of productivity, diversity, and stability.

The results of Walters *et al*. (2004) are in line with the recommendations from the Johannesburg WSSD (UN, 2002) that $\mathbf{F}_{\mathrm{MSY}}$ should be considered a default for a **limit** reference point, i.e., that management should avoid Fs as high as $\mathbf{F}_{\mathrm{MSY}}$ with high probability. ICES did not use $\mathbf{F}_{\mathrm{MSY}}$ as a major guide to setting limit reference points in the original round, but WGECO notes a growing number of studies which suggest that F should not be set higher than $\mathbf{F}_{\mathrm{MSY}}$. WGECO hopes that this matter will be dealt with fully in the planned revision of the reference points used in fisheries advice.

In conclusion, WGECO noted that several multispecies models exist and results of preliminary analyses are available. The models continue to be under development, and their results cannot yet easily be made fully operational as a basis for fisheries advice. WGECO encourages continued and rapid progress in further development of these models. WGECO has reviewed such models in the past. We repeat that although the class of multispecies models representing predator-prey dynamics have an important role in providing a sound basis for fisheries advice, it is very important that the factors which influence the performance of such models be well understood, and be fully acknowledged in the advice.

Notwithstanding the need for further development of these models, and in line with the conclusions of Stefánsson (2003), **the available results from multispecies modelling studies suggest that biomass reference points are likely to be higher, and fishing mortality reference points more variable, when derived in a multispecies context. This implies that any achievable targets in a multispecies context will be more restrictive than the corresponding targets based on single-species assessment alone**.

3.3 Long-term variability in environmental forcing of stock dynamics

SSB reference points have been aimed at maintaining a low risk that the spawning biomass would fall below a conservation limit set by the dependence of recruitment on SSB. The ICES advice is then risk-averse, i.e., it aims at keeping the risk of harming the productivity of the stocks below a safe limit. Biological target reference points are also part of the Precautionary Approach (FAO, 1996a, 1996b), but setting targets for fisheries management involves identifying desired socio-economic considerations. Therefore, ICES cannot propose values for Target Reference Points until Management Agencies have identified management objectives based on socio-economic benefits. ICES is in the process of revising reference points and a new framework which includes yield-related reference points will be developed and introduced (ACFM October 2003).

Under the current framework, the SSB reference points assume stable productivity dynamics and meet specified conditions regarding what comprises "serious harm". Hence they are only expected to change slightly due to improvement of biological data and time series updating. However, there is substantial evidence that productivity dynamics of stocks are not stationary in the long term. There are many stocks where climatic conditions play a major role in stock productivity. In at least some cases, there is evidence that the system can shift to a new set of oceanographic and climatic conditions in which the previously estimated reference points no longer meet the defining conditions. The idea of regimes in climatic, oceanic, and biological systems was first introduced to fishery scientists by Isaacs (1975), who called these persistent trends *regimes*. These have been defined as multi-year periods of linked recruitment patterns in fish populations or as stable conditions in physical data series (Schwartzlose *et al*., 1999; Beamish *et al*., 1999, 2000).

Evidence for Decadal-scale Variation in the Ocean Environment

In the Northeast Atlantic, this type of work is in progress. For the North Sea, Reid *et al*. (2001) identified a regime shift occurring about 1988 and O'Brien *et al*. (2000) found yield-per-recruit functional relationships whose parameters differed according to the sea temperature and corresponded to different periods of time. Interannual changes in the North Sea cod recruitment have been shown to be related to changes in sea surface temperature (SST) (Brander, 1996; Planque and Fox, 1998; Planque and Fredou, 1999). The relationship is negative with increased February to June SSTs resulting in reduced recruitment to the stock. Fluctuations in water temperature also have a secondary continuous effect on cod individuals, through their influence on growth rates and weight at age (ICES, 2002).

Brander and Mohn (in press) used the North Atlantic Oscillation (NAO) during winter (December through March) as a proxy for temperature, wind, and precipitation during the next Spring. They examined the NAO-index effect on the stock-recruitment relationship fitted to thirteen North Atlantic cod stocks using Ricker stock-recruitment relationship. Strongly negative effects were found for the relationship between NAO and recruitment occurring in the North Sea, Baltic Sea, and Irish Sea. They also examined the geographical pattern of the effects of the NAO on SST, finding that the relationship between NAO and SST is strongly positive. They concluded that, for the North Sea cod where the effect of NAO is strong, for medium- and long-term assessments of recruitment and yield of the cod stocks, NAO should be considered as an explicit variable. Although the NAO index has the advantage of being operational because of the substantial research on climatology, the disadvantage is that the mechanisms of its impact on each geographical system and species domain are not always clear and, therefore, demand that *in situ* research is timely in place to monitor fish biological vital rates.

Similarly, the size of the water body with salinity, temperature, and oxygen content conducive to egg survival of central Baltic cod (the "reproductive volume") has been shown to be an important covariate in explaining recruitment time series for this species. The size of the reproductive volume is hypothesized to have changed with different conditions of salt-water inflow to the Baltic Sea, creating two separate environmental "states" to which separate recruitment models can be fitted better than one model can be fit to all the data (Jarre-Teichmann *et al*., 2000). Köster and Möllmann (2000) additionally emphasize the intraspecific predation on early life stages as a factor stabilizing species dominance in the central Baltic Sea.

For the fish stocks in the Barents Sea, the impact of long-term climatic oscillations is discussed by Yndestad (2004).

In the Pacific Ocean, climatic regime shifts are well defined (Benson and Trites, 2002). Many studies have found low frequency, high amplitude, and sometimes abrupt changes in species abundance, community composition, and trophic organization, which occur concurrently with physical changes in the climate system. These are considered indicative of a regime shift (Beamish *et al*., 1999, 2000; Alexander *et al*., 2001). Studies from the North Pacific indicate that regime shifts can have opposite effects on species living in different domains, or can affect similar species living within a single domain in opposite ways (Benson and Trites, 2002).

The importance of these patterns over very large spatial scales was shown by Klyashtorin (1998). He found simultaneous oscillations of the main commercial species in the Atlantic and Pacific, and in sub-tropic, sub-Arctic, and Arctic zones—including herring, cod, sardines, anchovy, and a number of other species. Although surface air temperature was too variable to correlate with these oscillations, an index of atmospheric circulation was closely related to long-term fluctuations in the main commercial stocks.

For climate regimes and their effects on stock dynamics to be evaluated, it is necessary that available time series be long, ideally spanning at least two shifts and a transitional period. In addition, in the ideal condition, the mechanisms and processes underlying the shifts have to be explainable with functional relationships over time in order to be possible to guide the fisheries management strategies with some operational time delay. These ideal conditions will rarely be met, although modelling to explore scenarios can help to address these problems to some extent.

Notwithstanding the scientific challenges of rigorously documenting the patterns and causal linkages of regime changes, one of the goals of medium- and long-term fisheries management is to minimize the risk of harm to stock productivity. Given that recruitment, growth, and survival rate and, hence, productivity can be affected by environmental factors, it raises the question of whether advice on achieving medium-term and long-term goals should also take the longer-term environmental forcing into account.

Some fisheries management agencies have tried to address these regime shifts in their management. There are examples where fishing mortality was maintained very low in the recruitment-unfavourable regimes and allowed to increase in the recruitment-favourable regimes (e.g., management of the South African small pelagics fishery, de Oliveira *et al*., 1998), ideally with some time delay in the transitional period. Success of such a strategy depends on having operational environmental indicators, which can detect the transitional period between regimes and guide harvested control rules. Experience with these management regimes is preliminary, but users feel that such indicators are species-specific, must be adjusted to the specific domain, and cannot be used in other species or domains even in the same geographical area (see also Bograd *et al*. (2004)).

Work in this area is developing rapidly. Although it is premature to leap immediately to putting long-term environmental factors into all reference points and EcoQOs regarding fish stocks, it is important that work commence within ICES to take greater account of the effects of longer-term environmental variation on stock dynamics. In particular, with current knowledge:

1) In the short term, estimation of biological reference points needs to consider environmental forcing. Work from SGPRISM and SGGROMAT provides some relevant guidance, but more work focused specifically on reference point estimation is needed.

2) Work is needed urgently to determine whether and exactly how reference points for SSB, fishing mortality, and other parameters should be estimated and used, when there is evidence of long-term, regime-scale environmental forcing of stock dynamics.

3) Many environmental factors affecting recruitment, growth or survivorship may show long-term directional trends. We are aware of very little work on whether and how reference points should vary with gradients in environmental conditions, or of the risks associated with failing to take account of such gradients. Given the evidence for the effects of climate change, and changes in species composition of North Sea (and other) fish communities, commencement of such work should also be a priority.

References

Alexander, V., Bychkov, A., Livingston, P., and McKinnel, S.M. (eds.). 2001. Proceedings of the PICES/CoML/IPRC Workshop on Impact of Climate Variability on Observation and Prediction of Ecosystem and Biodiversity Changes in the North Pacific. PICES Science Report, No. 18. 215 pp. Available at http://www.pices.int/publications/scientific_reports/Report18

Beamish, R.J., Noakes, D.J., McFarlane,G. A.Klyastorin L., Ivanov, V.V., and Kurashov, V., 1999. The regime concept and natural trends in the production of Pacific salmon. Canadian Journal of Fisheries and Aquatic Sciences, 56: 516–526.

Beamish, R.J., McFarlane, G.A., and King, J.R. 2000. Fisheries Climatology: Understanding Decadal Scale Processes That Naturally Regulate British Columbia Fish Populations. Fisheries Oceanography: An integrative approach to fisheries ecology and management. Fish and Aquatic Resources Series, 4: 94–139.

Benson, A.J., and Trites, A.W. 2002. Ecological effects of regime shifts in the Bering Sea and eastern North Pacific Ocean. Fish and Fisheries, 3: 95–113.

Bograd, S.J., Schwing, F.B., Mendelssohn, R., and Palacios, D.M. 2004. The morphology and dynamics of North Pacific climate regimes. Paper presented at the SCOR/IOC Symposium on Quantitative Ecosystem Indicators for Fisheries Management, 31 March–04 April 2004, Paris, France.

Brander, K.M. 1995. The effect of temperature on growth of Atlantic Cod (*Gadus morhua).* ICES Journal of Marine Science, 52:1-10.

Brander, K. 1996. Effects of climate change on cod (*Gadus morhua*) stocks. *In* Global warming: Implications for freshwater and marine fish, pp.255–278. Ed. by C.M. Wood and D.G. McDonald. Cambridge University Press.

Brander, K., and Mohn, R.(accepted). Effect of the North Atlantic Oscillation (NAO) on recruitment of Atlantic cod (*Gadus morhua*). Canadian Journal of Fisheries Aquatic Sciences.

de Oliveira, J.A.A., Butterworth, D.S., Roel, B.A., Cochrane, K.L., and Brown, J.P. 1998. The application of a management procedure to regulate the direct and bycatch fishery of South African sardine *Sardinops sagax*. South African Journal of Marine Science, 19: 449–469.

FAO. 1996a. The precautionary approach to marine capture fisheries and species introductions. FAO Technical Guidelines for Responsible Fishing, 2. 73 pp.

FAO. 1996b. Precautionary approach to fisheries. Part 2. Scientific Papers. FAO Technical Paper 350. 210 pp.

Gislason, H. 1999. Single and multispecies reference points for Baltic fish stocks. ICES Journal of Marine Science, 56: 571–583.

ICES. 2000. Report of the ICES Advisory Committee on Fishery Management, 2000. ICES Cooperative Research Report, 242.

ICES. 2002. Report of the Study Group on the Incorporation of the Process Information into Stock/Recruitment Models. ICES CM 2002/C:02.

ICES. 2003a. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 2003/ACE:05.

ICES. 2003b. Report of the Study Group on the Further Development of the Precautionary Approach to Fishery Management. ICES CM 2003/ACFM:09.

ICES. 2003c. Report of the Study Group on Multispecies Assessments in the North Sea. ICES CM 2003/D:09.

Jarre-Teichmann, A., Wieland, K., MacKenzie, B., Hinrichsen, H.-H., Plikshs, M., and Aro, E. 2000. Stock-Recruitment relationships for cod (*Gadus morhua callarias*) in the central Baltic Sea incorporating environmental variability. Archive of Fishery and Marine Research, 48(2): 97–123.

Klyashtorin, L.B. 1998. Long-term climate change and main commercial fish production in the Atlantic and Pacific. Fisheries Research, 37(1–3): 115–125.

Köster, F.-W., and Möllmann, C. 2000. Trophodynamic control on recruitment success in Baltic cod? ICES Journal of Marine Science, 57: 310–323.

O’ Brien, C.M., Fox,C.J., Planque, B., and Casey, J. 2000. Climate variability and North Sea cod. Nature, 404: 142.

Planque, B., and Fox, C.J. 1998. Interannual variability in temperature and the recruitment of Irish Sea cod. Marine Ecology Progress Series, 172: 101–105.

Planque B., and Fredou, T. 1999. Temperature and the recruitment of Atlantic cod (*Gadus morhua*). Canadian Journal of Fisheries and Aquatic Sciences, 56: 2069–2077.

Reid, P.C., Borges, M.F., and Svendsen, E. 2001. A regime shift in the North Sea circa 1988 linked to changes in the North Sea horse mackerel fishery. Fisheries Research, 50: 163-171.

Schwartzlose, R.A., J. Alheit A. Bakun, T.R. Baumgartner, R. Cloete, R.J.M. Crawford, W.J. Fletcher, Y. Green-Ruiz, E. Hagen, T. Kawasaki, D. Lluch-Belda, S.E. Lluch-Cota, A.D. MacCall, Y. Matsuura, M.O. Nevarez-Martinez, R.H. Parrish, C. Roy, R. Serra, K.V. Shust, M.N. Ward and J.Z. Zuzunaga. 1999. Worldwide large-scale fluctuations of sardine and anchovy populations. South African Journal of Marine Science, 21: 289–347.

Sparholt, H. 1996. Causal correlation between recruitment and spawning stock size of central Baltic cod? ICES Journal of Marine Science 53(5): 771-779.

Stefánsson, G. 2002. Multi-species and ecosystem models. *In* Responsible fisheries in the marine ecosystem. Ed. by M. Sinclair and G. Valdimarsson. CAB International.

United Nations. 2002. Plan of Implementation – World Summit on Sustainable Development. New York. 44 pp

Walters, C., V. Christensen, S. Martell and J. Kitchell. MS, 2004. Single species versus ecosystem harvest management: Is it true that good single species management would result in good ecosystem management? Paper presented at the SCOR/IOC Symposium on Quantitative Ecosystem Indicators for Fisheries Management, 31 March – 04 April 2004, Paris, France.

Yndestad, H. 2004. The cause of Barents Sea biomass dynamics. Journal of Marine Systems, 44(1–2): 107–124.

4 FURTHER DEVELOPMENT OF THE ECOQO RELATING TO THE BREEDING PRODUCTIVITY OF BLACK-LEGGED KITTIWAKES AS AN INDEX FOR THE LOCAL AVAILABILITY OF SANDEELS IN THE NORTH SEA (TOR B)

The term of reference states:

Continue development, on the basis of the criteria for sound EcoQOs established by ICES in 2001, of related metrics, objectives and reference levels for the EcoQOs relating to the local availability in the North Sea of sandeels for black-legged kittiwakes, based on the output of WGSE, and reconsider the formulation of the EcoQO, determine whether a more specific EcoQO is needed in terms of its specification to the metric, time and geographical area, and as necessary propose (a) more specific EcoQO(s) [OSPAR 2004/1].

4.1 Assessment of the metric against the ICES Criteria

4.1.1 Previous progress

The background to this EcoQO has been well documented in previous WGSE and WGECO reports (ICES 2003a, 2003b, 2004). Based on the results of black-legged kittiwake population dynamics modelling, WGSE suggested in 2001, and repeated in 2003, an EcoQO target for mean black-legged kittiwake chick productivity in each year of at least 0.5 chicks per nest in all relevant areas of the North Sea. The concept of breeding success, expressed as the number of chicks per nest, is easily understood by non-scientists and it is an index that is easily assessed by trained personnel. Such monitoring data are available for six separate coastal regions, which when combined account for the majority of breeding black-legged kittiwakes feeding in the North Sea, and for most of these areas, these data have been collected for nearly two decades. Thus, this metric scores well against four of the criteria for sound EcoQOs established by ICES in 2001: criteria a, d, f, and g. However, WGSE felt that the metric performed less well when assessed against the remaining three criteria. Realising the importance of the environmental influence on variation in the abundance of small pelagic fish, WGSE pointed out that changes in pelagic fish abundance were only partially driven by fishing activity. Thus, they concluded that the metric would not be tightly linked to fishing activity (Criterion c), and may not necessarily be particularly responsive to fisheries management action (Criteria b and e). Despite this, WGSE (2003a) concluded that black-legged kittiwake breeding productivity was sensitive to changes in the abundance of their food supply (principally sandeels). WGECO (2003b), with reference to a case study in the Wee Bankie area off the southeast coast of Scotland, examined the robustness of this conclusion.

Prior to 1991, there had been little industrial fishing activity off the southeast coast of Scotland, but in 1991 a fishery started and in 1993 over 100,000 t of sandeels was taken in the Wee Bankie/Marr Bank area. Catches were lower in subsequent years, but still amounted to over 30,000 t in most years. Simultaneously with the growth of the fishery, black-legged kittiwake breeding productivity at colonies in the Firth of Forth and elsewhere down the east coast of Scotland declined markedly. The correlation between black-legged kittiwake breeding productivity and sandeel abundance suggested that the fishery may have had a negative effect on black-legged kittiwake breeding productivity in 1993 (Rindorf *et al*., 2000). In 2000, a precautionary approach was taken and sandeel fishing off the east coast of Scotland was stopped from 2000 to 2002, except for a small commercial monitoring fishery required to continue the collection of data used to evaluate the sandeel population abundance in the area. The three-year closure was extended in 2002. Black-legged kittiwake breeding productivity increased immediately following the closure of the fishery in 2000, appearing to confirm the causal link. However, in 2001 and 2002, despite acoustic surveys in the area indicating a continued and marked increase in the local abundance of sandeels, black-legged kittiwake breeding success once again declined. Black-legged kittiwake diets on the Isle of May, the largest black-legged kittiwake colony in the Firth of Forth and the closest to the Wee Bankie, consist of 1+ aged sandeels in May, but in June and July they switch almost entirely to 0-group sandeels (Wanless *et al*., 2002). The fishery in the area was prosecuted in June in most years and almost exclusively targeted 1+ sandeels (Pedersen *et al*., 1999). Consequently, WGECO (2003b) considered that the fishery and black-legged kittiwakes rarely targeted the same resource at the same time and place. 0-group sandeels were the principal prey fed to black-legged kittiwake chicks (Wanless *et al*., 2002), and the fishery would rarely have been responsible for a direct reduction in the abundance of this prey resource. Furthermore, survey data available to WGECO indicated no obvious reduction in 0-group sandeel recruitment to the area in 2001, certainly nothing to explain the reduction in black-legged kittiwake breeding productivity. WGECO (2003b) concluded that factors affecting the availability of sandeels to black-legged kittiwakes tended to decouple the link between sandeel abundance and black-legged kittiwake breeding productivity.

4.1.2 Recent progress and future requirements

WGSE (ICES, 2004) re-examined the relationship between sandeel abundance and black-legged kittiwake breeding productivity. As noted in ICES (2001), Furness (1999) had correlated black-legged kittiwake productivity with sandeel abundance as derived from VPA-based stock assessments. WGSE (ICES, 2004) found that this relationship held in subsequent years around Shetland and, in addition, a relationship was found with a trawl survey-based index of sandeel abundance in the area. Both the VPA- and the survey-based indices were strongly influenced by the abundance of 1+ aged sandeels. When black-legged kittiwake breeding productivity along the southeast coast of Scotland was examined in relation to a trawl-based index of 1+ aged sandeel abundance in the Wee Bankie area, rather than the acoustic survey-derived abundance estimate used by WGECO (2003b), which is strongly influenced by 0-group abundance, a highly significant correlation was obtained. Approximately 90% of the variation in black-legged kittiwake breeding productivity in this region of the North Sea could be explained by local variation in the abundance of one-year old and older sandeels. The evidence presented by WGSE (ICES, 2004), therefore, makes a compelling argument that, even in the Wee Bankie region, variation in black-legged kittiwake breeding productivity can provide an indication of local variation in the abundance of their fish prey, in this case specifically one-year old and older sandeels.

If there is a need to redress fully the shortcomings of this metric with respect to criteria b, c, and e, the underlying mechanism(s) linking black-legged kittiwake breeding productivity and 1+ aged sandeel abundance need to be elucidated. Why should black-legged kittiwake breeding productivity be so strongly determined by variation in the abundance of 1+ aged sandeels when 0-group sandeels form the major part of the chicks' diets? Clearly the abundance of 1+ aged sandeels early in the season is the critical factor. Several hypotheses may be put forward. For example, if the 1+ sandeel food resource is insufficient early in the season, then adult black-legged kittiwakes may fail to achieve adequate body condition to even initiate a breeding attempt and breeding fails at the egg-laying stage. Under such circumstances, variation in the abundance of 1+ sandeels might act as a "switch", turning breeding productivity "on" or "off". Alternatively, variation in adult black-legged kittiwake body condition during the chick-rearing stage, dependent upon 1+ aged sandeel abundance earlier on in the season, may determine the allocation of effort between chick provisioning and adult maintenance. Such a mechanism might predict that failure occurs during the chick-rearing stage. Testing these and other possible alternative hypotheses should help to clearly identify the underlying process by which 1+ aged sandeel abundance affects black-legged kittiwake breeding productivity.

The rapid increase in sandeel abundance, indicated by CPUE data and both the demersal trawl and acoustic survey data, following the closure of the sandeel fishery would seem to confirm the effect of the fishery on local 1+ aged sandeel abundance. Consequently, clearly establishing the mechanistic link between 1+aged sandeel abundance early in the breeding season and subsequent black-legged kittiwake breeding productivity would strengthen the case that this metric is sensitive to a manageable human activity (criterion b), since the sandeel fishery primarily targets 1+ aged sandeels. Similarly, if 1+ sandeel abundance was to be critically reduced by a fishery at the same time that adult black-legged kittiwakes, building up to breeding condition, were utilising the same resource, then one might expect the metric to be tightly linked in time to the fishing activity (criterion c). Breeding productivity two months later on in the same season would be compromised. The performance of this metric with respect to criterion e would also be strengthened, but this would still remain the weakest aspect. Fishing activity can have a large effect on 1+ aged sandeel abundance, but it is not the only determining factor. Sandeel abundance is also strongly influenced by variation in recruitment and this is largely governed by environmental fluctuation.

Various methods of assessing the abundance of sandeels have been applied, e.g., acoustic surveys, demersal trawl surveys, nocturnal grab surveys, dredge surveys, VPA stock assessments, and commercial catch per unit effort (Harwood *et al*., 2000; Rindorf *et al*., 2000; Wright *et al*., 2002; ICES, 1993a, 2003b, 2004). The signals provided by these various methods are not always consistent due to the confounding effect of diel, seasonal, and annual variation in the behaviour of sandeels. Sandeels have the capacity to bury into the sediment and emerge to swim freely in the water column. They tend to bury during hours of darkness and for much of the year, emerging into the water column during the day in the late spring and early summer to feed. This behaviour is strongly controlled by local variation in environmental factors and strongly affects sandeel "detectability" by the various abundance estimation methods. For example, dredge and grab samples will not detect sandeels that are in the water column; conversely, acoustic surveys will not detect sandeels buried in the sediment. Environmental variation, both seasonal and diel, and between years, therefore influences the abundance estimates derived by the different survey methods. Data are now available for five different survey methods for the period 1997 to 2003, and the relationships between the different abundance assessment methods can now be examined. An examination of the differences between these methods may help to further define the relationship between black-legged kittiwake breeding productivity and sandeel abundance off the east coast of Scotland. Further, once a peer-reviewed method has been defined which can be used to measure changes in sandeel availability, a performance analysis of the indicator, i.e., the breeding success of black legged kittiwakes, can be carried out.

4.2 Defining an Ecological Quality Objective

WGSE (ICES, 2004) pointed out that the relationship between sandeel abundance and black-legged kittiwake breeding productivity cannot be linear over an unlimited range of prey abundance. Black-legged kittiwake breeding productivity is capped by the fact that black-legged kittiwake average clutch size is two eggs, of which some fail to develop while others are lost through natural processes. The data available to WGSE suggest that maximum productivity would rarely exceed 1.5 chicks per nest. After assessing the data, WGSE (ICES, 2004) concluded that "high" sandeel abundance would lead to black-legged kittiwake breeding performance in the range 0.7 to 1.2 chicks per nest, whereas "low" sandeel abundance results in breeding productivity in the range 0 to 0.5 chicks per nest. Consequently, WGSE suggested a revised wording for the EcoQO. After further consideration, WGECO consider that the EcoQO should be formulated as:

- "black-legged kittiwake breeding success should exceed (as a three-year running mean) 0.6 chicks per nest per year in each of the following coastal segments: Shetland, north Scotland, east Scotland, east England".

Limitation of the EcoQO to only the western coastline of the North Sea reflects three factors. Firstly, colonies on these coasts have been more extensively monitored over a longer period of time. Secondly, numbers of breeding black-legged kittiwakes are lower on the east coast of the North Sea, and thirdly, black-legged kittiwakes in the eastern North Sea appear to be less reliant on sandeels. A value of 0.6 chicks per nest was chosen because in most years the observed value would lie either well above, or well below, the objective level, thus minimising the scope for using uncertainty as an excuse for doing nothing. The use of a three-year running mean introduces some protection against false alarms. Thus, either one or two exceptionally low values or a series of three marginally low values would have to be recorded, before any mitigation action would be triggered. This introduces the possibility that an EcoQO formulated in this way may not be sufficiently sensitive to trigger mitigating management action quickly enough. However, seabirds are long-lived animals and it is unlikely that three poor breeding years would seriously affect local populations. An EcoQO for black-legged kittiwake breeding productivity formulated in this way would therefore seem to be a good compromise between the cost of responding to "false alarms" and the ecological consequences of failing to respond to "misses".

References

Furness, R. W. 1999. Does harvesting a million metric tonnes of sand lance per year from the North Sea threaten seabird populations? *In* Ecosystem Approaches for Fisheries Management, pp. 407–424. Alaska Sea Grant College Program, AK-SG-99-01, Fairbanks.

Harwood, J., Hall, A. J., McConnell, B., Pomeroy, P., Duck, C., Fedak, M., Matthiopoulos, J., Walton, M. J., Greenstreet, S. P. R., Emmerson, H. M. F., Armstrong, E., Gibb, I., Doyle, C. J., Wanless, S., Rindorf, A., Pedersen, S. A., Jensen, H., Davis, J., and Ligdas, C. N. 2000. Effects of large-scale industrial fisheries on non-target species (ELIFONTS) Report of EU project Nº 95/078. Sea Mammal Research Unit, St. Andrews.

ICES. 2001. Report of the Working Group on Seabird Ecology. ICES CM 2001/C:05. 72 pp.

ICES. 2003a. Report of the Working Group on Seabird Ecology. ICES CM 2003/C:03. 92 pp.

ICES. 2003b. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 2003/ACE:05. 206pp.

ICES. 2004. Report of the Working Group on Seabird Ecology. ICES CM 2004/C:05.

Pedersen, S.A., Lewy, P., and Wright, P.J. 1999. Assessments of lesser sandeel (*Ammodytes marinus*) in the North Sea based on revised stock divisions. Fisheries Research, 41: 221–241.

Rindorf, A., Wanless, S., and Harris, M.P. 2000. Effects of changes in sandeel availability on the reproductive output of seabirds. Marine Ecology Progress Series, 202: 241–252.

Wanless, S., Daunt, F., Wilson, L., Hamer, K., Humphreys, L., Hennicke, J., and Garthe, S. 2002. Seabird foraging performance and diet (colony based studies) summer 2002. Report 2002-006 of the IMPRESS Project. 23 pp.

Wright, P.J., Jensen, H., Mosegaard, H., Dalskov, J., and Wanless, S., 2002. European Commission's annual report on the impact of the Northeast sandeel fishery closure and status report on the monitoring fishery in 2000 and 2001.

5 DEVELOPMENT OF ECOQO RELATING TO (L) PROPORTION OF LARGE FISH, (O) DENSITY OF SENSITIVE AND OPPORTUNISTIC SPECIES, AND (B) PRESENCE OF THREATENED AND DECLINING SPECIES

5.1 Introduction

The TOR read as follows:

Continue the development, on the basis of the criteria for sound EcoQOs established by ICES in 2001, of related metrics, objectives and reference levels for the EcoQOs relating to (l) changes in the proportion of large fish and hence the average weight and average maximum length of the fish community, based on input from WGFE and Assessment Working Groups; (o) density of sensitive (e.g., fragile) species, and (p) density of opportunistic species, based on input from SGSOBS; and (b) presence and extent of threatened and declining species and habitats in the North Sea [OSPAR 2004/1]. In this respect,

i) for EcoQ element (l), taking into account all potential sources of relevant information, determine what information it will be possible to collect in future to assess whether the EcoQO is being met (taking into account practicability and costs), and develop draft guidelines, including monitoring protocols and assessment methods, for evaluating the status of, and compliance with, those EcoQOs,

ii) for EcoQ elements (o) and (p), identify possible species in the respective categories, consider further the spatial scale requirements of sampling and the adequacy of existing monitoring activities to determine their status and trends, and provide further basis for advice based on scenario considerations on the applications of possible EcoQOs,

iii) for EcoQ element (b), consider the invertebrate and fish species and the habitats on the Draft OSPAR list of threatened and declining species for their relevance and usefulness as a basis for EcoQOs for the North Sea,

iv) where possible and appropriate, reconstruct the historic trajectory of the metrics and determine their historic performance (hit, miss or false alarm) relative to the objective being measured, as a basis for evaluating their relationship to management;

This was a continuation of the TOR given to WGECO last year, and covered in section 5 of the report (ICES, 2003). By repeating it this year we have been able to give some more thought to developing metrics for these ecosystem components, and bring together some developments that have taken place in the preceding 12 months.

As requested, we have dealt with the three components of the TOR sequentially, making use of case studies to illustrate important points where necessary.

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

5.2.1 Introduction

ICES (2001a) proposed criteria for metrics that would make them suitable as indicators to support the Ecosystem Approach to Management (EAM). The indicators should be:

- relatively easy to understand by non-scientists and those who will decide on their use;
- based on an existing body or time-series of data to allow a realistic setting of objectives;
- measurable over a large proportion of the area in which the indicator is likely to be used;
- easily and accurately measured, with a low error rate;
- sensitive to a manageable human activity (e.g., fishing) and responsive primarily to that activity, with low responsiveness to other causes of change;
- relatively tightly linked in space and time to that activity.

Frid (2002) distinguished two types of metrics: environmental state (surveillance) metrics and decision support (performance) metrics (see Section 9). How these two types of metrics relate to the criteria was not defined but the suggestion is that the surveillance metrics only need to meet the first four criteria, while weak performance metrics should also meet the fifth criterion and strong performance metrics should meet all criteria.

Many of the fish community metrics that have been put forward (ICES, 2003a) meet the first four criteria and are thus potential surveillance metrics. The challenge is to identify to what extent the suite of fish community metrics meet the last two criteria and can therefore be used as performance metrics. Fish community metrics can only be used in a management context if these metrics are sensitive to fishing impacts and respond rapidly to management action, so that managers can assess whether changes in the fish community are a desirable or undesirable response to management. It should also be possible to estimate metrics with sufficient precision so that changes in the community can be detected on management time scales of a year to a few years. In addressing these points we will focus on what were considered the most promising fish community metrics: mean weight and mean maximum length.

5.2.2 Mean weight and mean maximum length: further exploration

The analyses presented in last year's WGECO report (Section 12: Further exploration of effects of fishing activities on fish assemblages and marine ecosystems, TOR j) suggest that mean weight and mean maximum length show a consistent response to fishing in areas that differ in fishing intensity and that the trends estimated from different surveys do not contradict. However, further analyses were considered necessary to assess whether the trends could be detected reliably on time frames relevant to management. The analyses of the changes of the two community metrics after closure of the plaice box did not show a result that is in accordance with theory: a decrease in effort in the box area should result in the downward trend becoming less negative in the box area while the trend in the reference area should become more negative as effort from the box area is reallocated to the reference area. Therefore, these community metrics are probably unsuitable to detect local responses of the community to spatial changes in fishing effort resulting from the establishment of a protected area. Thus, the concerns with these indicators are that the theoretical understanding of their response to fishing is not well developed, because the underlying spatial processes are not understood and they are influenced by both direct and indirect fishing effects.

The analyses done at WGFE (2004) confirmed the relationship with fishing activity and consistency between surveys, and further explored the direct and indirect fishing effects by conducting a detailed analysis of absolute rather than relative CPUE. This analysis showed that a widespread, long-term increase has occurred in the abundance both of small fish and of small species (low maximum length). In addition, the absolute abundance of large fish has declined significantly, but the reduction in large species (high maximum length) was not significant. 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 as it releases predation pressure on them.

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. 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. As different surveys cover different time periods and/or areas, a cross-calibration of surveys would allow an expansion of the spatial extent of coverage.

Nicholson and Jennings (2003) tested the power of a large-scale annual trawl survey (North Sea International Bottom Trawl Survey, IBTS) to detect trends in six community metrics, among them mean weight and mean maximum length. Their analyses showed that the power of the trawl survey to detect trends is generally poor. While the community metrics do provide good long-term indicators of changes in fish community structure, they are unlikely to provide an appropriate tool to support short-term management decisions. If fish community metrics are to provide effective support for ecosystem-based management, and management time scales cannot be extended, then the power of many surveys to detect trends in fish community structure will need to be improved by increased replication and standardization.

With current knowledge there is no ability to 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 a 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 that integrate a relatively quick and short-term direct effect on the large fish and a delayed and longer-term indirect effect on the small fish. In addition there may be an even longer-term (decadal) genetic effect. These effects are then 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.

In summary, it appears that there is a relationship with fisheries for metrics of both mean weight and mean maximum length of fish in the community. However, this relationship is not straightforward, not well understood and certainly not tightly linked in space and time. As such these two metrics will be poor performance metrics, and should preferably be used only for surveillance of the fish community.

5.3 EcoQ elements (o) and (p) density of sensitive (e.g., fragile) and opportunistic species

5.3.1 Introduction and developments since 2003

WGECO has provided extensive critiques of proposals for the development of EcoQOs for the benthic systems ((ICES, 2000, 2001b, 2002, 2003b). These have been made in order to influence positively the development of sound (i.e., matching the ICES (2001a) criteria) EcoQOs that can contribute to the implementation of the ecosystem approach to marine management. Following the consideration of these EcoQO elements by WGECO and the Benthic Ecology Working Group in 2003, ACE (2003) established a group, The Study Group on Ecological Quality Objectives (EcoQOs) for Sensitive and for Opportunistic Benthos Species (SGSOBS). SGSOBS was asked to specifically consider all aspects concerned with the development of the EcoQ elements (o) and (p).

SGSOBS met at ICES HQ from 22–24 March 2004 and we are very grateful to the Chair, K. Essink, for making a draft of their report available to us.

5.3.2 Identify possible species in the respective categories

SGSOBS used the following definitions:

> *Sensitive species* – A species easily depleted by human activity and, when affected, is expected to recover over a long period or not at all.

As such, the term "sensitivity" takes into account both the tolerance to and the time needed for recovery (largely species dependent) from the stressor. Fragile species are considered to be especially susceptible to physical/mechanical disturbance.

> *Opportunistic species* - Species (second- and first-order, based on Borja *et al.,* 2000, ecological groups IV and V) that follow the reproductive (*r*) strategy (*sensu* Pianka, 1970), with short life-cycle (<1 year), small size, rapid growth, early sexual maturity, planktonic larvae through the year, and direct development.

These species proliferate after intense disturbance or pollution episodes. Surface or subsurface deposit-feeders dominate.

In 2003 WGECO (ICES, 2003b), based on the data for the North Sea soft sedimentary environments provided by the NSBS database and our limited additions, recorded a total of 180 taxa as meeting the criteria for sensitive species, this includes biogenic structure-forming species as well as those with fragile morphological features, and 69 taxa as meeting the criteria for opportunists, this includes the opportunistic scavengers. WGECO considered this to be an initial and incomplete list. SGSOBS identified 242 sensitive species in genera beginning with the letter A alone and 54 taxa as first-order opportunistic species and 119 as second-order opportunistic species (i.e., 173 opportunistic taxa). As previously stated by WGECO and SGSOBS, there remains a massive literature and incomplete knowledge of many species such that these estimates still remain conservative. However, they further serve to illustrate the problems of attempting to manage benthic systems to achieve a metric based on the density of individual sensitive and/or opportunistic taxa (see Section 5.2.5 of ICES (2003b) and Section 9.3 of the 2004 SGSOBS report).

In **Section 5.3.3** we consider six "species" (some are at the genera level), three opportunistic and three sensitive. We examine how even starting with a base of six species, the number of EcoQOs that might need to be addressed increases rapidly as one seeks to use them in more biologically meaningful ways. In **Section 5.3.4** we use a case study, where ecological changes in the benthos have been demonstrated, to examine the ability of EcoQOs set for the density of individual sentinel species to respond to a number of potential operational management scenarios.

5.3.3 Consider further the spatial scale requirements of sampling and the adequacy of existing monitoring activities to determine their status and trends

This issue was addressed in Section 5.2.3 of the 2003 WGECO report (ICES, 2003b).

The design of benthic monitoring schemes will need to account for the close interaction between the physical habitat and the benthos it supports. Sampling the North Sea according to areas or grids which have no biological references, such as ICES rectangles, may not provide high quality information about population and distribution trends in the species to be monitored, as the underlying cause of species distribution is not addressed. However, this approach does simplify the monitoring process.

Here we consider how setting EcoQOs based on the same original data series will vary depending on the approach taken to the issue of unit or scale of application, i.e., whole North Sea, regional, or habitat based.

We selected three opportunistic and three sensitive species on the basis that they fit the criteria of sensitive and opportunistic species presented in Section 5 of the 2003 WGECO report and used in the OSPAR 2004 report, and that there were at least two other scientific journal papers to support their categorisation and no published contradictory evidence (Table 5.3.3.1).

The density and distribution of all the opportunistic species and *Spisula* were obtained from the 1986 North Sea Benthos Survey (NSBS; ICES 1997), which used grabs to obtain samples which were then sieved through a 1.0 mm mesh. The distributions of the other two sensitive species in the North Sea (*Arctica* and *Pennatula*) were taken from the Dutch BTS which used an 8-m beam trawl and 4-cm stretched mesh. The data from the Dutch survey only covered the southern North Sea.

The 2003 WGECO report discusses the difficulties associated with collecting different types of benthic fauna. The smaller opportunistic species are unlikely to be observed in the trawl samples and require sampling by grabs. The larger epifauna and deeper burrowing infauna are likely to be sampled more effectively using trawls and dredges.

The mean density and variance of the selected species were calculated at different scales across the North Sea. At the broadest scale, the mean density and variance of the opportunistic species and *Spisula* were calculated for the entire North Sea, whilst the mean density and variance of *Arctica* and *Pennatula* sensitive species were calculated across the entire southern North Sea (south of 56°N). The scale of the density estimates was then reduced, as the mean density and variance of opportunistic and sensitive species were calculated per quarter of the North Sea (divided into four quadrants NE, NE, SW and SE centred on 56°N 02°E) (Figure 5.3.3.1). The mean densities and variance of opportunistic species and *Spisula* were then calculated according to biologically relevant spatial scales. This was achieved for the opportunistic species by using the eight infaunal communities identified by Künitzer *et al*. (1992) (Figure 5.3.3.2).

Table 5.3.3.1. Species identified as "sentinel" species based on their performance against the criteria defined in Section 5.3.2, above, for sensitive species. (The term sentinel species is used as defined in Section 5.2.5.2 of ICES (2003b)).

SENSITIVE SPECIES	Easily depleted by human activity	Recovery over a long period or not at all	Fragile – sessile or slow moving	Fragile – rigid bodies or tubes	Fragile – sensitive to physical damage	Fragile- Adult Body size >2 cm	Fragile - Epifauna	Fragile – Sub-surface infauna	Reference
Pennatula phosphorea	N	Unknown	Y	N	Unknown	Y	Y	Unknown	Hughes, 1998; OSPAR, 2004; MacDonald *et al*., 1996.
Arctica islandica	Unknown	Unknown	Y	Y	Y	Y	Unknown	Y	Frid *et al*., 2000; ICES, 2004; Piet *et al*., 1998; Fonds, 1991; Klein and Whitbard, 1995; MacDonald *et al*., 1996; ICES, 2003b; ref Lis
Spisula spp.	Y	Unknown	Y	Y	Y	Unknown	Unknown	Y	Kröncke and Bergfeld, 2001; DeBoer *et al*., 2001; ICES, 2004.

Table 5.3.3.2: Species identified as "sentinel" species based on their performance against the criteria defined in Section 5.3.2 for opportunists. (The term sentinel species is used as defined in Section 5.2.5.2 of ICES (2003)). * = long distance disperser/ high reproductive rate.

OPPORTUNISTIC	Early maturation	High fecundity	High colonisation potential*	Opportunistic feeders	Mobile over scales of 10s of metres	References
Capitellidae	Y	Y	Y	N	N	Pearson and Rosenberg, 1978; ICES, 2003b; Bustos-Baez and Frid, 2003; Frid *et al*., 2000; MAFF, 1980
Cirratulus cirratus	Y	Y	Y	N	N	ICES, 2003b; Borja *et al*., 2000
Chaetozone setosa	Y	Y	Y	N	N	ICES, 2003b; Borja *et al*., 2000

Figure 5.3.3.1. Overview of North Sea stations and the positions of the chosen four regions. Stations inside 56.25°N and 61°N are characterised as Region 1 for -4°E and 2°E, and Region 2 for 2.5°E and 9°E. Stations within 51°N and 56°N are characterised as Region 3 for -4°E and 2°E, and Region 4 for 2.5° E and 9°E (Modified figure from ICES, 1997).

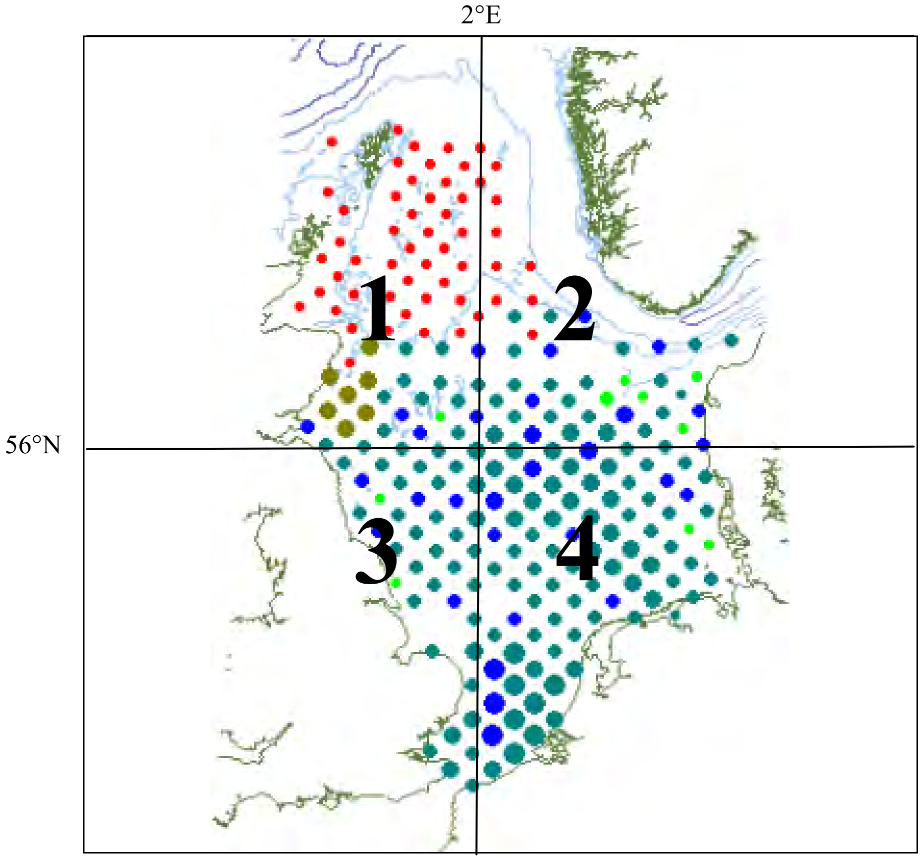
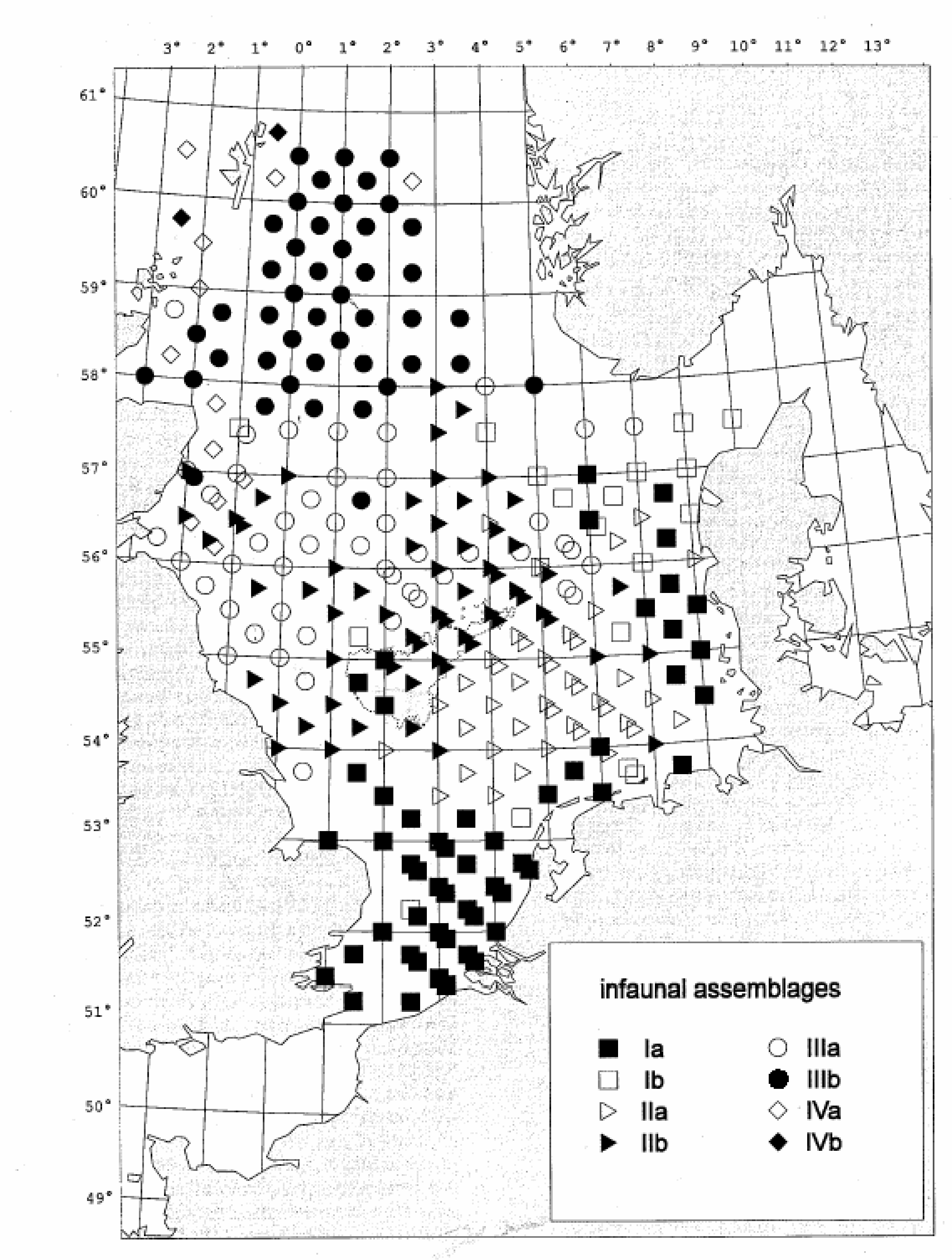


Figure 5.3.3.2. Overview of "infaunal assemblage groups" as defined in ICES (1997).



5.3.3.1 Spatial scale considerations

For the sensitive taxa sampled by beam trawl (*Arctica* and *Pennatula*), there are clear differences in abundance between the SW and SE North Sea such that an EcoQO developed on the overall North Sea or southern North Sea level would fail to account for this variation (Figure 5.3.3.1.1a-d). For *Spisula* (sampled by grab), there was little variation in the mean at the different levels of application of the EcoQO, but the variation was greatest at the North Sea scale (Figure 5.3.3.1.1e-f). This suggests that the power to detect changes in the metric would be least if it were applied at the North Sea scale.

For the opportunistic taxa (all sampled by grab), the means showed little difference as the data were aggregated to differing degrees (Figure 5.3.3.1.2). In two cases (*Chaetozone setosa* and Capitellidae), the variance was highest when data were considered at the North Sea scale, however for *Cirratulus cirratus* the variance did not vary greatly with changing scales (Figure 5.3.3.1.2 b, d, and f).

These analyses confirm the intuitive expectation that, if an EcoQO is applied over all sampling points in the North Sea, the variance will be higher and hence the survey power less to detect a change in the EcoQO metric of density. The corollary of this is that the number of EcoQOs must be increased. At the most biologically realistic scale, the assemblage types defined by the NSBS, there would be eight EcoQOs for each species with a North Sea-wide distribution; however, most species would not occur in all eight assemblage types. Interestingly for the four species we considered (one sensitive and three opportunistic), there was little additional benefit of moving from a coarse geographical division to the fully resolved assemblage level; however, the general interpretation of this result needs further consideration.

5.3.3.2 Statistical considerations

In considering the power of such temporal analyses, WGECO recognises that there are a number of statistical issues involved in identifying appropriate variance estimates. In most analyses comparing two surveys in different times, the variance obtained at each time point is actually the estimate of spatial variation around the mean density estimate. This is, in fact, not the appropriate variance estimate for temporal analysis. Only one temporal observation is made in each year, the mean density across the entire spatial unit. Thus with a sample size of 1, no true estimate of temporal variance can be ascertained. Comparing annual survey data in this way will require an alternative approach. Straightforward temporal trend analysis of annual means may be more appropriate, or one could perhaps aggregate years of data, and compare the means of five annual mean density estimates between two groups of five years of data, deriving the temporal variance from each set of five individual annual means. Alternatively, with the benefit of adequate mapping, appropriate spatial "units" for each metric could be identified, within which spatial variation might be considered to be minimal, or at least irrelevant to interpretation of the results. Variance derived from a number of samples obtained from such spatial units might therefore be considered to be estimates of the variance in each year, i.e., estimates of temporal variance that would be appropriate for temporal analysis.

Figure 5.3.3.1.1. Sensitive species.

Mean density and mean variance of (a, b) *Arctica islandica*, and (c, d) *Pennatula phosphorea*, in the southern North Sea and the SW and SE regions 56°N 03°E and mean density and mean variance of (e, f) *Spisula* spp, in the whole North Sea, Geo(graphic) zones (SW, SE, NW and NE regions, centred on 56°N 02°E), together with Zoo(geographic) zones (=infaunal assemblages in ICES, 1997).

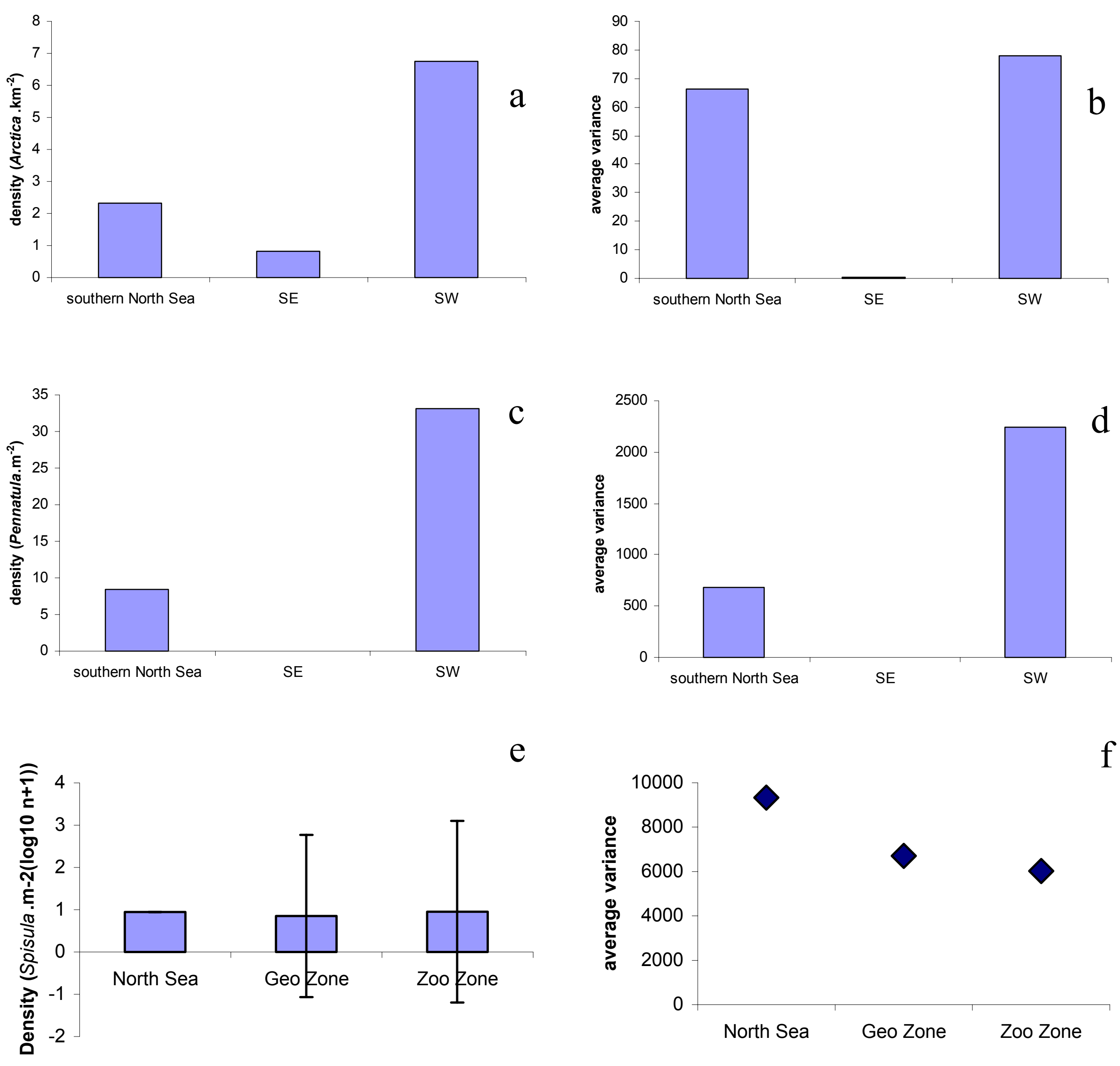
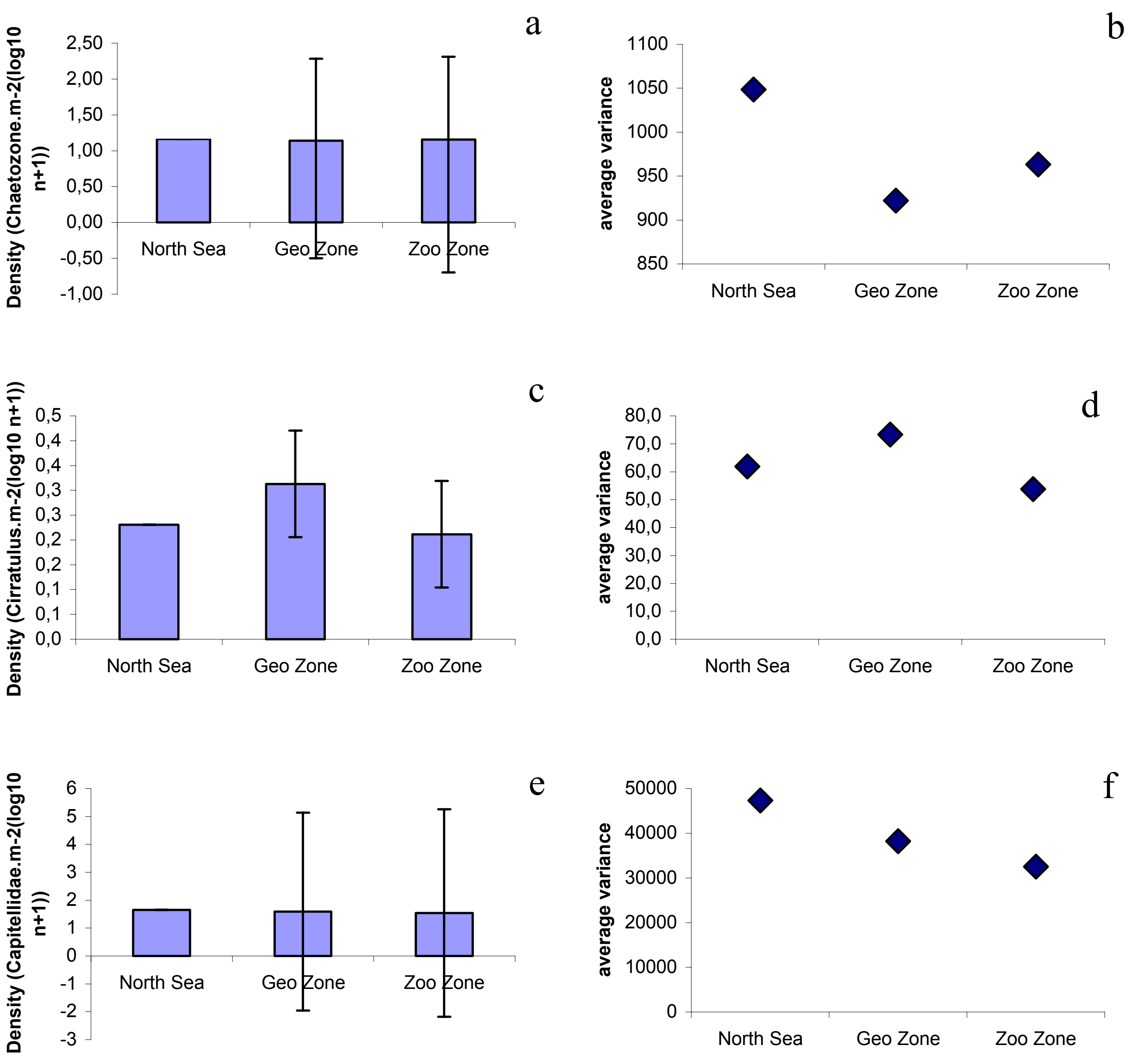


Figure 5.3.3.1.2. Opportunistic species.

Mean density and mean variance of (a, b) *Chaetozone setosa*, (c, d) *Cirratulus cirratus*, (e, f) Capitellidae, in the whole North Sea, Geo(graphic) zones (SW, SE, NW and NE regions, centred on 56°N 02°E), together with Zoo(geographic) zones (=infaunal assemblages in ICES, 1997).



5.3.4 Provide further basis for advice based on scenario considerations on the applications of possible EcoQOs

In 2003 WGECO examined five scenarios that use different possible EcoQO combinations covering these elements (see Text Box 5.3.4.1). The scenarios, in decreasing order of their information requirements were:

- Scenario 1: EcoQOs set for each species on the whole list and for the whole North Sea;
- Scenario 2: EcoQOs set for each species on the whole list and for each habitat/assemblage;
- Scenario 3: An index of opportunists or sensitivity;
- Scenario 4: The proportion of all species that are opportunists or sensitivity;
- Scenario 5: The density of a selection of sentinel species.

Text Box 5.3.4.1. The five scenarios considered for EcoQO elements (o) and (p) by WGECO in 2003.

Scenario 1: Whole list and whole North Sea

This approach is the most direct application of the proposed EcoQ elements, with the density of each sensitive/opportunistic species being monitored with the objective of maintaining each at some target level relative to a reference level. Our analysis of the ICES benthic database shows 179 species that fulfil the criteria for being considered under these two elements. Most of these species are restricted to certain assemblages (Table 5.1) and these, in turn, are associated with particular aspects of the physical benthic environment – habitats (Künitzer *et al.*, 1992). As such any EcoQO of North Sea density would have to be based on abundance of the species, weighted by the natural distribution of the different habitat types. Leaving aside WGECO's often repeated caution about the practicality of actively managing human activities to give a resultant abundance of benthos, this implies a massive workload.

Scenario 2: Whole list by habitat/assemblage

Considering a reference level and target (EcoQO) for each species within each of the habitats/assemblages is intuitively a more ecologically realistic approach. It obviates the need to carry out any form of weighting to the data but increases the number of targets to be monitored and managed.

Scenario 3: An index of opportunists or sensitivity

Given that the sampling methods used for marine benthos (Section 5.2.2) yield information on all the taxa present, there is no additional cost in gaining data on all species. The high cost of Scenarios 1 and 2 arises from the costs of setting and managing for the high number of EcoQOs. An approach that reduces this cost, and may increase the communicability of such a large body of information, would be an index describing the status of the species of concern. A metric, for example based on the proportion of individuals in the assemblage that are "opportunists" or "sensitive", would serve this purpose.

Such a metric while increasing communicability will be much more difficult to manage for. For example, if the proportion of sensitive species were to fall this could be the result of damaging activities or the result of some process benefitting other components of the community, for example something as simple as a good recruitment event.

Scenario 4: The proportion of all species

The proportion of the species present that are opportunistic/sensitive provides a less labour-intensive metric than, say, the proportion of individuals that are allocated to a particular category. However, it is likely to be much less sensitive to changes in the status of the system and may have a very high "false alarm" rate.

Scenario 5: The density of a selection of sentinel species

The concept of using sentinel or indicator species is well established. For example, in the terrestrial environment, birds are now widely used in this role (Section 4). They are relatively easy to census (often using volunteers) and have a high value to many stakeholder groups. Their ecological role near the top of the food chain also increases their utility as ecosystem indicators. Cold-water corals could be strong candidates for a benthic sentinel species, as could other epibenthic sessile forms such as seapens. The advantage of this approach is its likely public acceptability, ease of communication and direct links to physical damage to benthic communities.

WGECO favoured development of Scenario 5, while SGSOBS felt that Scenarios 3 and 4 offered the best way forward. Both groups agreed that Scenarios 1 and 2 were impractical. SGSOBS felt that… *scenario 5 on using indicator (sentinel) species is considered too simple an approach because of the virtual absence of stressor-specific indicator species/taxa among the sensitive and opportunistic species* (Section 9.3 in ICES, 2004). Scenarios 3 and 4 were therefore favoured by default. WGECO had discounted Scenarios 3 and 4 as they ... *seek to use indices to summarise the information at a community level. As such, they hide much of the detail and will be difficult to manage for* (Section 5.2.5.3 in ICES, 2003b).

It would therefore appear that at this time there is no scenario for the management of benthos using EcoQ elements (o) and (p) that receives strong universal support.

WGECO believes that it may be possible to develop an appropriate community level index that would allow Scenarios 3 or 4 to be applied. However, we lack the necessary understanding of the biology of many of the species involved and the dynamics of marine benthic communities to make this a realistic prospect in the short term.

SGSOBS also pointed out the diversity of indices and multivariate statistical techniques routinely used in the analysis of community data and which have a good track record for the identification of perturbations, which are differentially species-specific in relation to the perturbation force. While acknowledging the possible role of such indices and techniques as “descriptive surveillance metrics” (see also **Section 9**), WGECO notes that these approaches cannot form the basis of a metric of an EcoQO that can be managed for (for a more extensive discussion see (ICES, 2000)).

WGECO also agrees with SGSOBS that there is a complete absence of any evidence demonstrating a specific link between individual manageable human activities and any opportunistic species, because of the nature of the organisms. It is unlikely, therefore, that Scenario 5 can be applied to EcoQ element (p). However, at least for physical impacts of towed fishing gears, a subject on which WGECO has some expertise, we believe it is possible to identify “sentinel” species to allow the application of scenario 5 to EcoQ element (o). There are a number of groups within the ICES community and elsewhere which have a track record in the development of objectives for specific perturbations and which change in specific ways. WGECO must take account of this expertise in a management structure which requires the development of objectives for a broader range of perturbations.

In this section, a number of criteria are tested for operational management based on the EcoQOs applied to sentinel species (i.e., in accordance with Scenario 5). A case study is used to illustrate the performance of the EcoQOs.

Case Study - Dove Time Series, Station P

A thirty-year time series of infaunal abundance data was available from a fixed station (Station P) situated 11.5 miles off the northeast coast of England (55° 07’N, 01° 15’W) and in 80 m of water. Due to the length of the time series, previous analyses of the data have always been at the genus level in order to avoid any problems due to errors of misidentification at the species level, or changes in taxonomy leading to problems with homonyms (Frid *et al.*, 1996). This conservative approach was also applied in this exercise and thus any reference to sentinel “species” is actually in relation to the genus.

Station P is situated within a fishing ground that is targeted primarily for *Nephrops* with otter trawls (Robson, 1995). Fishing effort data for the area (ICES statistical rectangle 39E8) were obtained from DEFRA (CEFAS, Lowestoft) and mean annual swept area (km^2 $year^{-1}$) was calculated by multiplying the total annual effort hours trawling, by the area swept per hour (average width of trawl (km) × distance per hour (km/hour)). This value was a proxy measure of the area of benthos impacted by trawlers over time.

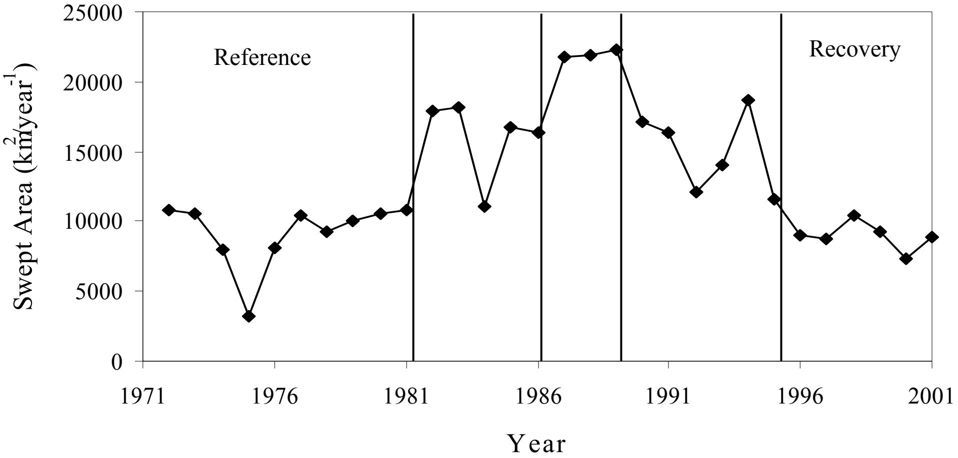


Figure 5.3.4.1.1. Variation in fishing effort (swept area) in ICES statistical rectangle 39E8.

Five phases of fishing effort were determined, based on the level of fishing effort and the stage in development of the fishery. These phases were low effort 1972–1981 (mean: 10,223 km^2/year); medium effort 1982–1986 (mean: 16,082 km^2/year); high effort 1987–1989 (mean: 21,977 km^2/year); subsequent medium effort 1990–1995 (mean: 14,987 km^2/year) and subsequent low effort 1996–2001 (mean: 8,938 km^2/year). Phase 1 was used as the reference period for the EcoQO management model and Phase 5 as the recovery period (see text for explanation).

5.3.4.1.1 Sentinel species

Of the five sensitive species chosen in Section 5.3.3, only three (*Virgularia*, *Arctica*, and *Spisula*) were represented in the Station P time series and, of these, only one, *Virgularia*, was present as more than 1% of the total density in any one sample. A further four sentinel "species" (*Acanthocardia*, *Ampharete*, *Amphiura*, and *Nuculoma*) were selected based on their specific vulnerability to fishing disturbance (physical disturbance) (ICES, 2003b, 2004).

All of the five opportunistic "species" chosen in Section 5.3.3 were represented in the Station P time series, but only two (*Chaetozone setosa* and Oligochaeta) were present as more than 1% of the total density in any one sample. A further three opportunistic "species" were selected based on the opportunistic criteria defined in Section 5.3.2 (*Heteromastus*, *Nephtys*, and *Ophiuroidea*).

5.3.4.2 Operational management by EcoQOs

For the density of each of the sentinel species, a mean and variance were determined for the years of Phase 1 (low effort between 1972 and 1981) to be used as a reference level of the EcoQO. Having established the reference level, a number of scenarios were explored over the time series data (1982–2001) to ascertain the extent to which various applications of the EcoQO would correctly trigger a management response.

Conditions triggering a management response – sensitive species:

1. The density of a sentinel species is 25% lower than the reference level over a three-year period.
2. The density of a sentinel species is 50% lower than the reference level over a three-year period.
3. Where either management scenario 1 or 2 applies, there is also a negative trend in the three-year running mean of sentinel species density maintained over a three-year period.
4. Where either management scenario 1 or 2 applies, there is also a negative trend in the three-year running mean of sentinel species density maintained over a five-year period.

Conditions triggering a management response – opportunistic species:

1. The density of a sentinel species is 75% greater than the reference level over a three-year period.
2. The density of a sentinel species is 100% greater than the reference level over a three-year period.
3. Where either management scenario 1 or 2 applies, there is also a positive trend in the three-year running mean of sentinel species density maintained over a three-year period.

4. Where either management scenario 1 or 2 applies, there is also a positive trend in the three-year running mean of sentinel species density maintained over a five-year period.

Where potential EcoQO metrics would have initiated a management response, data were examined to see if the mean value between 1996 and 2001 (Phase 5 – recovery period) had returned to that found in the reference period (Phase 1: 1972–1981). Prior to Phase 5 there was a 50% drop in fishing effort over a three-year period (Figure 5.3.4.1.1). Unlike the assessment undertaken by WGECO in relation to the spawning stock biomass EcoQO (ICES, 2003b), this method will only be able to detect "hit" rates which trigger a management response, and will not be able to detect false alarms.

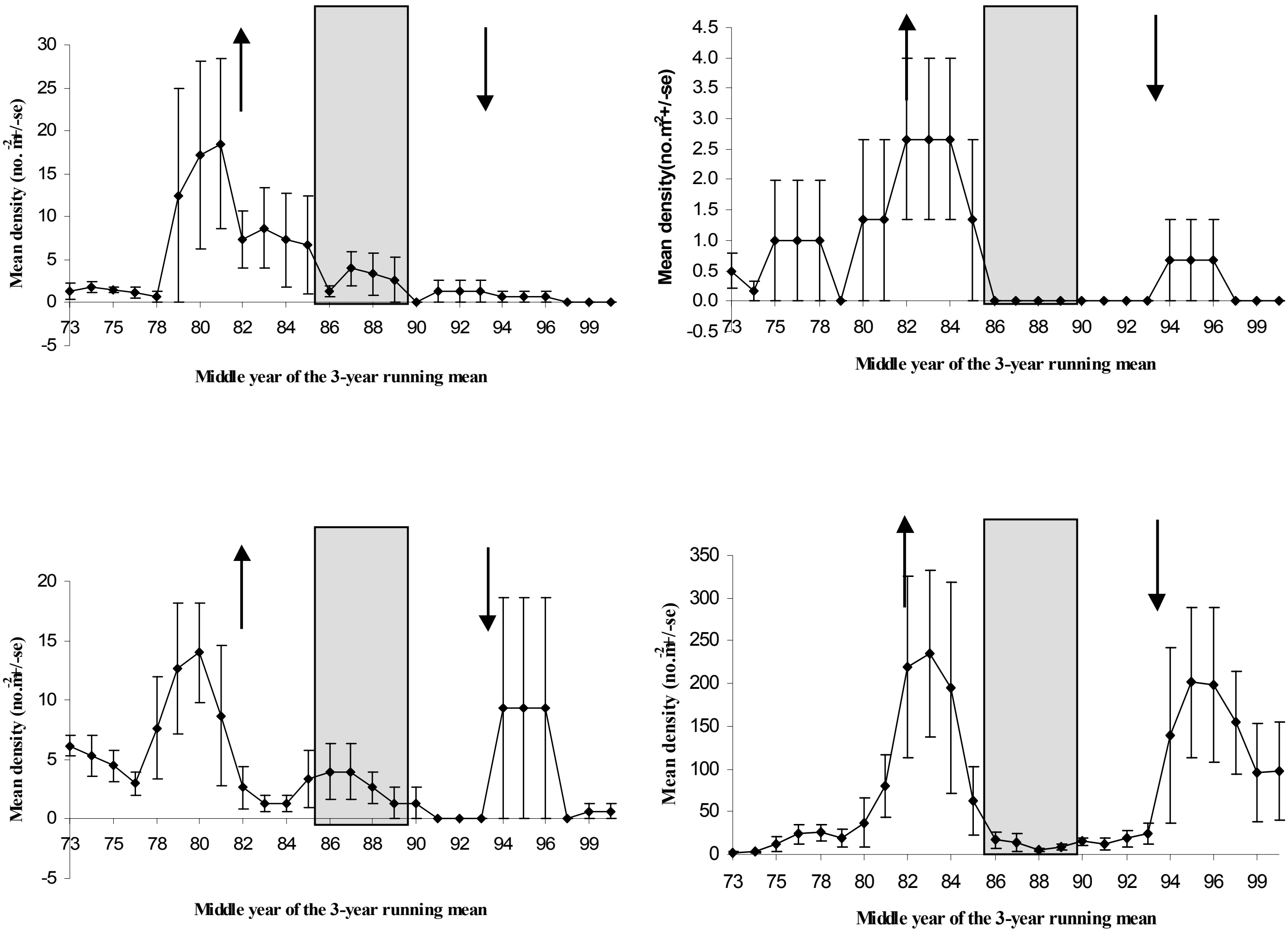
5.3.4.3 Results

Of the seven potential sensitive "species", three (*Acanthocardia, Ampharete*, and *Amphiura*) triggered management in all four scenarios, and one triggered management in the first three (*Virgularia*). In each case there was a reduction by at least 50% of the reference density consistently over a three-year period and also a negative trend in the three-year running mean over a five-year period, with the exception of *Virgularia,* where the negative trend only occurred over a three-year period at any one time (Figures 5.3.4.3.1 a–d). *Arctica*, *Spisula*, and *Nuculoma* did not respond to any of the suggested management scenarios, either because they were so rare in the system that there were no clear trends at all, or in the case of *Nuculoma*, because there was no consistent reduction in density in relation to the increase in fishing effort from 1982. All four species that did decrease consistently in density over the period of increasing fishing effort were present in very low numbers during the period of highest fishing effort (grey-shaded box in Figure 5.3.4.3.1), but the response time to the increase in effort varied between species, indicating that the management response to either scenario 1 or 2 would vary depending on the species selected as sentinels. Further to this, only one species, *Amphiura*, returned to a density that was at least as great as its reference density in the five years following a reduction in fishing effort by 50%. *Virgularia* did not appear to recover in density at all and an initial increase in both *Acanthocardia* and *Ampharete* was followed by a subsequent decrease. It is important to note that *Acanthocardia* was present in very low densities and the implications of sampling error on the use of rare species as EcoQOs of benthic communities must be considered (Table 5.3.4.3.1).

Of the eight potential opportunistic species, four (*Capitella*, *Polydora*, *Cirratulus*, and *Ophiuroidea*) were present in such low numbers that it would be impossible to establish any trends. This is not a sampling error as a 0.5 mm sieve was used throughout the time series analysis (Frid *et al*., 1996). Of the remaining four, two (*Heteromastus* and *Nephtys*) responded to the first three scenarios, showing a consistent increase by at least 100% of the reference value and a positive trend in the three-year running mean over a three-year period. *Chaetozone* responded to all four scenarios, showing a consistent increase by at least 100% of the reference value and a positive trend in the three-year running mean over a five-year period (Figures 5.3.4.3.2 a–c). However, for all three opportunists that responded to the potential scenarios for EcoQOs of sentinel species, none actually showed a clear, tightly linked response to the change in fishing effort. In fact all three were decreasing over the years of increase to the highest fishing effort period (grey-shaded area in Figure 5.3.4.3.2) and it is more likely that the increase in density of opportunists at the end of the reference period (e.g., early 1980s) was actually a response to increased input of organic matter at this time. The relationship between increases in primary production and increased density of macrofauna at another station in this area has been described previously (Frid *et al*. 1996) (Table 5.3.4.3.1).

a) *Virgularia*

b) *Acanthocardia*

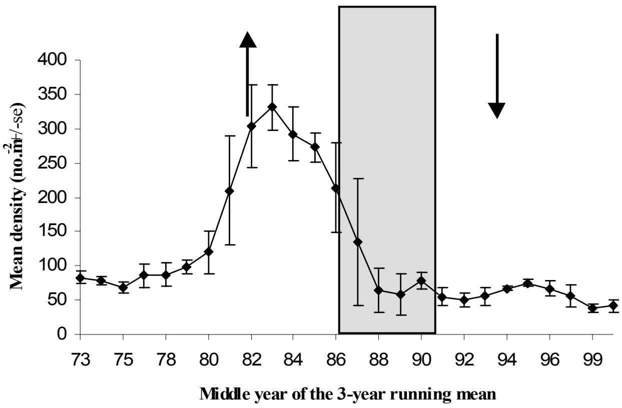


c) *Ampharete*

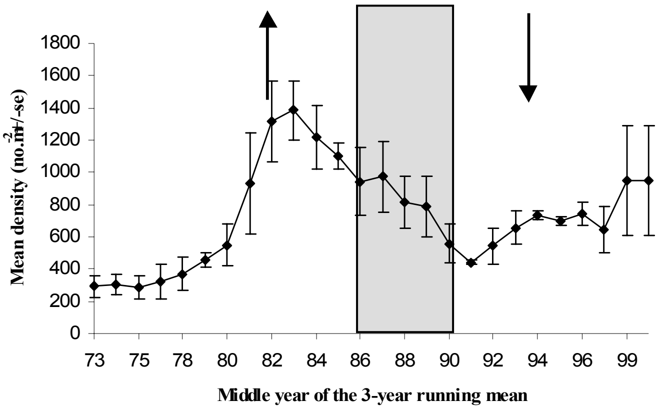
d) *Amphiura*

Figure 5.3.4.3.1. Trends in sentinel sensitive species based on three-year running means. The arrows represent the beginning of the increase in fishing effort from the low reference level and the beginning of the decrease in effort to the recovery period. The grey-shaded area highlights the years of highest fishing effort (Phase 3 in Figure 5.3.4.1.1).

a) *Chaetozone*



b) *Heteromastus*



c) *Nephtys*

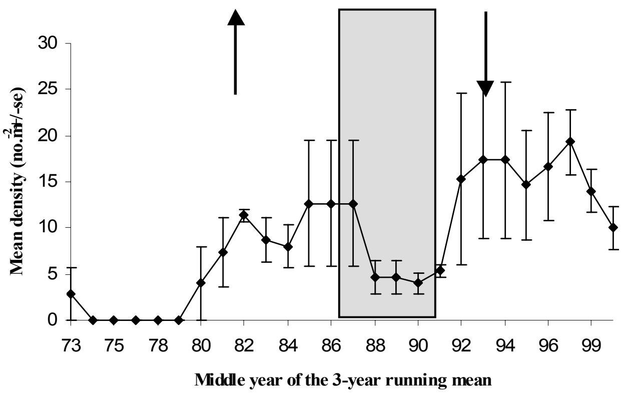


Figure 5.3.4.3.2. Trends in sentinel opportunistic species based on three-year running means. The arrows represent the beginning of the increase in fishing effort from the low reference level and the beginning of the decrease in effort to the recovery period. The grey-shaded area highlights the years of highest fishing effort (Phase 3 in Figure 5.3.4.1.1).

Table 5.3.4.3.1. For each sentinel EcoQO species, a reference mean and variance is given for a period of low fishing effort at the beginning of the time series and for those species that did respond in the direction predicted during the period of increasing fishing effort (e.g., increase in opportunists, decrease in sensitive sentinels), a recovery mean and variance for the period at the end of the time series is also given (see Figure 5.3.4.1.1). The performance of the EcoQO species was modelled against a number of potential operational management scenarios. Where the species did respond to a management scenario a ✔ is given. Management scenarios 1 and 2 refer to a consistent decrease in density from the reference mean for sensitive species and increase in density for opportunistic species at two levels of magnitude. Where scenarios 1 and 2 were found to apply, trends in the three-year running mean of density following the reference period were examined for consistent negative trends in the case of sensitive species and consistent positive trends in the case of opportunistic species (see text in results section for detail of each scenario).

	Reference mean	Reference variance	Scenario 1	Scenario 2	Scenario 3	Scenario 4	Recovery to reference level	Recovery mean	Recovery variance
Sensitive sentinels									
Arctica	0	0							
Spisula	0.17	0.25							
Virgularia	6.56	153.97	Y	Y	Y			0	0
Acanthocardia	0.94	2.28	Y	Y	Y	Y		0	0
Ampharete	7.72	38.38	Y	Y	Y	Y		0.4	0.8
Nuculoma	8.5	71.00							
Amphiura	21.22	927.19	Y	Y	Y	Y	Y	108.8	9411.2
Opportunistic sentinels									
Chaetozone	96.22	1349.51	Y	Y	Y	Y	Y	52	506
Oligochaeta	88.83	4030.06							
Capitella	0	0							
Polydora	0	0							
Cirratulus	0	0							
Heteromastus	388.06	40,172.53	Y	Y	Y			877.6	19,0934.8
Nephtys	2.28	21.19	Y	Y	Y			14.8	59.2
Ophiuroidea	0	0							

5.3.4.4 Conclusions

5.3.4.4.1 Identifying possible species for EcoQ elements (o) and (p)

Both WGECO and SGSOBS have warned against the adoption of the proposed metrics for extensive lists of all species that meet the criteria as opportunistic or sensitive. The primary objection is the logistical problems associated with the vast number of management targets this would require.

However, we also note that there remain no sets of criteria for listing species that are unequivocally sensitive (e.g., fragile) or opportunistic. Here we sought species that appeared on a number of independently derived lists and on lists that appeared in the primary, and hence peer-reviewed, literature. In spite of the very large total pool of species placed on such lists by their proponents, the number of species common to several lists was remarkably small.

5.3.4.4.2 Consideration of sampling scales and existing data series

Any monitoring programme must be undertaken with sufficient effort to ensure that the design has sufficient statistical power to detect changes of the order of magnitude that are of concern. In many survey designs, this is not the case (see Section 5.5.2 in ICES, 2003b).

Having selected a number of species that meet the criteria for sensitive (e.g., fragile) or opportunistic species and which are well represented in the North Sea Benthos Survey or the various beam trawl survey databases, we examined the issue of the scale at which an EcoQO metric may be applied. These analyses confirmed that the setting of indices at the North Sea scale, which has the advantage of limiting the number of management objectives, does increase the between sample variance and so reduces power to detect change. This is fully in line with expectations. Of more interest is the fact that there was little difference in the estimates of the metric when this was done by biological assemblage (the ecologically most relevant scale) or regionally. If this result is generic this could represent a major saving in sampling and management effort, although this preliminary conclusion still needs to be tested with a wider suite of benthic species.

Our analyses show that with properly designed time series it is possible to use the metrics of density of both sensitive and opportunistic benthic species. Some of the sensitive species showed changes that would have correctly initiated a management response. However, the opportunistic species did not show changes that were linked to fishing impacts and so failed to correctly trigger management actions. It is also of note that even with the rigour of a true time series and multiple replicate samples, many of the candidate indicator taxa appeared in such low or variable numbers as to render it impossible to draw meaningful conclusions.

We also note that previous analyses of these data using, amongst other techniques, species richness, diversity and multi-variate community structure, also all highlighted changes in the system at the time fishing intensity changed. This is another indication of the potential value of such descriptive surveillance metrics (see Section 9).

5.3.4.4.3 Consideration of possible scenarios for the application of the metrics for EcoQ elements (o) and (p)

Given the lessons that emerged from our consideration of the SGSOBS report and our own analytical work, and despite the lack of unanimous agreement from other expert groups, WGECO continues to find Scenario 5—selection of a very limited set of sentinel species for application of this metric—to be the most promising approach. We are confident that this approach could be made operational, at least for the physical impacts of towed fishing gears on the benthos. An important stage in this process will be the agreement on a set of criteria that can be used to convert a long list of potential sentinel species to a short list of the most suitable candidates.

5.3.4.4.4 Recommendations

WGECO recommends that:

- BEWG further examine, using the 1986 and 2000 data from the North Sea, the ability to apply metrics on the regional scale as opposed to either a North Sea scale or at the assemblage level.

- OSPAR consider dropping the EcoQ element (p) concerned with opportunistic species, as these are ubiquitous and provide no link to human impacting activities.

- EcoQ element (o) concerned with the density of fragile (sensitive) species be advanced by the use of a selection of a very limited suite of “sentinel” species.

- Criteria are developed for selecting sentinel species that take into consideration data availability.

5.3.5 References

Borja, A., Franco, J. and Pérez, V. (2000) A Marine Biotic Index to Establish the Ecological Quality of Soft-Bottom Benthos within European Estuarine and Coastal Environments. Marine Pollution Bulletin, 40(12): 1100–1114.

Bustos-Báez, S., and Frid, C. 2003. Using indicator species to assess the state of macrobenthic communities. Hydrobiologia, 496: 209–221.

DeBoer et al. 2001.

Fonds, 1991.

Frid, C.L.J., Buchanan, J.B., and Garwood, P.R. 1996. Variability and stability in benthos: twenty-two years of monitoring off Northumberland. ICES Journal of Marine Science, 53: 978–980.

Frid, C.L.J., Harwood, K.G., Hall, S.J., and Hall, J.A. 2000. Long-term changes in the benthic communities on North Sea fishing grounds. ICES Journal of Marine Science, 57: 1303–1309.

Hughes, 1998.

ICES. 1997. Atlas of North Sea Benthic Infauna. ICES Cooperative Research Report, No. 218.

ICES. 2000. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES, Copenhagen, ICES CM 2000/ACME:02, 93 pp.

ICES. 2001a. Report of the ICES Advisory Committee on Ecosystems. ICES, Copenhagen, ICES Cooperative Research Report No. 249, 94 pp.

ICES. 2001b. Report of the Working Group on the Ecosystem Effects of Fishing Activities. ICES, Copenhagen, ICES CM 2001/ACME:09, 106 pp.

ICES. 2002. Report of the Working Group on the Ecosystem Effects of Fishing Activities. ICES, Copenhagen, ICES CM 2002/ACE:03, 195 pp.

ICES. 2003a. Report of the ICES Advisory Committee on the Ecosystem. ICES, Copenhagen, ICES CRR 262, 241 pp.

ICES. 2003b. Report of the Working Group on the Ecosystem Effects of Fishing Activities. ICES, Copenhagen, ICES CM 2003/ACE:03, 193 pp.

ICES. 2004. Report of the Study Group on Ecological Quality Objectives for Sensitive and for Opportunistic Benthos Species. ICES CM 2004, ACE:01.

Klein, and Whitbard, 1995.

Kröncke, and Bergfeld, 2001.

Künitzer, A., Basford, D., Craeymeersch, J.A., Dewarumez, J.M., Dorjes, J., Duineveld, G.C.A., Eleftheriou, A., Heip, C., Herman, P., Kingston, P., Niermann, U., Rachor, E., Rumohr, H. and Dewilde, P.A.J. (1992). The Benthic Infauna of the North-Sea - Species Distribution and Assemblages. ICES Journal of Marine Science, 49: 127-143.

Macdonald, D.S., Little, M., Eno, C., and Hiscock, K. 1996. Disturbance of benthic species by fishing activities: a sensitivity index. Aquatic Conservation: Marine and Freshwater Ecosystems, 6: 257–268.

Nicholson, M., and Jennings, S. 2003.

OSPAR. 2004.

Pearson, T., and Rosenberg, R. 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanography and Marine Biology Annual Review, 16: 229–311.

Pianka, E.R. 1970. On r- and K- selection. American Naturalist, 104(940): 592–597.

Piet et al. 1998.

Robson, C. 1995. Chapter 9: Human Activities. *In*, Barne, J., Robson, C., Kaznowska, S., Doody, J., and Davidson, N. Coasts and Seas of the United Kingdom. Region 5. Northeast England: Berwick-upon-Tweed to Filey Bay. Joint Nature Conservation Committee. Peterborough, UK, pp. 151-158.

5.4 EcoQ element (b) presence and extent of threatened and declining species

5.4.1 Overview and recent history

For clarification, there are two different EcoQOs dealing with threatened and declining species and habitats:

2. Threatened and declining species	(b) Presence and extent of threatened and declining species in the North Sea
8. Habitats	(s) Restore and/or maintain habitat quality

Although the discussion that follows can equally well refer to both habitats and species, the EcoQO related to the “threatened and declining” issue strictly speaking only applies to species (see above). Habitats are covered thoroughly by ToR i) (Section 11).

5.4.2 Statistical power of the North Sea groundfish survey to detect trends in the abundance of rare and/ or declining species

In principle, the management action necessary to reverse species decline, and the measure of success (recovery of extent/abundance), is relatively simple to define. It will, however, be necessary to evaluate the success of management measures put in place to reverse declining trends in species abundance. This section describes the statistical power of our existing surveys for monitoring rare and vulnerable species.

Decreases in the abundance of species that are vulnerable to fishing have often been described from retrospective analyses of monitoring data. Changes in the abundance of vulnerable species described from retrospective surveys are usually highly significant. This is because many years of data are available and because abundance was often relatively high at the beginning of the time series when fishing effort was lower.

An assessment of the value of surveys for monitoring rare and vulnerable species can be made using statistical power analyses. Such analyses can quantify the magnitude of changes in abundance that are likely to be detected, and the number of years of monitoring data needed to detect them (Nicholson and Fryer, 1992). Knowledge of the power of surveys to detect abundance trends is vital if survey data are to guide management decision-making (Nicholson and Jennings, 2004; Rochet and Trenkel, 2003). Managers typically make decisions and assess the success of management on time scales of one to a few years rather than the time scales of decades over which retrospective abundance trends have been described. If power is low, it may take many years of monitoring to detect a significant trend and the survey may have to be modified to provide useful information for managers. Conversely, if power is high, then surveys reliably describe the status of rare and vulnerable populations and will support short-term management decision-making.

Here, we assess the power of a long-term large-scale fisheries survey to detect decreases and increases in the abundance of rare and vulnerable species. To assess power, we assume that the maximum rate of decrease in abundance due to fishing would not exceed 50% of biomass or adult individuals per year (Gislason, 1994) and that the maximum observed rate of recovery from very low population size can be approximated by mean population growth rate in the absence of density dependence (Myers, Mertz & Fowlow, 1997).

5.4.2.1 Methods

Maxwell and Jennings (pers. comm.) have reported on a method to assess the power of the English North Sea groundfish survey to detect trends in the biomass and numerical abundance of fishes. They calculated trends in numerical and biomass abundance from 1982–2002. Trends were calculated for biomass abundance of all individuals >8 cm total length and the numerical abundance of adults, where adults were defined as individuals longer than the length at 50% maturity (assuming “knife-edge” maturity). They included the ten most abundant and well-sampled bottom-dwelling (demersal) species and the ten most vulnerable demersal species. Vulnerability was assumed to be determined by body size, but species were not included if < 150 individuals had been caught in the history of the survey or if the North Sea was outside the main part of their range.

Maxwell and Jennings calculated statistical power for geometric trends assuming constant percentage increases or decreases in abundance year on year. For increases and decreases in biomass, and decreases in numerical abundance, the percentage changes were defined directly. The power calculations involved using the magnitude and pattern of the trend and the number of years of sampling to calculate the expected deviation of the data from a null hypothesis. Then for a specified statistical test and significance level, the probability of the test detecting the trend was calculated for a given level of variability, following the formulation in Fryer and Nicholson (1993). The calculations were used in three ways: to estimate power for a given period and magnitude of trend, to estimate the number of years required to have 90% power of detecting a given trend, and to estimate the magnitude of trend that could be detected with 90% power after a given number of years. Since species with similar life histories exhibit similar responses to a given rate of fishing mortality, we also considered the power of the monitoring survey to detect trends in an indicator that describes changes in the relative abundance of a suite of vulnerable species.

5.4.2.2 Results

For some of the most vulnerable species, such as tope and thornback ray, over twenty years of monitoring would be required to detect a 20% year-on-year increase in adult abundance. Presenting this analysis in terms of the power to detect a 20% annual biomass increase after two, five, and ten years revealed that power after ten years exceeded 0.9 for only three of the ten vulnerable species: anglerfish, hake and wolf fish. For all species, there was little power to detect such a change after 2–5 years. Thus, the survey has very little power to detect species-specific increases in abundance that might be associated with gradual reductions in fishing effort.

The power of the survey to detect year-on-year percentage decreases in abundance depends on the magnitude of these decreases. The power to detect decreases of < 20% after five years was very low for all of the vulnerable species (Figure 5.4.2.2.1). Even year-on-year decreases in adult abundance of 50% are unlikely to be detected after five years for thornback ray, spurdog, cuckoo ray, and spotted ray. For the vulnerable species, power to detect biomass decreases of < 50% per year was less than 1 after five years for all species except wolf fish, saithe, anglerfish, and hake (Figure 5.4.2.2.2).

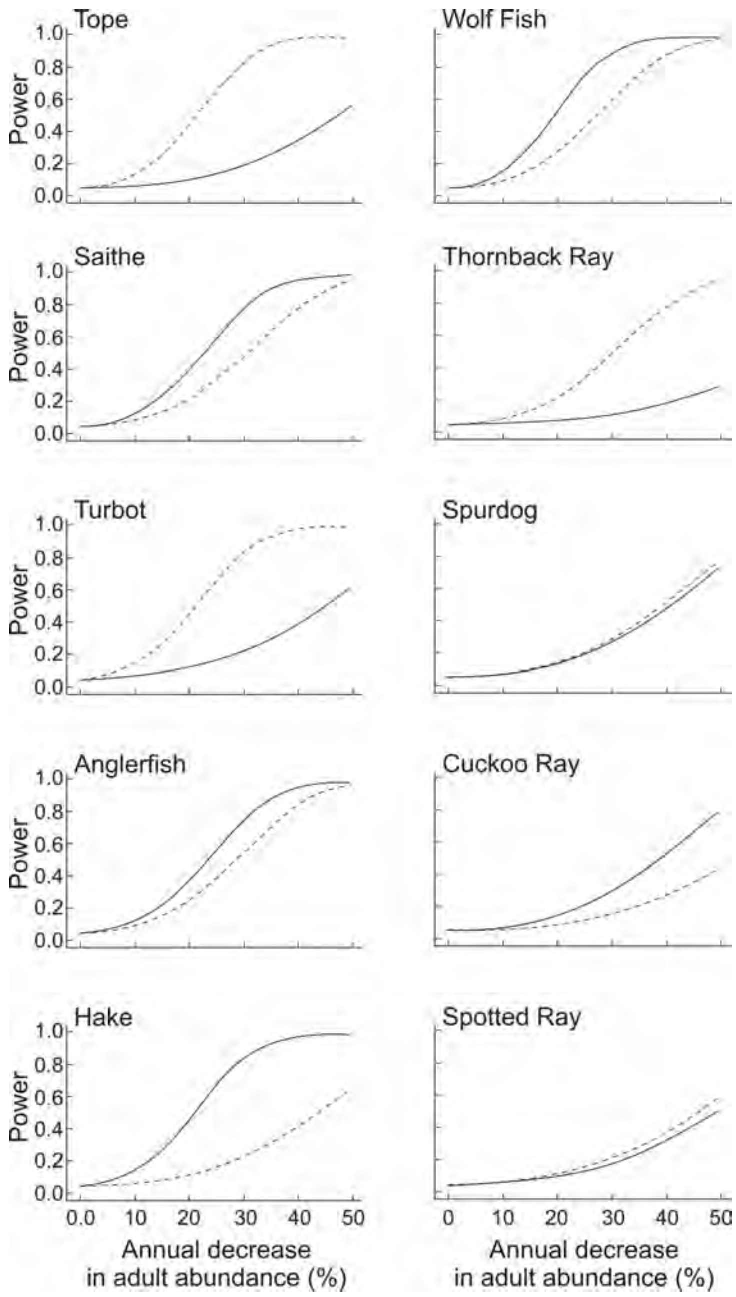


Figure 5.4.2.2.1. Power of the monitoring survey to detect percentage year-on-year decreases in the numerical abundance of the adults of vulnerable species after five years. Continuous lines represent all sites and broken lines fixed sites.

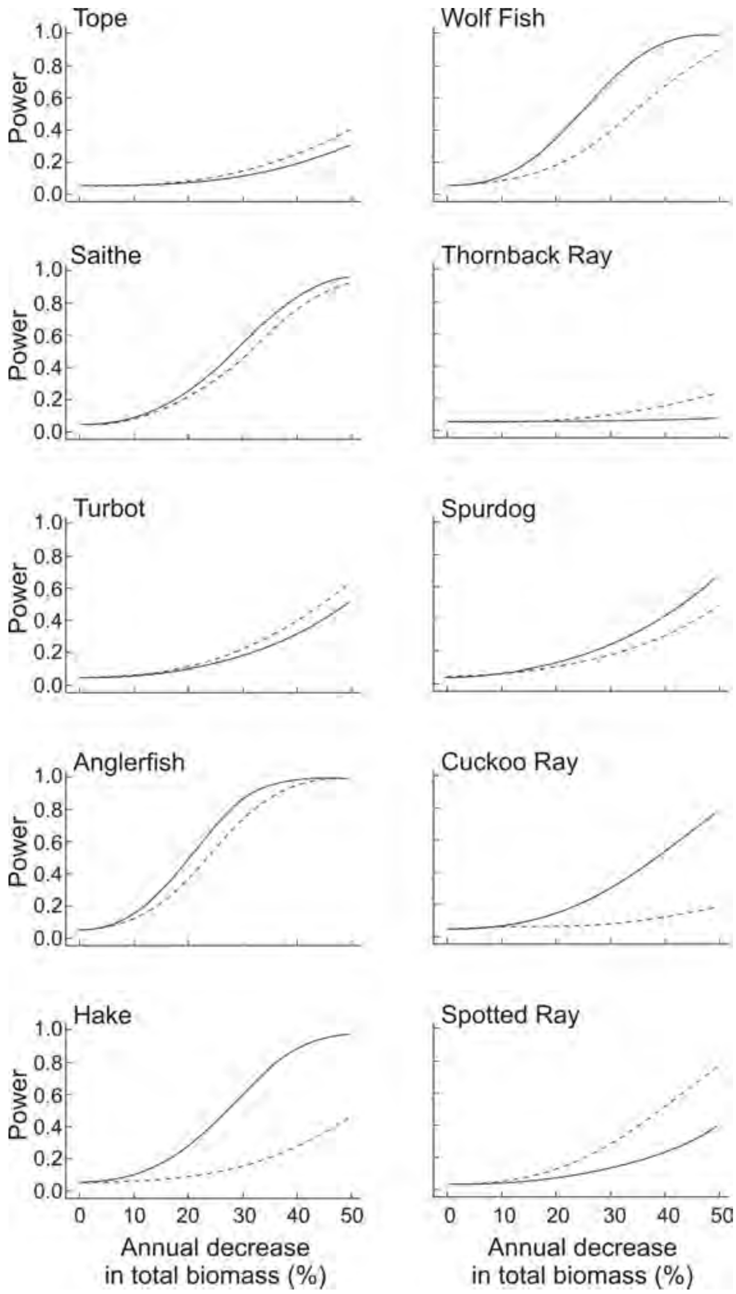


Figure 5.4.2.2.2.Power of the monitoring survey to detect percentage year-on-year decreases in the total biomass of vulnerable species after five years. Continuous lines represent all sites and broken lines fixed sites.

The value of a composite indicator of the abundance of rare species was lower from 1988 than during the reference period 1982–1987 (Figure 5.4.2.2.3). The inter-annual variance in this indicator, as determined by the non-parametric difference-based method, was low relative to inter-annual variance in the abundance of the individual vulnerable species used to derive the indicator. The power to detect different rates of increase in the value of the indicator is given in Figure 5.4.2.2.4.a, and the time required to detect a ≥10% year-on-year increase in the indicator with a power of 90% was less than ten years (Figure 5.4.2.2.4.b).

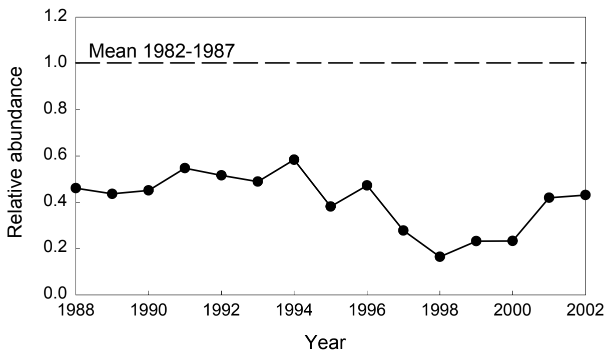


Figure 5.4.2.2.3. Trends in the composite indicator of adult abundance for vulnerable species, as calculated from monitoring survey data.

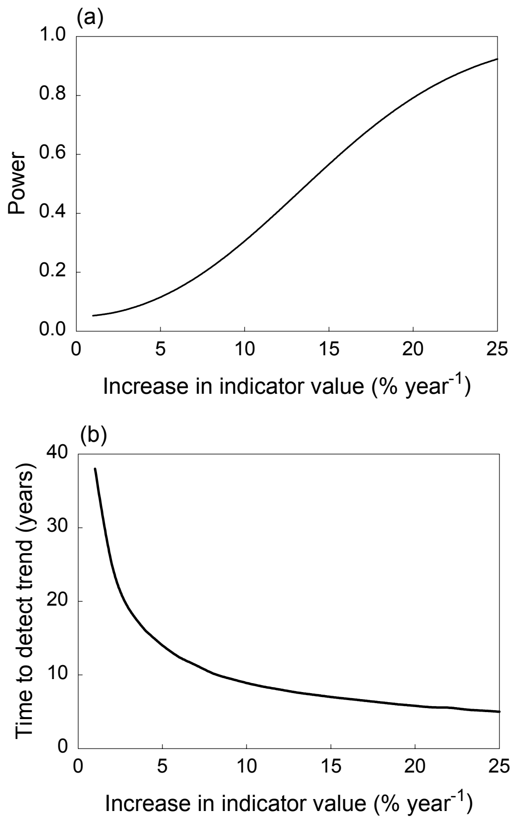


Figure 5.4.2.2.4. Power of the monitoring survey to detect annual percentage increases in the value of the composite indicator (a) and the number of years of monitoring required to detect annual percentage increases in the value of the indicator with a power of 90% (b).

5.4.2.3 Key points from the power analysis

Some formerly abundant but vulnerable species are now too scarce to be caught during monitoring surveys. So, when conservation concern is greatest, monitoring may provide little or no information on whether species are further declining or starting to recover in response to management action.

For rare and depleted species, the power to detect rapid decreases in abundance on time scales < 10 years was poor. Moreover, even if conservation were effective and populations recovered at the maximum potential rate of increase, 5–10 years of monitoring would often be required to detect recovery.

Given limited resources to increase replication on surveys, and given that improvements to one survey would often be to the detriment of another, it is impossible to monitor short-term changes in the abundance of most rare fishes affected by overexploitation. However, despite the often low catchability of the routine fish monitoring surveys for rare species, cost-effective improvements would be possible in the North Sea if all IBTS-contributing countries heeded Daan (2001) and made real efforts to increase the accuracy of taxonomy and recording for rare species. The IBTS research vessel sampling protocol describes the necessary work that is needed to achieve this goal.

Power can be increased by developing composite indicators that track trends in the relative abundance of a suite of species with similar life histories. Such an indicator provides a useful overview of the conservation status of large and vulnerable species, and hence the impact of fishing on part of the fish community. However, given the current political focus on species-based conservation, the species responsible for trends in this indicator would still need to be identified.

5.4.3 The way forward

In deciding how to progress this work further, we were aware of the ongoing process in OSPAR to review the initial list of threatened and declining species. It is undoubtedly important that OSPAR Contracting Parties are made aware of such species in their waters, keep up-to-date records of their status as an ongoing activity, and use the list as a basis for taking national action. However, we are not convinced that such a listing process is the best way to deal with the generic issue of the threat to biodiversity for which threatened and declining species are one possible metric. The development of EcoQOs for the listed species will obviously only deal with those that are listed, while those species that are unlisted will still remain under threat. The analysis of the power of our most comprehensive surveys reported above confirms that even for this selected subset, we may not be able to identify a response to management action on a meaningful time scale. The WWF paper to OSPAR BDC (BDC 04/2/13) usefully reminded the scientific community that the intention of the EcoQ element b) was to develop an objective to progressively reduce the number of threatened and declining species. The following discussion elaborates on options for reporting on the success of an objective that comprises multiple species, and suggests alternatives.

Our analyses suggest that observed trends in the abundance of vulnerable species provide little information to help with short-term management decisions, since trends over management time scales of 1–5 years can be detected with very low power. There are a number of options for resolving this, and they are listed below.

5.4.3.1 Use species-specific EcoQOs but improve the power of the survey to detect trends in the abundance of vulnerable species

This is unlikely to be practical because additional funding for monitoring rare species in offshore locations is unlikely to be provided in the foreseeable future and because improvements to survey design in one region (e.g., increased replication or sampling duration) would often be to the detriment of other surveys. However, power could be increased if some of the concerns raised by Daan (2001) are taken into account and a major international effort were made to improve the standards of taxonomy and data recording for rare species on the IBTS, and the advice in the IBTS sampling protocol is followed.

5.4.3.2 Improve power of species-specific EcoQOs by adopting a composite indicator that tracks trends in the relative abundance of a suite of vulnerable species with similar life histories

One option for recording trends in abundance of threatened and declining species, and of reporting on management success, is to prepare aggregated indices for a number of species. This is a possibility, since power can be increased by developing a composite indicator that tracks trends in the relative abundance of a suite of vulnerable species with similar life histories. This process mirrors that described for fish communities (Section 5.2.2), but uses a selection of species which represent the best examples of threat and decline, rather than the entire fish community sample. Such an indicator provides a useful overview of the conservation status of vulnerable species on a shorter time scale than the abundance trends of individual species. However, since any composite indicator is not based on species with exactly the same responses to fishing mortality and their environment, positive trends in the indicator could mask significant decreases in the abundance of some species. Given the current political focus on species-based conservation, the species responsible for trends in this indicator would still need to be identified and the power of the survey to detect trends in their abundance would still be low.

5.4.3.3 Use species-specific EcoQOs but extend the time scale of management to match the time scale over which trends in abundance can be detected

This parallels the development of multi-annual methods in fisheries management. However, it is unlikely that there will be a significant change from the *status-quo* when economic and social drivers typically operate on short time scales.

5.4.3.4 Avoid the use of EcoQOs related to trends in the abundance of rare or depleted species and focus on "Response" indicators of human activity as tools for assessing the success of conservation measures

An alternative objective for threatened and declining species, which had as its purpose the progressive reduction in the occurrence of this phenomenon, would be an evaluation of the proportion of listed species for which a recovery plan had been prepared and implemented. This "response" indicator would have the advantage that it did not require the centralised development of recovery plans for each species on what may be a long list (this would be the responsibility of expert groups in relevant countries), but would simply assess the rate of progress of this activity. This objective would need to have appropriate performance metrics and be given a meaningful time frame. There are a number of objectives for such a metric, the most obvious being to achieve 100% adoption of performance measures for listed threatened and declining species; however, the final choice of objective is a societal one. Such a framework could, for example, build on the work already undertaken by the FAO Elasmobranch Action Plan, or the UK Government's response to the CBD, the Biodiversity Action Plan.

5.5 Conclusions and recommendations

5.5.1 (l) Proportion of large fish

In relation to the further development of fish community metrics, it is useful to refer to the recommendations made in ICES (2003b) (Section 5.6), which describe the difficulties that will be experienced in the specification of precise EcoQO. One year on these difficulties remain. Key messages were that data collection to develop fish/benthic community metrics will be sample- and gear-specific, and that reference levels can only be identified when we have developed an understanding of the theoretical basis underpinning the relationship between fishing disturbance and the size composition of the fish community. Continuation of our work this year has confirmed that there is a relationship with fisheries for metrics of both mean weight and mean maximum length of fish. However, this relationship is not straightforward, not well understood, and certainly not tightly linked in space and time. As such, these two metrics will be poor performance metrics, and we recommend that they are used only for surveillance of the fish community.

5.5.2 (o) Density of sensitive and opportunistic species

Despite recent effort, there are no sets of criteria for listing species that are unequivocally sensitive (e.g., fragile) or opportunistic. The process undertaken in this section, which identified species on a number of independently derived lists, showed that in spite of the large number of species on such lists, the number of species in common was remarkably small. Both WGECO and SGSOBS have warned against the adoption of the proposed metrics for extensive lists of all species that meet the criteria for being opportunistic or sensitive. The main objection is the logistical problems associated with having a large number of separate management targets. Our analyses show that opportunistic species did not show changes that were linked to fishing impacts and so failed to correctly trigger management actions. There is, therefore, a fundamental problem with the use of opportunistic species as an EcoQO as they are ubiquitous and provide no link to human activities, as they respond to any perturbation. We recommend that OSPAR consider dropping the EcoQ element (p) concerned with opportunistic species. Bearing in mind the work of SGSOBS and our own analysis of sensitive species in this section, we remain convinced that a selection of a very limited set of sentinel species for application of EcoQ element (o) concerned with the density of fragile (sensitive) species is the most promising approach. We are confident that it could be made operational, at least for the physical impacts of towed fishing gears on the benthos. This would require, amongst others, a further examination of the behaviour of metrics on a range of different scales, and the development of a set of criteria for the rational selection of sensitive species. This could be a productive area for further work for the relevant expert group.

5.5.3 (b) Presence of threatened and declining species

At the 2003 meeting, the application of a four-step process for an evaluation of threatened and declining species and habitats to the OSPAR “Initial list” selected four species/habitats which WGECO felt were appropriate to support an EcoQO (oysters, oyster beds, littoral chalk communities, intertidal mudflats) (ICES, 2003, Section 5.4.3). While we were not able to provide a detailed specification of the metrics for these elements, it was made clear that the most obvious objective for threatened and declining species and habitats was to reverse the downward trend in abundance and/or extent of each. This remains the view of WGECO. It was also concluded that there is a substantial workload involved in designing simple metrics with suitable monitoring and assessment strategies for each of the listed species/ habitats, and which have sufficient statistical power to show significant improvement. As before, there is insufficient expertise in the group to develop specific metrics, reconstruct trajectories, and determine their historic performance. In addition, work described in this report, and in ICES (2003), shows that the statistical power of the major fisheries surveys is low, and is unable to report on whether the environment responds to management measures in a short (<5 year) time period. This suggests that monitoring programmes for threatened and declining species will need to be well planned and comprehensive, and that this will require a significant investment.

Our analyses in this report suggest that observed trends in the abundance of vulnerable species provide little information to help with short-term management decisions, since trends over management time scales of 1-5 years can be detected with very low power. Options for improving the power of surveys (further investment, using composite indicators, extending the time scale of assessment) are unlikely to be successful.

An alternative objective would be an evaluation of the proportion of listed species for which a recovery plan had been prepared and implemented. This “response” indicator would not require the centralised development of recovery plans for each species on what may be a long list, but would simply assess the rate of progress of this activity. This objective would need to have appropriate performance measures and be given a meaningful time frame. There are a number of objectives for such a metric, but the most obvious would be to achieve 100% adoption of performance measures for listed threatened and declining species. This would alter the focus of the international community away from species-based conservation, and towards a higher-level assessment of conservation action. The development of individual species Recovery Plans, which could take a number of forms depending on the availability of data, level of knowledge, etc., would be the responsibility of local or regional management with involvement by relevant Contracting Parties.

5.6 References

Daan, N. 2001.

Fryer, R., and Nicholson, M. 1993.

Gislason, H. 1994.

ICES. 2003. Report of the Working Group on the Ecosystem Effects of Fishing Activities. ICES, Copenhagen, ICES CM 2003/ACE:03, 193 pp.

Myers, Mertz, and Fowlow, 1997.

Nicholson, M., and Fryer, R. 1992.

Nicholson, M., and Jennings, S. 2004.

Rochet, and Trenkel, 2003.

6 INCREMENTAL ADDITION OF ECOSYSTEM CONSIDERATIONS TO THE ICES ADVISORY FRAMEWORK

Term of Reference: d) begin consideration of the means by which ecosystem considerations can be incrementally added to the ICES advisory framework with specific consideration of the approaches adopted by the existing advisory committees.

6.1 Introduction

Ecosystem considerations have been part of ICES advice for some time, especially if fish stocks are considered to be part of the ecosystem. The Advisory Committee on Ecosystems was established in 2000 and has been advising on ecosystem considerations explicitly since then. WGECO therefore interpreted this term of reference as meaning "how can ICES advice be better integrated" – and interpreted "ICES advice" as being all advice derived from ACE, ACME, and ACFM. WGECO notes that in general the advice given by ACME and ACE tends to already consider some, if not all, aspects of the ecosystem, including fish stocks. ACFM have been integrating 'bottom-up' ecosystem aspects, through the work of groups such as SGPRISM and SGGROMAT, and fish species interactions through studies and advice such as those on cod and capelin. This term of reference encourages WGECO to help continue this process.

Hans Lassen presented a diagram of the latest draft of the advisory structure that ICES is moving towards (Figure 6.1). In discussion of the presentation, it was noted that while ICES strived to produce objective scientific advice, the influence of external stakeholders and customer needs is greater in the committees and groups to the right of Figure 6.1. The plan suggests a need for ICES to be responsive to the needs of the clients – and that this need could be better met by the inclusion of participants from client groups in the Advisory Steering Group.

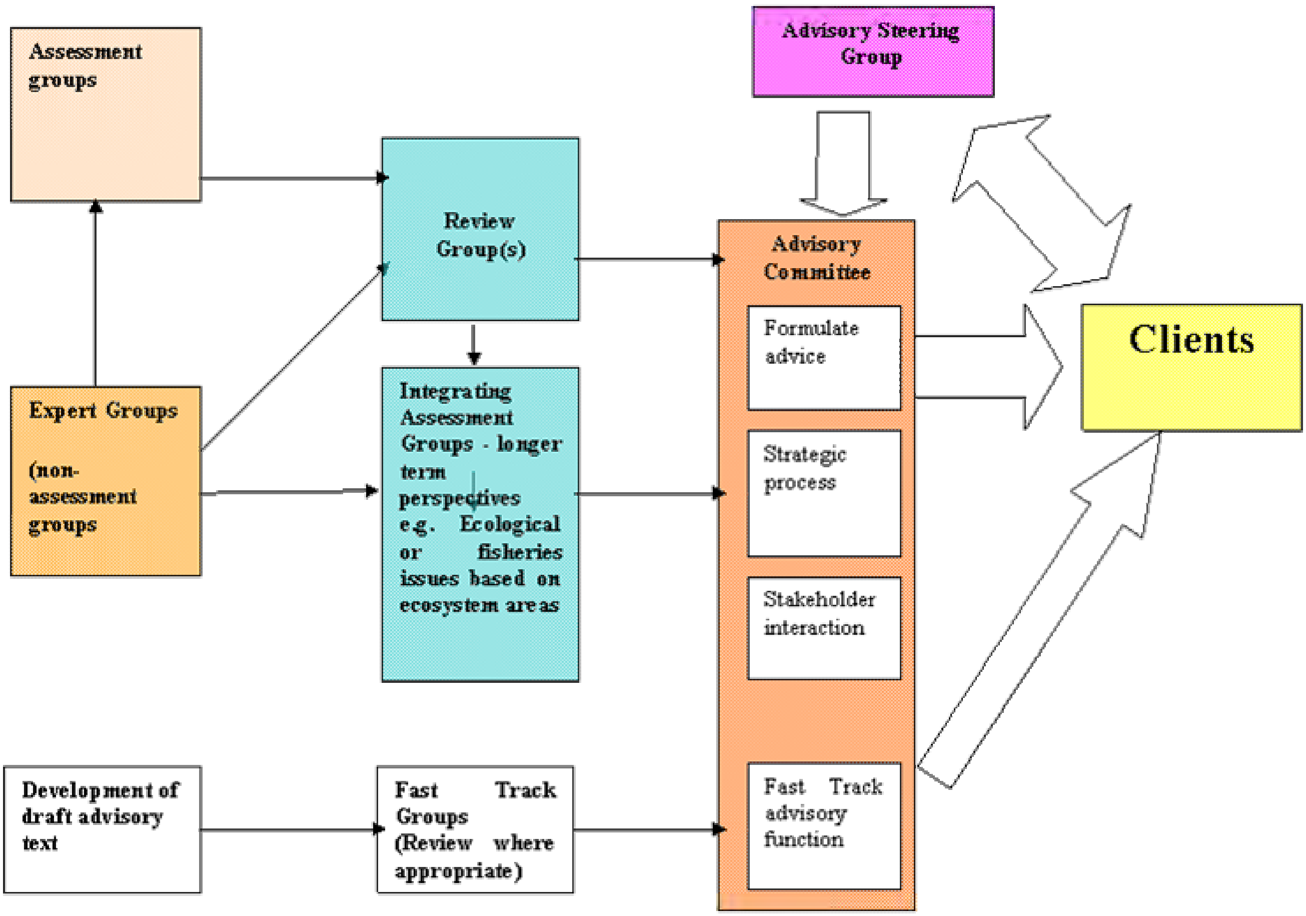


Figure 6.1. Draft diagram of new ICES advisory structure (ICES, 2003a, 2003b).

A draft layout of the new form of unified advice volumes was also provided. ICES advice deriving from ACE will now be combined with that from ACFM in two books (possibly each with multiple volumes) (Box 6.1). The integration of advice from ACME into this structure is planned for 2005. The first book covers specific advice requests from clients, and then deals with fisheries management advice on a region-by-region basis to cover the whole ICES area. The current plan is for there to be eight regions. A further section will deal with widely distributed and migratory stocks and populations, and a final section will deal with deep-water populations and habitats.

1 ICES Advice
- 1.1 Introduction
- 1.2 The scientific base for advice
  - 1.2.1 Introduction
  - 1.2.2 Quality of fishery statistics
  - 1.2.3 Catch projections for the current and following year
  - 1.2.4 Mixed fisheries
  - 1.2.5 Reference points for the status of fish stocks
  - 1.2.6 Ecological quality objectives
  - 1.2.7 Environment impact on fish stocks
- 1.3 The form of ICES advice
  - 1.3.1 Classifying fish stocks on the basis of an assessment of their status – terminology

2 General Advice
- 2.1 Answers to special requests
  - 2.1.1 EC DG Fish
    - 2.1.1.1 Impacts of industrial fisheries
  - 2.1.2 Helsinki Commission
    - 2.1.2.1 Design and establishment of a monitoring programme for Baltic Sea seal populations
    - 2.1.2.2 A marine habitat classification and mapping system for the Baltic Sea
  - 2.1.3 IBSFC
  - 2.1.4 NASCO - information on North Atlantic salmon
  - 2.1.5 NEAFC
  - 2.1.6 OSPAR requests for the further development of a number of ecological quality objectives
  - 2.1.7 Governments
    - 2.1.7.1 ICES Member Countries on providing information and advice on habitat mapping and classification

3 Regional Advice
- 3.1 Northeast Arctic (Subareas I and II)
  - 3.1.1 Fisheries Advice
    - 3.1.1.1 Nominal catches by year and Division
    - 3.1.1.2 Ecosystem impact of fisheries
    - 3.1.1.3 Mixed fisheries and fisheries interactions
    - 3.1.1.4 Single-stock exploitation boundaries and critical stocks
    - 3.1.1.5 Advice for fisheries management
    - 3.1.1.6 Regulations in force and their effects
    - 3.1.1.7 Information from the fishing industry and factors affecting fishing operations
    - 3.1.1.8 Quality of assessments and uncertainties
  - 3.1.2 Ecosystem and environmental considerations

- 3.2 North-Western Areas (Division Va and Subareas XII and XIV)
  - 3.2.1 Fisheries Advice
    - etc.

- 3.9 Widely distributed and migratory stocks and populations
  - 3.9.1 Fisheries Advice
    - etc.

- 3.10 Deep-water populations and habitats
  - 3.10.1 Fisheries Advice

Book 2 deals with individual stock and population advice for each of the chapters from 3.1 to 3.10 in Book 1

Box 6.1. Example structure of book 1 of new ICES advisory book (edited from ICES, 2003a, 2003b).

It can be seen that 'ACE' advice fits in at a number of places in this scheme. In the plan, Chapter 1 will include relatively straightforward descriptions of the scientific basis of Ecological Quality Objectives and evaluations of the impact of the environment on fish stocks (this might be better termed as effect of environment on fish stocks). In Chapter 2, ACE presently deals with special requests from the European (not just DG Fish), Helsinki, and OSPAR Commissions. Within the regional advice sections (Chapter 3 onwards), sections will be needed on the regional ecosystem impact of fisheries and on ecosystem and environmental considerations. It appears that the advice for fisheries management, although now given by fleet, will still not include many ecosystem factors.

6.2 Commentary on proposed changes

These proposed changes are obviously a step forward in the integration of advice in order to better support an ecosystem approach to management of marine activities. In the case of advice to DG Fish, this means an ecosystem approach to fisheries management. In 2002, WGECO (ICES, 2002a) and ACE (ICES, 2002b) considered that there were three areas where management needed most immediately to adopt a wider ecosystem approach. These were:

- reduction in impacts on non-target species and on sensitive habitats;
- preservation of genetic diversity;
- protection for species that are ecologically dependent on other species affected by fisheries.

In addition, WGECO noted that there was a high priority to protect habitats which are essential to species at risk or are themselves at risk. The three areas where management advice needed to be adapted were considered in three separate chapters in ICES (2002a, 2002b). We note that advice is now being provided on a fleet basis and this will need full catch (landed + discard) by species as input to the advice provision. It is unclear whether this is happening routinely and at a sufficiently disaggregated level that is fully geo-referenced. In addition, if these data are being gathered by a monitoring programme, it would be a very useful next step to fully record non-target as well as target species discards. Once these data become available, WGECO will be able to help incorporate advice on non-target species into ICES fisheries advice.

6.2.1 Written advice

Both the new advisory structure and the style of the advisory books are intermediate steps for integration; most obviously ACME and ACE's input to the process has not yet occurred. WGECO were uncertain how much client input has been taken account of and would support any efforts to include this.

In the introductory chapter of Book 1 of ICES Advice (Box 6.1), we suggest that Sections 1.2.3 and 1.2.4 be moved to follow what is presently Section 1.2.7 as we would logically expect the catch projections for the current and following year to occur after the integration of ecological and environmental considerations. WGECO thinks that it would be helpful to distinguish between the oceanographic/hydrographic/climate effect on fish stocks and the effects of other parts of the ecosystem on fish stocks in the present Section 1.2.7, perhaps by adding another Section. Ocean climate effects can probably be updated on a near annual basis. If the methods identified by SGPRISM and SGGROMAT are implemented, this will necessarily integrate oceanographic data with fisheries advice.

Evaluations of the effects of predator-prey interactions on stock status and dynamics are likely to be updated less frequently. Aside from a few stocks in the Barents Sea, species interactions are included explicitly in assessments and advice only through periodic updating of the M2 vectors (predation mortality by age and species) of some stocks in the Baltic and North Seas. WGECO is informed that it would be problematic to extend this particular approach to other ICES areas, or make it more integrative. Integrating predator-prey ecosystem considerations with fisheries advice will require new approaches to this problem and correspondingly more work on the development of analytical tools.

Reference points for the status of fish stocks (Section 1.2.5) should include consideration of the preservation of genetic diversity. This should be expressed in advice within Section 3.1.1.4 (etc.); genetically-separate spawning aggregations need to be considered separately. As noted elsewhere, ecological quality objectives are at present not the only basis for scientific advice on the ecosystem. For example, objectives deriving from the various European Union Directives relating to habitats, species and birds, or to water quality need to be included.

Within the regional advice sections, we assume that longer-term strategic advice for fisheries management is included in the ecosystem impact of fisheries and fisheries interactions sections of fisheries advice, with the section on ecosystem and environmental considerations relating more to non-fisheries advice particular to these regions. WGECO suggests splitting Section 3.1.2 into two sections: Ecosystem Advice and Environmental Advice, each with relevant sub-sections as necessary. Both ACE and ACME are likely to have views on these subsections. We were unclear why the deep-water section (3.10) was the only one to include 'habitats' in its title. We assume that habitat advice will be included in all sections.

6.2.2 Advisory structure

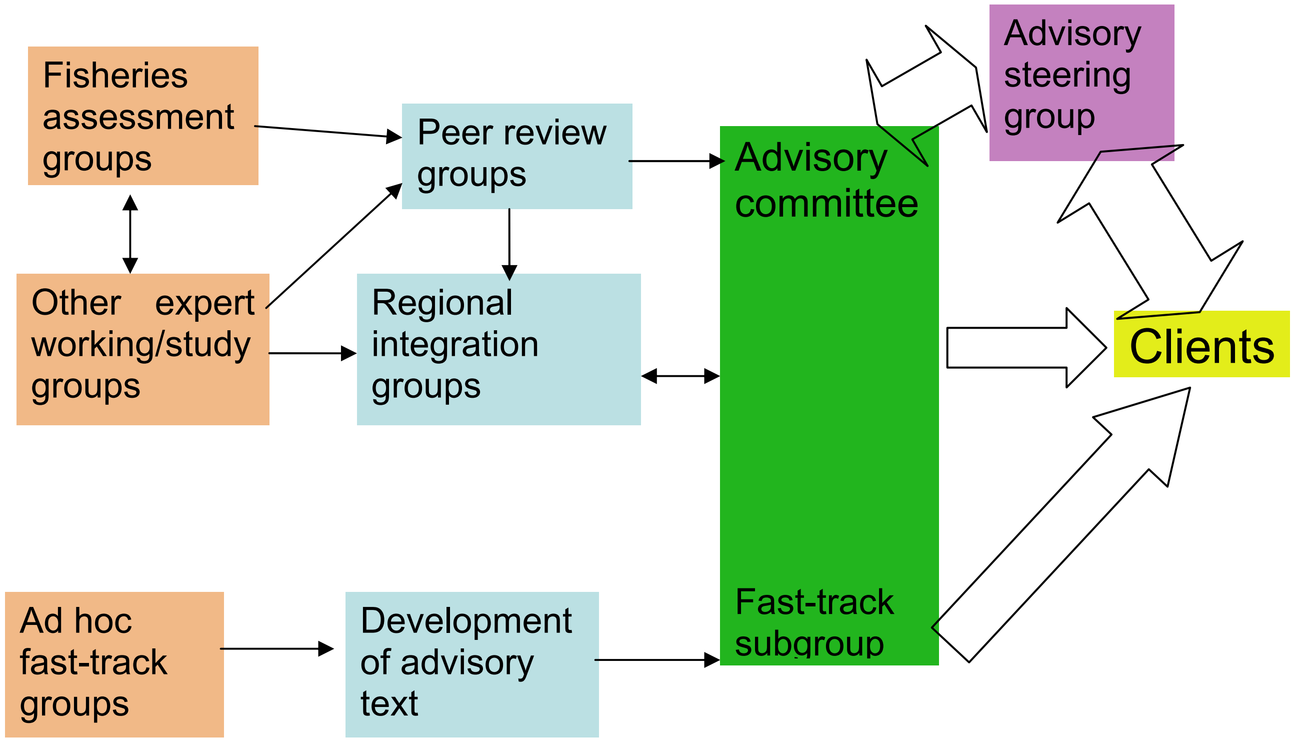


Figure 6.2.2.1. Draft diagram of new ICES advisory structure, with WGECO suggestions.

Figure 6.2.2.1 shows a few changes suggested by WGECO to the flow of work ahead of advice provided to us (Figure 6.1). Starting from the top left, we note that a two way flow of information will be needed between the Fisheries assessment groups and the Other expert working/study groups. For instance, Assessment groups will need input on hydrographic conditions, while the Other expert groups such as WGECO will need input on the state of fish stocks. We note that in the fast track process, development of scientific information is likely to precede the development of advice. Finally we note that there will be a necessary interaction of the Advisory Steering Group and the Advisory Committee(s), rather than a one-way flow. There are of course many other feedback routes within ICES between the groups that occur on an approximately annual basis.

6.3 Role of WGECO

WGECO fits into the above structure as an "Other expert working group". WGECO's role in integrating ecosystem considerations could be several. We suggest a suite of roles as follows:

Evaluating ecosystem effects of fishing by gear (and method) and region across the ICES area.

In order to help provide advice on a fleet basis, WGECO could evaluate the effects of fishing on the ecosystem on a metier basis. This information could then be provided to each of the Regional Integration Groups in a standardised fashion. These reviews could be on a rolling basis, with one or two gears (and method) being reviewed per year. Inputs would be needed from Fisheries Assessments Groups and other specialist Working Groups. Outputs from WGECO would be much improved if information on effort and catch (landings + discards) was made available in as temporally and geographically disaggregated form as possible and in as timely a fashion as possible (see also Section 12). WGECO has pointed out this need repetitively in the past, and fully supports the call for similar information by ACFM in its autumn 2003 advice. We note the need to continue to lower the size limit of fishing vessels for which effort data is collected, both to allow a fuller evaluation of fishing effects and to account for the tendency towards smaller, more powerful vessels.

Evaluating ecosystem effects of non-regional fisheries (migratory and deep-water stocks)

We assume that these evaluations will feed via the peer review groups into the Advisory Committee.

Developing EcoQ metrics/indicators

This is a role undertaken by WGECO for some years (see also Section 7 of this report). We envisage this continuing, along with the role of ensuring consistency among the suite of metrics/indicators by further developing a framework. WGECO may be able to advise on appropriate values for EcoQOs and could evaluate performance against the EcoQO, especially if no other specialist group is covering a particular issue.

Contribute to the development of Strategic Environmental Assessments for fisheries

If, as seems likely, some form of formal environmental assessment of fisheries is introduced in the European Union, WGECO would be able to contribute to the scientific development of appropriate procedures and frameworks.

Watching for new information – both on new issues and new information on existing issues

An important role to ensure that ecosystem advice is kept fully up-to-date.

6.4 Role of Regional Integration Groups

Clearly the Regional Integration Groups have a crucial role in the process. REGNS and BSRP are the first such groups and are piloting integrated assessments for 2006. There is at present substantial uncertainty in planned working methods. WGECO would find it helpful to have a clear idea of how these groups will carry out their assessments by the time of WGECO 2005, so that we can comment on the process(es) and design effective inputs.

6.5 References

ICES 2004. Delegates Meetings Decisions. in ICES Annual Report 2003, pp. 122-135. International Council for the Exploration of the Sea.

ICES 2003. Report of the Management Committee for the Advisory Process (MCAP Meeting Sunday, 21 September 2003). ICES CM 2003/Del:22, 4 pp.

7 TOR E) COMMENCE DEVELOPMENT OF THE SCIENTIFIC COMPONENTS OF THE FRAMEWORK AND GUIDELINES FOR THE CONSIDERATION OF MULTIPLE ECOQO'S AS INTEGRATED SETS FOR USE IN APPLIED CONTEXTS

7.1 Approach

There are three aspects to consideration of multiple EcoQOs as integrated sets for use in applied contexts. The first is choosing the set of EcoQOs wisely to begin with. This issue is dealt with at length in Section 7.2, and relevant material is also included in Section 7.4.

The second is aggregating the information in multiple EcoQOs into a smaller number of EcoQOs through some algorithm or at least exercise in logic (Section 7.3 and Section 9). When we refer to aggregation of indicators, this has two aspects. Sometimes the ecosystem status covaries tightly on the metrics associated with a number of separate EcoQs, because the separate EcoQ metrics are measuring generally the same ecological response to the same environmental condition. In those cases, "aggregation" is simply pooling results of a number of indicators whose covariances are known and high. (For example the UK Farmland Bird Index first requires establishing the recent trend in each pre-identified farmland bird species separately and, if their trends are all generally the same, a combined index is calculated.) This elimination of known redundancy is different from aggregating the performance of a variety of EcoQOs and their metrics, when the covariances are either unknown or not particularly strong. Section 7.3 deals with the latter condition, which is common in marine conservation (notwithstanding the frequent use of ordination methods with ecological metrics.)

The third is actually guiding the decision-making in applied contexts, on the basis of the status of multiple metrics linked to the EcoQOs, relative to their reference points. This is addressed concisely in Section 7.4.

In addressing this Term of Reference, WGECO concluded that the DPSIR framework—Driver-Pressure-State-Impact-Response—(OECD, 1993; UNEP, 2000; IIED, 2002) would provide a useful structure in which to organise the selection of suites of EcoQOs for management. This framework has played a prominent role in selecting indicators and sometimes objectives in areas of environmental quality and sustainable development. Although it is the basis for work on indicator selection by the European Environment Agency in support of, for example, the Water Framework Directive (EEA, 2003), it has received little attention in many parts of the fisheries science and advisory community.

Conceptually, the DPSIR framework is compatible with evaluating the ecosystem effects of fishing, although some aspects of the application require careful thought. The relationship among the five parts of the framework is shown in Figure 7.1.1 (after EEA – reference). Considered individually in a fisheries context:

Drivers – These are the forces which exert pressure on the ecosystem and its components. They may be anthropogenic or part of the natural environment. For ecosystem effects of fishing, the direct drivers are economic and social policies of governments, and economic and social goals (implicit or explicit) of those who prosecute fisheries. Environmental drivers such as oceanographic conditions also affect fish populations and marine ecosystems as well, but would not be the subject of EcoQOs for keeping fisheries sustainable.

Pressures – These are the ways that the drivers are actually expressed, and the specific ways that ecosystems and their components are perturbed. For ecosystem effects of fishing, the central pressure would be Fishing Effort, of which there are many aspects and indicators.

State – These are the properties of the ecosystem itself, and where humans are considered part of the ecosystem, properties of the fishery. For ecosystem effects of fishing, there is a vast list of potential State properties, from biomasses, total mortality rates, and size composition of targeted and non-targeted stocks through an array of community measures and including properties of the physical habitat. State indicators of the fishery itself include fleet size and composition, jobs provided, and landed value of catches.

Impact – These are the changes in State caused by the Pressures. For ecosystem effects of fishing these would be things like fishing mortality and increase in the slope of the size spectrum.

Responses – These are society's actions, taken in response to impacts judged to require remediation. Examples for ecosystem effects of fishing might be a decommissioning policy for excessive fishing capacity, or a closed area to protect a specific habitat feature.

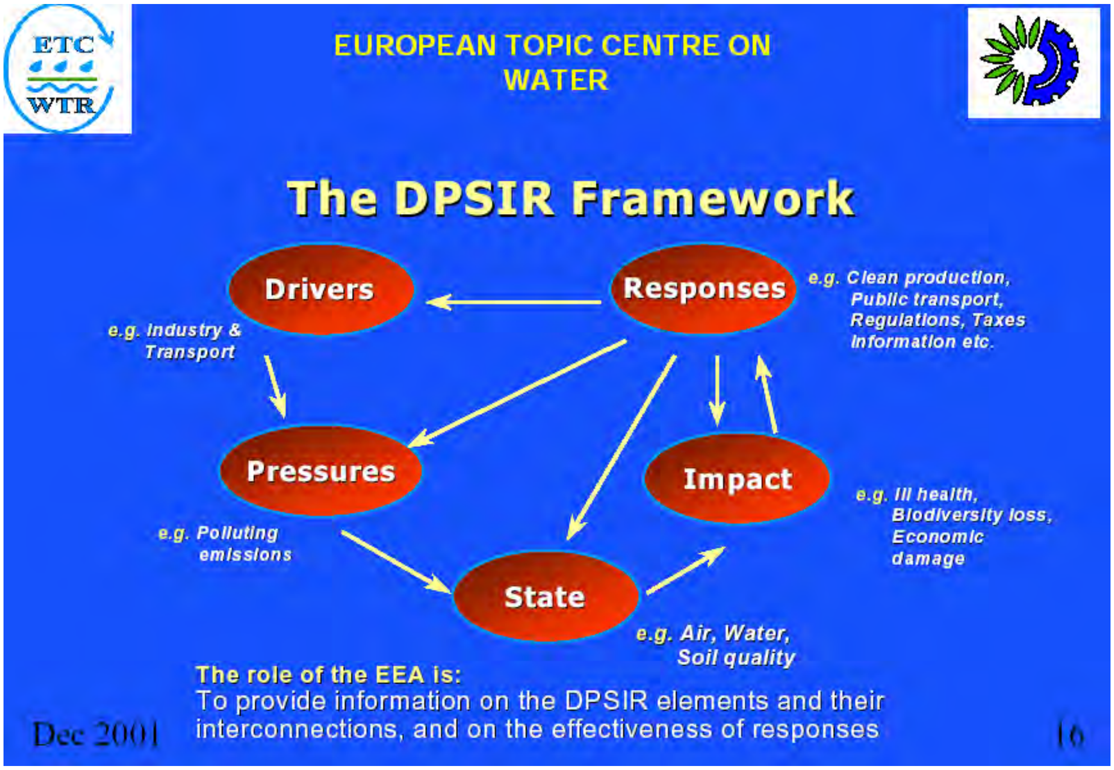


Figure 7.1.1 The DPSIR frame as used by EEA.

The DPSIR framework can become confusing, because the classification of at least some things as a Pressure or State or Impact can depend very much on the context. For example, to the fishery, "Catch" will be viewed as a State property, whereas to the species being exploited and the ecosystem, "Catch" will be viewed as an impact. Similarly, as the arrows in the figure suggest, indicators can start off associated with one type of property but change to another with a change in management. For example, Days Fished can be an indicator of the Pressure "Fishing Effort", but once an effort control programme is introduced, Days Fished is also an indicator for the Response "Effort reduction".

There is also a risk of confusing the Objective set in one of the five classes with the indicators, or metrics, used to measure how well the Objective is being achieved. As high-level conceptual objectives like "Keep stocks healthy" are unpacked to more and more specific EcoQOs, in fact, the transition from an Objective like "Keep spawning biomasses within their zone of high productivity" to "Keep SSB above 25,000 t with 95% probability" pretty much eliminates any distinction between a statement about an objective and a statement about the indicator used to measure achievement of the objective.

This evaluation will try to maintain focus on evaluating EcoQ *Objectives* within each class of DPSIR. The availability of suitable indicators is a factor in the evaluation approach, but it is only one of several criteria in the evaluation. Using the DPSIR framework is considered to be worth a trial, for a better reason than merely curiosity. In the general public, environmental concerns are largely made synonymous with concerns about State properties, and hence there would be a predisposition to set EcoQOs for State properties of ecosystems. However, there is no guarantee that management is guided as effectively by EcoQOs for State properties as it would be by EcoQOs for Pressure, Impact, and possibly even Response properties. The evaluation framework was developed and applied to see the degree to which each type of property would be a suitable basis for EcoQOs which really would be effective in applied contexts.

7.2 Evaluating EcoQOs in a DPSIR Framework

7.2.1 The evaluation concept

It is widely acknowledged that suites of EcoQOs will be needed for the conservation, protection, and sustainable use of marine ecosystems (Trenkel and Rochet, 2002; Rice and Rochet, 2004, and see also FAO, 2003). To be operational, individual EcoQOs have to be fairly specific. Because society has a large number of ecological, social, and economic goals, the suites of EcoQOs will have to be large to address the sets of goals comprehensively.

At present, EcoQOs and their metrics appear to be accumulating opportunistically rather than according to a structured approach to comprehensive coverage of the full range of ecological, social, and economic goals. This situation may result from a lack of a forum for such coordinated planning, because it is proving to be more tractable to develop EcoQOs and their metrics in some areas than others, or simply because there is more enthusiasm for the task in some groups of specialists than in others. Whatever the cause, there are some undesirable consequences of the current situation. We may be moving towards a management regime where managers, policy-makers, and science advisors are struggling to support or apply decision-making constrained by many EcoQOs, yet some of the EcoQO constraints may be redundant or even contradictory, and some important ecosystem properties (or societal goals) are not being protected or advanced by the EcoQO-based management framework.

This evaluation is intended to consider whether a variety of types of EcoQOs are comparably tractable, and whether currently there are important gaps in coverage where EcoQOs for management can readily be developed. It combines the DPSIR process for evaluating human impacts and sustainability, previous WGECO work on properties of good metrics to be used as the implementation tool for EcoQOs, and new insights from work on ToR g) of this meeting.

7.2.2 The evaluation matrix

There are FIVE components to the DPSIR evaluation model: Drivers, Pressure, State, Impact, Response.

There are TWO fundamentally different types of tasks served by EcoQOs:

- EcoQOs may be used to prevent “serious or irreversible harm” (*sensu* Rio Declaration text on the application of precaution) to ecosystem components. These EcoQOs to “avoid harm” are usually phrased so that the conservation LIMITS of the associated indicator(s) are a key consideration in management advice.

- EcoQOs may be used to achieve some socially desirable “good” state for the ecosystem component. These “aspirational” EcoQOs are usually phrased so that the management TARGETS are a key consideration in management advice.

Conservation and Aspiration in EcoQOs – Different or the Same?

There is debate about whether these two roles mean that there are two different types of EcoQOs, or a single type of EcoQO enacted with two types of reference points—avoiding limits and achieving targets. This is more a debate of semantics than of concepts. At a conceptual scale, ecological (as contrasted with social or economic) EcoQOs are to maintain healthy ecosystem components: healthy and productive fish stocks, rich and diverse marine communities, etc. When such conceptual objectives are made operational, however, and linked to specific metrics and reference points, the expressions associated with the two roles are necessarily different. EcoQOs to avoid harm are phrased as keeping the assessed value of the metric away from the conservation reference point with high probability. Aspirational EcoQOs to achieve societal desires are expressed as moving the assessed value of the metric to the target reference point in a risk-neutral mode (i.e., be as close as possible to the target, and above it on average half the time). For simplicity of writing, these will be called two types of EcoQOs. However, nothing important in the framework is changed if one prefers to consider them one type of EcoQO with two different phrasings linked to two different types of reference points.

In an ideal world, management would be striving to achieve positive targets for all ecosystem properties, and the EcoQOs would all be expressed in aspirational language. Unfortunately, many ecosystem components are in poor condition; depleted fish stocks are at risk of collapse and need to be rebuilt, some coastal areas are suffering from eutrophication or harmful algal blooms, many species and habitats are listed as threatened or endangered. Therefore, much of the text which follows will give more attention to EcoQOs intended to avoid harm than to those intended to achieve good. Both types are important, but there is a biological imperative for the avoidance of serious or irreversible harm (Rio Declaration, past WGECO reports). From a scientific/ecological perspective, management must give priority to serious conservation issues over pursuit of economic goals. Correspondingly, the EcoQOs which inform and guide management must be in place to direct the conservation and recovery efforts effectively. Debate about the exact nature of the EcoQOs associated with some desired but far distant state of the ecological quality is largely a distraction from the urgently needed conservation efforts.

WGECO (1999) identified seven criteria of good metrics for use in association with EcoQOs. Although the criteria were for metrics and not EcoQOs themselves, the underlying properties are relevant when evaluating the usefulness of EcoQOs for informing management decision-making. The seven criteria for good **metrics** are:

Criterion	Property
A	Relatively easy to understand by non-scientists and those who will decide on their use
B	Sensitive to a manageable human activity
C	Relatively tightly linked in time to that activity
D	Easily and accurately measured, with a low error rate
E	Responsive primarily to a human activity, with low responsiveness to other causes of change
F	Measurable over a large proportion of the area to which the EcoQ metric is to apply
G	Based on an existing body or time-series of data to allow a realistic setting of objectives

Three of the criteria have parallels which are particularly relevant for the evaluation of **EcoQOs** as well. These are:

- Relatively easy to understand (A) – For an EcoQO, the corresponding criterion is that the EcoQO is <u>interpreted in a consistent way</u> by technical experts, managers and politicians, and the general public.

- Sensitive to a manageable human activity (B) – For an EcoQO, the corresponding criterion is that <u>feasible management actions can be identified and applied</u> in pursuit of the EcoQO.

- Easily and accurately measured, with a low error rate (D) and Based on an existing body or time-series of data (G) – For an EcoQO, the corresponding criterion is that <u>informative indicators/metrics can be identified</u> which fulfill the criteria listed above for good metrics. This includes being able to specify conditions of the objective associated with harm and/or with society's desires.

These will be the three key criteria for our evaluation of EcoQOs, although where other considerations may come into play, we present them. WGECO notes that similar criteria have been identified by other researchers considering important criteria for tools used in management support (e.g., Degnbol, 2003), suggesting that thinking on how to make objectives function effectively in support of fisheries and environmental management may be converging.

In Section 9 (ToR g), we have identified seven components of ecosystems for which EcoQOs are considered likely to be needed, in order to ensure comprehensive coverage of marine systems:

1) Physical and chemical habitat/substratum features;
2) Nutrients;
3) Phytoplankton and zooplankton;
4) Benthos;
5) Fish;

6) Seabirds;
7) Marine mammals.

For all the biotic components, there are three different aspects for which EcoQOs might be appropriate in order to support management:

a) EcoQOs relating to the population status of individual species or stocks (biomasses, numbers, mortality rates);

b) EcoQOs relating to the status of aggregate [community] properties (richness, diversity, size spectra, etc.);

c) EcoQOs relating the "health" of organisms, individually or in aggregate (incidence of disease, contaminant burdens, etc.).

For fish communities, we differentiate commercial and non-commercial species for some aspects of the evaluation, because both the desired performance and the strengths and weaknesses of EcoQOs differ between the two groups on some of the criteria.

In individual applications, not all of the seven ecosystem components may require EcoQOs for objectives-based management to be guided effectively. Likewise even for relevant ecosystem components, not all of the three aspects may require EcoQOs. On the other hand, there may be circumstances where several EcoQOs may be needed for the same aspect of a particular ecosystem component. For example, management may be guided best with a separate EcoQO for the population status of each species in an area which is exploited, or which is listed as threatened or declining.

The diversity of types of EcoQOs combined with the two different roles that EcoQOs can play in management mean that it may not be straightforward to provide full and effective coverage of all management needs, even with quite a few EcoQOs. There is a high risk that even moderately large suites of EcoQOs may provide haphazard and incomplete support for management decision-making. How should an appropriately mixed suite of EcoQOs be selected?

The nature of the threats to ecosystem properties will be an important consideration in selecting the suites of EcoQOs which are likely to be most efficient in supporting management. However, the threats will be application-specific, and so general guidance on selection may not be possible from that perspective. This question still may need further consideration. On the other hand, it may be the case that certain combinations of the DPSIR evaluation approach, the roles of EcoQOs, and the types of ecosystem properties (components and aspects) to be covered fit together very well, and other combinations fit together poorly or not at all. Consideration of these interactions may provide some insight into types of EcoQOs that may be developed very readily and function effectively in management, and types of EcoQOs which may be very difficult to develop and ineffective when implemented.

7.2.3 The evaluation process

For each of the five DPSIR considerations, a table was to be created for which the rows are the seven ecosystem components and their appropriate aspects. Owing to time constraints, only the table for EcoQOs for State could be completed fully at this meeting. The first three columns of each row include Y (likely to perform well), or N (likely to be very difficult to implement) for each of the three key evaluation criteria (Interpretability, Linkable to management measures, and Linkable to informative metric(s) which meet all our criteria for good metrics). The possibility existed to assign a 0 (not a relevant combination of considerations), but this score was never considered appropriate.

Although Y and N appear as absolute judgments, few combinations were cut-and-dried. In cases where N was awarded, it was sometimes possible to think of a few EcoQOs which would meet the criterion, just as a Y could be awarded despite the ability to imagine EcoQOs of the type which would perform poorly on the criterion. For cells where it was possible to think of many cases that would be Y and many that would be N, the ambivalent situation was reflected in extra notation in the column.

As an additional complication, there is growing interest in using reference *directions* in operational management when information is insufficient to identify reference *points* but the direction of harm and improvement are clear (e.g., Shin *et al*., MS 2004). Such considerations are relevant to a complete evaluation of sources of suitable EcoQOs, but we did not differentiate such levels of knowledge. We noted that in cases where reference points cannot be determined, and reference directions are used in management in order to reverse an undesired trend, that reversed trend cannot be expected to continue forever. Consequently, at some point in time, thresholds will still need to be included that indicate when conditions are met that do not need to trigger management action.

Notes on important considerations relative to our criteria, or additional considerations for the row, are in the following two columns. The column "Additional Considerations" was intended to be inclusive of many ecological and management factors, but because of time constraints treatment was not particularly thorough. Monitoring is addressed frequently here, but more time would have resulted in more extensive itemisations of important considerations. The last column of each row contains examples of corresponding EcoQOs currently in place or under development, primarily through Annex 3 in the Bergen Declaration. Where no current examples are known, possible candidates illustrative of the category are included.

Time did not allow the tables to be completed for Pressure, Impact, and Response EcoQOs intended to prevent harm. The entire exercise should be repeated for those tables, and for EcoQOs of all five types intended to achieve desired states as well. In Table 7.2.2, a few illustrative cases are presented as a start on the larger task. The first part of Table 7.2.2 focuses on the EcoQOs and EcoQ Elements proposed in Annex 3B of the Bergen Declaration, but addressing factors other than ecosystem State. The latter part of Table 7.2.2 takes a few EcoQOs for State from Table 7.2.1 and illustrates what possibly comparable EcoQOs for Pressure or Impact might look like. We stress these are *illustrations* of what an EcoQO of the particular type would look like. We are *not* recommending any of these as sound or preferred for use.

The entire evaluation addresses only ecologically-based objectives, but could be conducted for social and economic objectives as well.

7.2.4 Results

Overall EcoQOs for State tend to perform well on our three criteria for many of the ecosystem components. For the biological ecosystem components, there was a tendency for scores on "linkable to a management action" to become lower (or at least more reservations about the Y) as one moved from (ecological) population to community scale EcoQOs. This is consistent with arguments WGECO has made previously (2001 and 2002), and other studies (Degnbol, MS 2004) that community and ecosystem EcoQOs would serve more strategic than tactical functions in management. However, the need for suites of EcoQOs to be integrated and function together efficiently is just as great for strategic purposes as for tactical ones.

Superficially, the large proportion of Ys in Table 7.2.1 would be encouraging. However, when the Notes and other considerations are taken into account, it is noted frequently that large numbers of such EcoQOs for State would be required, in order to give reasonable protection to the relevant component of the ecosystem. Combined with the comments on monitoring needs to get a Y for the criterion "linkable to informative indicators", achieving conservation through the use of EcoQOs for State properties of ecosystems would appear to be a very demanding strategy for both science and management. WGECO expects that with a more complete development of these tables, it will become clear that EcoQOs for Pressures and Impacts might be just as workable as EcoQOs for State. Moreover, many fewer EcoQOs for Pressures and Impacts might achieve all the conservation goals achievable by EcoQOs for State. By being fewer in number, they also would be more cost-effective to implement and provide clearer guidance to consequences of management choices.

If the completion of the remaining tables supports this conjecture, then there is a particular importance to the latter part of Table 7.2.2. Let us suppose that it turns out that EcoQOs for some of D, P, S, I, and R can be made to provide equally effective protection to ecosystems, but some classes are much more readily implemented than others. In that case, it is valuable to note that for EcoQOs of one type, say State, it will often be straightforward to find corresponding EcoQOs of the other types, say Pressure or Impact. This inter-convertibility, if done explicitly, may allow society's tendency to think first of protecting State properties of ecosystems to be satisfied through EcoQOs of other properties which work more efficiently in practice.

Table 7.2.1 Application of evaluation approach to STATE EcoQOs INTENDED TO PREVENT HARM (see Section 7.2.3 for details). EcoQOs from the Bergen Declaration presently applied in a pilot project for the North Sea appear in italics under Examples.

Feature	Property	Inter-pretable	Link to manage ment	Link to Metric	Notes	Other Considerations	Examples
Physical and Chemical Habitat		Y	Y	Y	Need to specify spatial scale of EcoQOs for habitat clearly. If defined by structure (i.e., traits like grain size), need many of them. If defined by class (EUNIS level x), need lots of indicators to circumscribe what "protect" or "damage" means.	Need to have clear definition of whether "habitat" is defined by structure or function (i.e., by what species use it). Aspects of Table A (Bergen Declaration) EcoQ Element (s) of "restore and/or maintain habitat quality" require knowledge of what comprises "habitat quality"; see Section 11.	xx% of area of habitat type yy at EUNIS level 4 is not perturbed by physical disturbance. Concentration of Hg in sediments in region xx [is lower than yy] [does not increase above regional background levels]. *Oxygen concentrations, decreased as an indirect effect of nutrient enrichment, should remain above region-specific deficiency levels, ranging from 4-6 mg oxygen per litre*
Nutrients		Y	Y	Y	SOME sources of change in nutrients can be linked to manageable human activities, but others not. Limited knowledge for setting limit reference points, which are likely to vary regionally.	Nutrient monitoring programmes are already required in many areas, so this would be incremental use of existing programmes.	Concentrations of {[N] [P] [Si]} should not increase beyond xx% of regional natural background concentrations. *Winter DIN and/or DIP should remain below elevated levels, defined as concentrations >50% above salinity-related and/or region-specific natural background concentrations*.
Phyto-/ Zoo-plankton	Popula-tion	Y	N	N	Although some plankton monitoring programmes are in place, few give accurate and precise estimates of abundance/biomass at the individual population scale. Large numbers are needed to give adequate coverage to the ecosystem.		Abundance of *Calanus finmarchicus* during the peak of the spring bloom reaches at least xx $mg \cdot l^{-1}$

Feature	Property	Inter-pretable	Link to management	Link to Metric	Notes	Other Considerations	Examples
	Com-munity	Y?	N?	N	Programs like CPR provide time series for indicators, but weak scientific basis for setting reference points. Partitioning of variance due to manageable human activities and natural variation is not easy.		*Region/area-specific phytoplankton eutrophication indicator species should remain below respective nuisance and/or toxic elevated levels (and increased duration).* *Maximum and mean chlorophyll* a *concentrations during the growing season should remain below elevated levels, defined as concentrations > 50% above the spatial offshore and/or historical background concentrations.*
	Health	Y	Y	Y	Large numbers are possible, but knowledge of regional history should guide selection of the most important EcoQOs readily available. Knowledge of concentrations used as baselines often limited.	Substantial comparative research usually assumed, in order to determine the most sensitive or heavily impacted species/organisms for setting effective EcoQOs.	
Benthos	Popula-tion	Y	Y	Y	Large numbers are needed for adequate coverage of ecosystem. Little knowledge to set reference points for unperturbed states.	Few monitoring programmes in place for on-going evaluation of status on indicators associated with EcoQOs.	*Arctica islandica* density not reduced by more than xx% from regional/historical background levels, in all areas where historic densities were greater than xx nm^{-2}.
	Com-munity	Y? (Some)	Y? (Some)	N	Same comments as for populations. Aside from indicator species, many community EcoQOs would be for properties which are difficult to link to manageable human activities (e.g., diversity indices).	Substantial interest in use of benthic indicator species (see previous WGECO reports). This contains aspects of EcoQ Elements (o) and (p): Density of sensitive (e.g., fragile) species and density of opportunistic species; see Section 5.	Areas with density of *Virgularia* sp. (a sensitive species) at least xx% of historic mean regional abundance, not reduced by more than yy %

Feature	Property	Inter-pretable	Link to management	Link to Metric	Notes	Other Considerations	Examples
	Health	Y	Y	Y	Large numbers possible, but knowledge of regional history should guide selection of the most important EcoQOs readily available. Knowledge of concentrations to be used as baselines often limited.		*A low (<2) level of imposex in female dogwhelks, as measured by the Vas Deferens Sequence Index .*
Fish	Com-mercial popula-tions	Y	Y	Y	Most fisheries evaluations focus on spawning biomass and fishing mortality, but EcoQOs could be set for many other properties of commercial species, such as population structure, age composition, etc.	Monitoring of fishery catches, research surveys, and recurrent assessments should provide all necessary elements for EcoQOs	*SSB above precautionary [biomass, also taking into account fishing mortality] reference points for commercial fish species where these have been agreed to by the competent authority for fisheries management.*
	Non-com-mercial popula-tions.	Y	Y	Y	Large numbers are needed to give adequate coverage to the fish component of ecosystems. Not always easy to partition causes of changes in population status, so sometimes hard to link to manageable human activity.	Monitoring of total fishery catches and research surveys should provide information necessary for setting EcoQOs and evaluating population status relative to them. Should pick up aspects of Table A EcoQ Element (b) presence and extent of threatened and declining species – for the fish component.	Abundance of common skate in the [specify area] should not fall below xx percent of historic regional mean abundance.
	Comm.	Y? (Some)	Y? (Some)	Y? (some)	Same comments as for fish populations. More information than for benthic community indicators with regard to setting reference points, or at least reference directions.	Same comments as for fish populations. Should pick up aspects of Table A EcoQ Element (l) changes in the proportion of large fish, and hence average weight and maximum length of the fish community. See section 5.2.	Frequency distribution of maximum lengths reached by species in community [specify area] should not be reduced by more than xx% from distribution observed in surveys/sampling in [reference years]. Slope of the [specify area] community size spectra should not change by more than xx% from slope observed in surveys in [reference year].

Feature	Property	Inter-pretable	Link to manage ment	Link to Metric	Notes	Other Considerations	Examples
	Health	Y	Y	Y	Large numbers are possible, but knowledge of regional history should guide selection of the most important EcoQOs readily available. Knowledge of concentrations to be used as baseline often obtainable from historic samples.	Samples for monitoring performance relative to EcoQO standards usually readily available from fishery and/or surveys. Sometimes required for food safety anyway, so low incremental cost.	Dioxin levels in cod livers in [specify area] should not exceed xx $pg \cdot g^{-1}$.
Birds	Popula-tion	Y	N	N	Many causes of population fluctuations and temporary redistributions, so often difficult to link observed changes in human activities. EcoQOs needed for many species in order to provide adequate coverage to ecosystem.	Seabird monitoring programmes sometimes in place, but must be responsive to species annual behavioural cycles, so times when best population estimates can be obtained may not be most informative for linking to human activities. Should pick up aspects of Table A EcoQ Element (b) presence and extent of threatened and declining species – for the seabird component	Population of kittiwakes at [specify sites] during the breeding season does not fall below [specific number from historic monitoring] [xx % of historic baseline numbers]. Population of roseate tern should increase at least xx% by 2010, compared to population in 2000.
	Com-munity	Y	N	N	Same comments as for populations. Not clear if indicator species approach (preferred in Bergen Declaration) and community metrics (See WGECO 19xx] respond to same pressures in same ways.	Should pick up aspects of Table A EcoQ Element (k) seabird population trends as an index of seabird community health.	Could be same as kittiwake EcoQO in cell above, if kittiwakes are accepted as an indicator species for seabird community.

Feature	Property	Inter-pretable	Link to management	Link to Metric	Notes	Other Considerations	Examples
	Health	Y	Y	Y	Large numbers are possible, but knowledge of regional history should guide selection of the most important EcoQOs readily available. Knowledge of concentrations to be used as baseline sometimes obtainable from museum specimens or egg collections.	Should pick up aspects of Table A EcoQ Elements (g), (h), and (i) mercury concentrations in seabird eggs and feathers, organochlorine concentrations in seabird eggs, and plastic particles in stomachs of seabirds.	Organochlorine concentrations in bird eggs from [specify region] should not exceed [xx $\mu g \cdot g^{-1}$] [yy% of levels recorded in eggs between [give reference years]].
Mammals	Popula-tion.	Y	Y	Y	Often difficult to get reliable population estimates due to wide-ranging behaviour of some species, and unknown proportion of non-breeding individuals. Not always easy to partition causes of changes in population status, so sometimes hard to link to manageable human activity. Historic population estimates not available for some species, so reference points may be hard to set.	Monitoring programmes with reasonable accuracy and precision often expensive and difficult to implement. Should capture Table A EcoQ Elements (c) and (d) seal population trends in the North Sea, and utilisation of seal breeding sites in the North Sea. Are there threatened and declining marine mammals? If so that text goes here too.	*No decline in population size or pup production of > 10% over a period of up to 10 years.*
	Com-munity	Y	N	N		Relatively little study of community-scale dynamics of marine mammals in most regions	Aside from indicator species, hard to think of an informative community-scale EcoQO for marine mammals.

Feature	Property	Inter-pretable	Link to manage ment	Link to Metric	Notes	Other Considerations	Examples
	Health	Y	Y	Y	Large are numbers possible, but knowledge of regional history should guide selection of the most important EcoQOs readily available. Knowledge of concentrations to be used as baseline sometimes obtainable from historic samples. Bioaccumulation and slow turnover of many contaminants in marine mammals mean that response of concentrations to management actions will often be slow.	Substantial research interest in relationship between body burdens of contaminants and population dynamics parameters, but consequences often hard to demonstrate. Indicators of body condition (fat content of body, etc.) sometimes have been linked to onset of density-dependent population pressures.	Concentrations of PCBs in seal blubber for seals from [specify areas of particular concern] reduced by at least xx% from historic levels [give reference period] (where historic levels were considered a health risk and reductions necessary to avoid harm to population, or to subsistence hunters).

Table 7.2.2 EcoQOs and EcoQ Elements from Tables A and B of the Bergen Declaration which are not addressed in Table 7.2.1 with EcoQOs for State, or which readily illustrate addressing the same EcoQ Element with an EcoQO for a property other than a State. EcoQOs from the Bergen Declaration presently applied in a pilot project for the North Sea appear in italics.

EcoQO or EcoQ Element	D/P/S/I/or R	H(arm) or G(ood)	Inter-pretable	Link to Manage-ment	Link to Metric	Notes	Other Considerations
Annual by-catch levels [of harbour porpoise] should be reduced to levels below 1.7% of the best population estimate	I	H	Y	Y	Y?	Assumes that there are population estimates for porpoise on a regular basis.	Requires monitoring of both by-catches in fisheries and total population size for porpoise. Both programmes exist but are costly in resources.
The proportion of [oiled Common Guillemots] should be 10% or less of the total found dead or dying, in all areas of the North Sea	P	G	Y	Y?	Y	Considered an EcoQO to achieve good, because the low level of oiling is a desired state. No indication that *current* incidence of oiling is unsustainable.	Spatial scale on which 10% is calculated can have great impacts on how application of the EcoQO would affect conservation status of ecosystems and opportunities for human activities in the sea.
There should be no kills in benthic animal species as a result of oxygen deficiency and/or toxic phytoplankton species	I	H	Y?	N?	Y? (some)	Getting an N for manageable human activity would require a great deal of research on mortality sources for benthos. This is true in special cases, but not as a generalisation.	Some indicator species for some causes of benthic kills are known, but this would be a source of EcoQO only for situations in which the research base is exceptionally strong.
Local sandeel availability to black-legged kittiwake (j) expressed as "black-legged Kittiwake breeding success" should exceed 0.6 chicks per nest in each of the following coastal segments (see Section 4.2)	P	H	Y	Part	Y	Kittiwake breeding success varies with many factors, and evidence for overlap of kittiwake diet and fishery catches of sandeels is equivocal (see 4.2), so link to management measure is unclear.	Even if fishery was not the cause of the EcoQO to fail to be met, it might still be appropriate to apply spatial restrictions to the fishery to increase likelihood that the EcoQO could be met.

EcoQO or EcoQ Element	D/P/S/I/or R	H(arm) or G(ood)	Inter-pretable	Link to Manage-ment	Link to Metric	Notes	Other Considerations
Alternatives:							
SSB above precautionary [**fishing mortality**] reference points for commercial fish species where these have been agreed to by the competent authority for fisheries management.	I	H	Y	Y	Y	Contrast with the Fish Population row of Table 7.2.1. Same data and assessment sources as for the current biomass-based EcoQO.	Whereas Biomass is a State of the stock (and ecosystem), fishing mortality is an impact of the fishery. Otherwise the functional aspects of the SSB and F reference points are comparable.
Mobile fishing gears are excluded from operating in at least xx% of the areas classified as EUNIS level 4 type yy	P	H or G- depends on the percent	Y	Y	Y	Contrast with the Habitat row of Table 7.2.1. Links to management measures and metrics are direct and simple.	Monitoring requires only information on area of vessel operations, from, say VMS.
Hours fished by fleet xx do not exceed yy.	P	H or G depends on the percent	Y	Y	Y	Contrast with all the fish and benthic population and community rows, and the habitat row in Table 7.2.1.	Monitoring requires method of tracking fishing effort. Effort is assumed to be known in many current stock assessments (CPUE).
The Shannon-Wiener diversity of the fish community in [specify area] does not show any trend over time	S	H or G depends on view of current state	N?	N	Y	Contrast with fish community row of Table 7.2.1. Diversity index removes need for long lists of species, but poses problems for consistent interpretation by non-scientists, and difficult to link to management actions.	

In fact, as noted in Section 7.1, when the DPSIR framework was introduced, the boundaries between Pressure, State, and Impact (at least) are not rigid nor always clear. Not only can "Catch" be viewed as State from the perspective of the fishery and Pressure from the perspective of the exploited population, the ratio Catch to Population is clearly an Impact. Within WGECO, some members even argued that EcoQOs for Indicator species of a human activity are not State EcoQOs at all. Rather they are Pressure EcoQOs, as they are *de facto* often the only metrics available of the actual Pressure being inflicted by the human activity on the ecosystem. Some of these issues may be semantic, but at least some are conceptual. In either situation, a common understanding of what is a Pressure, a State, and an Impact must be developed among the science and management communities, and with the public, if EcoQOs are to be a central tool in achieving conservation and sustainable use.

7.2.5 Conclusions

Although this evaluation is preliminary, and only addresses one of the ten combinations of DPSIR with avoiding harm or achieving good, WGECO thinks that the approach has merit. The EcoQO approach is under discussion for more complete development as a central part of an ecosystem approach to management of human activities in the seas (EC, 2004). Central to the success of such an approach will be the wise and effective selection of the suite of EcoQOs to be used in guiding management. This is the first framework WGECO has examined which may have promise not just to select individually sound EcoQOs and associated metrics, but to identify suites of them which work together effectively and efficiently.

WGECO encourages all parties involved in those discussions to consider these preliminary results, and particularly the approach taken to achieve them. Despite working seriously on the scientific basis for EcoQOs since the mid-1990s, WGECO has gained some new insights from its preliminary exploration of this approach. WGECO would welcome a Term of Reference for its next meeting to complete the initial application of this framework as a basis for developing advice on integrated suites of EcoQOs and metrics, rather than just lists of them.

7.3 Collapsing EcoQOs into aggregates

This is a complex subject, and WGECO did not have time to address it in detail. There is relevant material in Section 11, on aggregate indicators of habitat quality, where some strengths and weaknesses of such metrics are discussed.

The issue of collapsing a number of EcoQOs into a single aggregate EcoQO requires facing the trade-off between the complexity of trying to interpret large amounts of information (reporting on multiple EcoQOs), and the risks inherent in collapsing information in ways which may be misleading in their apparent simplicity. Truly aggregate EcoQOs would be likely to have acceptable status when some constituent EcoQOs are being met well and others are not being met at all. Such EcoQOs could guide management towards states where some ecosystem properties are in very good condition but others are in highly undesirable condition.

Aggregate EcoQOs are likely to be difficult to relate to management actions as well. Keeping the ecosystem effects of fishing within sustainable bounds requires multiple operational objectives supporting dialogue or decision-making (ICES 2001, 2003; FAO, 2002). Many management actions are likely to have different effects on the probability of achieving several of them at once; for example, restricting the places where fishing can occur redistributes effort so by-catch rates of some species may be reduced, but by-catch of others can be increased (Jennings *et al*., in press). Hence, the directional pressure of fishing may facilitate achievement of some EcoQOs in the aggregate while moving other EcoQOs contributing to the aggregate in the opposite direction. The value of the indicators for the aggregate EcoQO will not be very informative, a consequence called the "eclipse" effect. Experienced science advisors and managers usually know to be vigilant for such obvious conflicts in the directional responses of effects of fishing on different target species in a mixed fishery, but the expected patterns of response of EcoQOs for many other ecosystem properties are not that obvious (e.g., EcoQOs regarding species diversity). Several of the presentations at the recent Symposium on Ecosystem Indicators for Fisheries Management (30 March–3April in Paris), which ICES co-sponsored, demonstrated that danger for a variety of types of ecosystem indicators and ecosystems. (Abstracts and Powerpoint presentations available at http://www.ecosystemindicators.org/). Hence, there is a risk that aggregate EcoQOs could hide effects of fishing, or ineffectiveness of management, when managers should be taking such effects into account in their decision-making.

When the EcoQOs are used in formal decision support, the fact that there are likely to be many associated indicators means that there will be correspondingly many inputs to the system of decision rules. There is little formal guidance available on how these many inputs should be treated in a decision system with multiple indicators, reflecting multiple EcoQOs. For example, Annex III-B of the Bergen Declaration includes five indicators of eutrophication. They are accompanied by a footnote saying "The ecological quality objectives for elements (m), (q), (r), (t), and (u) are an integrated set and cannot be considered in isolation", but no guidance is provided on how that is to be achieved. The

Precautionary Approach (FAO, 1996a, 1996b) could be interpreted as requiring management action to be matched to the EcoQO and indicator with the highest risk of being at or outside its conservation reference point. WGECO agrees with this view, as long as the EcoQOs relate to preventing serious or irreversible harm. However, for EcoQOs intended to achieve desired states, analytical risk management approaches might be interpreted as indicating that an overall risk profile should be built up across all the EcoQOs and their indicators, and it is that risk, not each constituent risk, which should be managed. This would present a major challenge in practice, when our analyses repeatedly have concluded that for many important ecosystem components, we lack basic data to even select reliable indicators and set reference points, let alone undertake formal risk analyses.

There have been diverse views expressed in some ICES expert groups on the subject that an EcoQO for SSB of commercial fish stocks, for example, might be to have some specified percentage of stocks above their precautionary reference points. The particular example is emphatically not endorsed by WGECO, because the $\mathbf{B}_{pa}$ and $\mathbf{F}_{pa}$ were both chosen relative to conservation limits, and when expressed in EcoQOs, they are intended to avoid serious or irreversible harm. Therefore, 100% of stocks must comply with their precautionary reference points, for minimum single-species conservation standards to be met. The only appropriate aggregate EcoQO would be for 100% of stocks to have SSBs above their $\mathbf{B}_{\mathbf{pa}}$s and fishing mortalities below their $\mathbf{F}_{\mathbf{pa}}$s.

Objective-based management decision-making can give managers structured insight into the likely effects of alternative management actions, which is essential in integrated management approaches. However, this is only true if the performance characteristics of the indicators associated with the EcoQOs are understood and their trends and current values relative to reference points are presented and interpreted correctly. This is likely to be diminished, rather than enhanced, by aggregating multiple EcoQOs into a few more highly inclusive ones. This is a particularly compelling reason to attempt a formal screening of the performance properties of candidate ecosystem objectives and indicators, as outlined in past WGECO work, even if the actual choices are to be made by partisan political processes rather than objective, scientific ones.

7.4 Decision-making in the context of Multiple EcoQOs

WGECO views this task differently, depending on whether the EcoQOs are intended to achieve societal desires or to avoid harm.

For EcoQOs intended to avoid serious or irreversible harm to ecosystem components, guidance from major international agreements such as Rio and Johannesburg is clear. The precautionary approach states that *all* ecosystem components should be protected from such harm. Therefore, however numerous the list, as long as **any** EcoQOs intended to avoid harm to ecosystem components are not being achieved, there is a serious conservation concern. Moreover, there is no ecological reason why all parts of a well-managed and sustainably used ecosystem should not be healthy enough to meet the standards of EcoQOs intended only to avoid serious harm.

From the ecological perspective management **must** give priority to measures likely to move towards achievement of the EcoQO as quickly as possible. (The international agreements differ in the weight given to “cost effectiveness” and social and economic considerations in choosing the speed with which EcoQOs intended to avoid harm must be met [Table 2 in Rice, in press], but for WGECO, the ecological perspective is the relevant consideration.)

For EcoQOs intended to achieve states of the ecosystem consistent with desired uses by society, the situation is nothing more than the usual situation of multi-criteria decision-making. ICES recently sponsored a major international symposium on this topic (Symposium on Confronting Uncertainty in the Evaluation and Implementation of Fisheries Management Systems, Cape Town, South Africa, 1998 (ICES, 1999). Although many of the contributions to that symposium addressed management to make balanced progress on social or economic and ecological goals, the concepts and approaches are directly applicable when management is striving to achieve multiple ecological EcoQOs at once, as long as all the EcoQOs are aspirational. The risk management tools and approaches in that volume and related work are a sound guide to management to achieve EcoQOs implemented to reach desired ecosystem states.

For both types of EcoQOs, under most circumstances decision-making will become more complex rapidly as the number of EcoQOs to consider increases. Moreover, unless the goals of all the EcoQOs are very similar, the increase in complexity is likely to be closer to multiplicative than additive with additional EcoQOs. For this reason, the evaluation begun in Section 7.2 should be completed as a priority. If our preliminary conclusion holds up to further analysis, if EcoQOs of ecosystem State are used as the approach to guide management towards sustaining healthy ecosystems, it will be necessary to use very large numbers of EcoQOs. Hence, decision-making will be very complex and difficult to guide with EcoQO-related management rules. However, if EcoQOs for Pressure and possibly some Impact and even Response properties are employed in management, many fewer EcoQOs may be needed to provide a desired level of protection of the ecosystem. This, in turn, would result in management with lower complexity, and with EcoQOs often linked very closely to management actions needed for remediation of any discrepancies.

There are important ecological considerations that require more exploration, though, with regard to the possibility of achieving a large number of ecosystem targets at once. For example, the discussion in Section 3.3 illustrates that because of predator-prey interactions among harvested species, it is not possible to obtain the maximum sustainable yield from all species simultaneously. Trade-offs would have to be made among such yield-based EcoQOs, but the risk management frameworks to explore the consequences of the trade-offs are known. The species interactions summarised in Section 3.3 are not simple, but they are simpler (or at least better studied) than many of the other ecological interactions that will have to be addressed when pursuing large numbers of aspirational EcoQOs simultaneously. Much work remains to be done in this area, although there is an important message to guide management under multiple EcoQOs at present. **Achievement of all the aspirational EcoQOs remain subordinate to achieving any EcoQOs intended to avoid serious or irreversible harm**.

7.5 References

Degnbol, P. 2003. Science and the user perspective: the gap co-management must address, p. 31-50. In D.C. Wilson, J. Raakjær Nielsen and P. Degnbol (eds.) The fisheries co-management experience: accomplishments, challenges and prospects., Fish and Fisheries Science No. 26, Kluwer Academic Publishers.

Degnbol, P. 2004. Indicators as means to communicate knowledge: institutional implications. MS, presented at the SCOR/IOC Symposium on “Quantitative ecosystem indicators for an ecosystem approach to management”, 31 March – 3 April 2004, UNESCO, Paris, France. Presentation available at www.ecosystemindicators.org, last accessed on 21 April 2004.

EEA 2003. Environmental Performance Indicators for the European Union. http://themes.eea.eu.int/indicators/all_indicators_box

EC (2004) Guidance on how to develop the ecosystem approach to managing human activities in the marine environment. (Draft) EAM(2) 04/3/1 Brussels.

FAO. 1996a.

FAO. 1996b.

FAO. 2003. Fisheries Management. 2. The Ecosystem Approach to Fisheries. FAO Technical Guidelines for Responsible Fisheries No. 4, Suppl. 2. Food and Agricultural Organisation of the United Nations, Rome, Italy.

ICES. 1999. Confronting Uncertainty in the Evaluation and Implementation of Fisheries Management Systems. ICES Journal of Marine Science, 56(6): 795–1074.

IIED 2002. National Strategies for Sustainable Development: the pressure – state – response framework. www.nssd.net/References/SDInd/PST.html

Jennings *et al*., in press

OECD. 1993. OECD Core Set of Indicators for for Environmental Performance Reviews. Paris.

Rice, J.C. in press. Challenges, Objectives, and Sustainability Benthic Community, Habitats and Management Decision-Making. Proceedings of Tampa Symposium on

Rice and Rochet. 2004. A framework for selecting a suite of indicators for fisheries management. Paper presented at SCOR Symposium on Ecosystem Indicators for Fisheries Management and in review for ICES journal.

Rochet and Trenkel – CJFAS.

Shin, Y.-J., M.-J. Rochet, S. Jennings, J.G. Field and H. Gislason. 2004. Using size-based indicators for an ecosystem approach to fisheries. MS, presented at the SCOR/IOC Symposium on “Quantitative ecosystem indicators for an ecosystem approach to management", 31 March – 3 April 2004, UNESCO, Paris, France. Presentation available at www.ecosystemindicators.org, last accessed on 21 April 2004.

Trenkel and Rochet 2002

UNEP. 2000. Global Change of Infoterra to UNEP-Infoterra: Enabling Countries to Build an Integrated Environmental Information Service. UNEP/INFO2000/WP/3.

8 IMPACTS OF INDUSTRIAL FISHING (TOR F)

The term of reference states:

f) complete the work started in 2003 in response to the EC request on ecosystem impacts of industrial fishing:

i) summarise information from relevant Expert Groups (Assessment Working Groups, SGDBI, WGFE) and prepare a compilation of the scientific information in response to this request,

ii) consider which aspects of this request require further work and propose plans to take forward such work.

WGECO was asked to look at the issue of the Baltic fisheries for industrial purposes and their direct and indirect effects on the ecosystem (ICES letter to Mr. John Farnell; EC DG-Fisheries, 3 December 2003. Ref. G.16/DG/MM).

WGECO also took the opportunity to review and comment further on the ecosystem impacts of industrial fisheries in the Blue Whiting fisheries, reporting new information.

WGECO was asked to comment further on the specific request from the EC to "Evaluate the relative benefits (in terms of economic and of ecological efficiency) of fishing 'industrial' fish for fish meal and using the product as feed, or of not fishing these species and obtaining higher yields from commercial fisheries." WGECO is not in a position to deal with a request for an evaluation of the economic efficiency in relation to the relative benefits of industrial fishing. However, it will deal with the ecological aspects of the issue (ICES letter to Mr. John Farnell; EC DG-Fisheries, 3 December 2003. Ref. G.16/DG/MM).

8.1 Baltic Ecosystem

The Baltic Sea is a semi-enclosed body of brackish water, occupying an area of about 412,600 km^2 and having a volume of about 21,600 km^3 (Figure 8.1). The sea is virtually non-tidal and receives a larger amount of fresh water via rivers and precipitation than it loses via evaporation, resulting in a surplus of fresh water leading to brackish conditions. The environmental conditions in the Baltic and their variability are strongly linked to the meteorological, hydrological, and hydrographic processes and their interactions. All these processes influence the temperature and ice conditions, inflow of fresh water from rivers, exchange of water between various Baltic Sea sub-basins and with the Skagerrak-Kattegat system and the resultant transport and mixing of water inside the Baltic Sea are complex. Broadly, stratification in the Baltic Sea is controlled by salt-water intrusions and river run-off. A marked halocline is found at 60–80 m depth in the Baltic Proper, and this hinders oxygenated water from the overlying layers of water from mixing downwards in the water column. Thus, the Baltic is characterised by anoxic conditions in its deep basins. Aperiodic inflows of highly saline and oxygenated water from the Kattegat into the Baltic Basins are characterised by two phases: (1) high pressure over the Baltic region with easterly winds, followed by (2) several weeks of strong zonal wind and pressure fields over the North Atlantic and Europe (Schinke and Matthäus, 1998). These inflows have significant effects on the biology and ecology of the species in the Baltic Sea, as many species are under permanent physiological stress due to the strong salinity and oxygen gradients.

Species diversity in the Baltic Sea is low (Hopkins, 2000; Pedersen and Snoeijs, 2001; Voipio, 1981), which is mainly due to the physiological stresses in the environment.

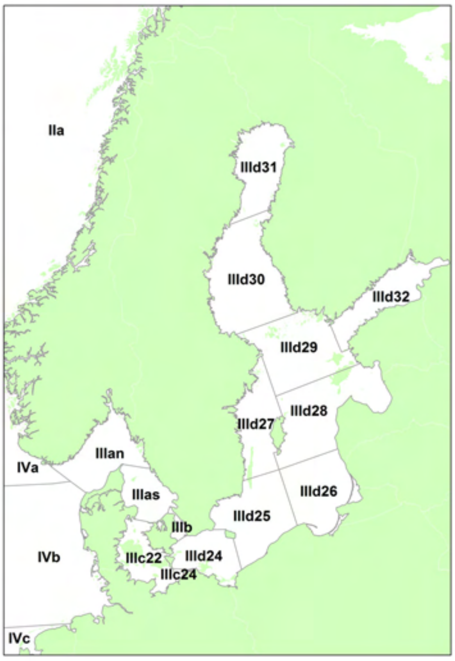


Figure 8.1. Map of the Baltic system and the ICES Subdivisions.

8.1.1 Baltic industrial fisheries

There are two principal industrial feed fish fisheries in the Baltic:

1) A sprat fishery which takes large amounts of herring as by-catch. This fishery is considered a mixed fishery. However, sprat catches in the deep basins of the central Baltic are relatively clean of herring.

2) A directed herring fishery which can take sprat as by-catch depending on the region of the Baltic being fished.

It is difficult to state categorically that herring and sprat in a specific fishery or area go to fish meal/oil production or to human consumption, as the ultimate use of the fish is driven to a large extent by market forces (Bengt Sjöstrand, pers. comm.).

8.1.1.1 Description of the industrial fisheries

8.1.1.1.1 Fisheries for sprat/herring

Sprat (*Sprattus sprattus*) are distributed through most of the Baltic Sea and are generally regarded as a single stock. Sprat spawn in open waters in the Baltic Proper, the Gulf of Riga, and the Gulf of Finland between March and August. The Gulf of Bothnia is not saline enough for sprat to spawn in.

The sprat stock is managed by one TAC agreed for the whole Baltic, and ICES classifies the stock (in Subdivisions 22–32) as being inside safe biological limits (ICES, 2003b). Denmark, Estonia, Finland, Germany, Latvia, Lithuania, Poland, Russia, and Sweden exploit sprat in the Baltic Sea. During the 1990s, total catches increased considerably, from 85,000 t in 1990 to 530,000 t in 1997. The increase in catches since 1992 is due to the development of the industrial pelagic fishery. In 2002, sprat catches were very similar to the 2001 level, amounting to 343,200 t. Poland took 81,200 t, Sweden took 77,300 t , Latvia took 47,500 t, and Denmark and Estonia took in the region of 41,000 t. Landings were, however, 35% less than the record high level of 530,000 t in 1997. In 2002, the TAC of 380,000 t set by IBSFC was not realized. Trawls operating demersally and pelagically account for most of the catches. In general, it is reported that the sprat fishery is prosecuted by trawls with mesh size <32 mm.

Fished sprat can be utilised for either human consumption or to feed, but it is mainly directed to fish meal/oil production in Danish, Polish, and Finnish landings. In Russia and Latvia, sprat is generally used for human consumption (Andrey Dolgov, pers. comm.) Recently, patterns of human consumption of sprat (and herring) have changed due to the perceived problems with dioxins (Section 8.1.8).

8.1.1.1.2 Fisheries for herring

Herring (*Clupea harengus*) is managed by two TACs: SD 22–29S, 32 and SD 29, 30, 31. The fishing mortality on herring has increased throughout the late 1990s (STECF, 2004). ICES classifies the stock in the Gulf of Riga as being inside safe biological limits, whereas herring in Subdivisions 25–29 and 32 (excluding Gulf of Riga herring) is considered to be harvested outside safe biological limits (ICES, 2003b; STECF, 2004). Landings in 2002, in Subdivisions 25–29 (excluding Gulf of Riga) and Subdivision 32, were reported at 129,300 tonnes, Sweden took 29,400 t, Poland took 28,800 t, Finland took 25,700 t, and Estonia 21,000 t. Russia, Denmark, Germany, Latvia, Lithuania took, in total, 24,500 t (ICES, 2003a). Harvey *et al*. (2003) determined that fishing was the chief source of mortality on herring using ECOPATH/ECOSIM modelling. However, a reduction in the biomass is not solely attributed to fishing mortality, but also, for example, to extrinsic drivers affecting recruitment (e.g., Harvey *et al*., 2003; Rahikainen *et al*., 2003). The herring trawl fisheries by the various countries are generally prosecuted using a mesh size of 32–40 mm using bottom and pelagic trawls (ICES, 2000a, 2003a). A fraction of the landed herring is taken with trap nets/pound nets and gillnets during spawning time and with insignificant by-catches of sprat (ICES, 2003a).

Poland, Sweden, Denmark, Finland, and other Baltic States target herring, and the species is used for either fish meal/oil production or human consumption. In Russia, herring is generally used for human consumption purposes, while in Sweden it is used for either fish meal/oil production or human consumption depending on the market condition (Bengt Sjöstrand, pers. comm.). Only a few Finnish vessels are reported to take herring exclusively for human consumption and approximately 70% of landings are used in animal feed (ICES, 2000a). Herring taken by Denmark and Poland usually goes to fish meal/oil production (EP, 2004).

8.1.2 Effects of industrial fisheries

8.1.2.1 Direct effects of industrial fisheries on fish

In general, the sprat and herring fisheries are thought to have little or no discarding of any by-caught species (ICES, 2000a) since the volume of catches (300–400 tonnes per trip) precludes sorting. There are anecdotal reports that the fishing skippers will change fishing grounds to avoid by-catches of juvenile cod (ICES, 2000a).

It was originally planned to estimate by-catch rates in the Baltic Sea industrial fisheries, and subsequently raise these figures to total weight of by-catch by species, age and fishery, using the common database BALTCOM. However, it was discovered that this database was not yet at a stage to allow for such calculations. Firstly, only a small fraction of the data that have been collected to estimate species compositions in the pelagic fisheries have been entered in the database. Secondly, the analyses that were planned to be carried out would, given the stage of the database, have required access to the raw data. In addition to these problems, fundamental deficiencies in the data available were discovered, i.e., only part of the landing data had been uploaded, which prevents a proper assessment of the effect of industrial fisheries on the by-catch species in the Baltic Sea.

A major problem is to distinguish between the fraction of the catches in the pelagic fisheries that are landed for human consumption and the fraction that are used for reduction. These fractions show large variations between countries and are also influenced by the market situation. This variation is not believed to be reflected in the official landing statistics. Consequently, distinguishing between pelagic landings for human consumption and reduction is presently not possible for the pelagic fisheries in the Baltic Sea as a whole. Furthermore, detailed information about the species composition in the pelagic fisheries is only sporadically reported and is likely to be biased for some of the fisheries. For example, ICES (2003a) states that the separation of herring and sprat in the catches in the mixed fishery for herring and sprat is imprecise. Further, due to misreporting of sprat and herring landings, WGBFA (ICES, 2003a) recommends that species compositions of the landed pelagic fish are historically re-evaluated/revised at national level. The proportion of herring in landings and in the Baltic Sea surveys is highly variable between countries and subdivisions, indicating that information on a detailed temporal and spatial level regarding species compositions in the pelagic fisheries will have to be available for all the countries to be able to estimate the total by-catch in these fisheries.

Due to lack of data, therefore, it is presently not possible to make an assessment of the effect of industrial fisheries on the by-catch species in the Baltic Sea. A description of the information available about by-catches in the Baltic Sea pelagic fisheries is presented and discussed below.

8.1.2.1.1 Direct cod by-catch in the sprat/herring fisheries

Germany carried out an experimental herring fishery in Subdivision 24 utilising pair trawls with a mesh size of 32 mm and reported that low by-catches of undersized cod were found only in shallow waters (ICES, 2000a). However, no specific quantifiable information was available to this group.

ICES (2001a) utilised the International Baltic Sea Sampling Programme (IBSSP) to assess cod by-catches in total pelagic fisheries in the Baltic and calculated the values of by-catch and discard of cod in combined weight of herring and sprat samples, stratified by year, country, quarter, and subdivision. The spatial and temporal distribution of cod by-catch in the herring and sprat fisheries in the Baltic was thought to relate to the co-occurrence of the three species on cod and sprat pre-spawning and spawning grounds. Between the years 1998–2000, the highest by-catches of cod in the herring and sprat fisheries were observed in the first and second quarters. The highest by-catch occurred in SD 24–26. The estimated total by-catch of cod in combined sprat and herring fisheries in the years 1998, 1999, and 2000 amounted to 1340 t, 1524 t, and 2091 t, respectively. ICES (2001a) determined that the total share of by-catch in total landings of cod was within the range of 1.3% to 2.0%. The by-catch in pelagic fisheries, therefore, appeared to have a minor effect on the cod population. ICES (2001a), however, expressed concern that it was not possible to evaluate how much cod by-catch in the official catch statistics were recorded and these results can only be considered as tentative.

8.1.2.1.2 Herring exploitation in the sprat/herring fishery

The fishery in Subdivisions 22–32 directed at sprat is known to have a by-catch of young herring which has been estimated at up to 35% by weight (reported in STECF, 2004). Finland reported that in the directed sprat fishery (Subdivisions 26, 28) more than half the herring landings are taken as by-catch in the directed fishery (ICES, 2000a).

8.1.2.1.3 By-catches in the Danish pelagic fisheries in the Baltic (DIFRES, unpublished information)

The Danish pelagic fisheries in the Baltic are categorised into two fleets: industrial and human consumption. The categorisation is made on the basis of the mesh-size used: vessels fishing with mesh sizes of 32 mm or less are regarded as fishing for industrial purposes (i.e., for reduction to fish meal and oil), whereas those with larger mesh sizes are regarded as fishing for human-consumption purposes. These classifications are not exact, as catches made by the "human consumption" fleet are sometimes used for meal and oil production depending on the market price and condition of the catch. Catches by fleet/month/subdivision/species have been estimated for these fisheries for 2000 and 2001, although these landing figures are DIFRES estimates and not officially reported figures.

The estimates of the total catches by species are based on the reported species composition which is verified by samples taken for enforcement purposes. Details of samples taken for this purpose in 2000 and 2001 are summarised in Table 8.1.2.1.3.1. The large majority of samples are consistent with a targeted sprat fishery, with catches in excess of 80% sprat; indeed, more than 50% of the samples in both years consist of at least 95% sprat. Of the remaining samples, a few can be attributed to a directed herring fishery, or to a small-bait fishery for sandeel to the east of Bornholm. This leaves nine samples which are not attributed to any of the other categories. Details of these samples are given in Table 8.1.2.1.3.2.

Of the uncategorised samples, the April 2000 sample is consistent with a directed herring fishery, as in excess of 88% of the sample consisted of herring. The six samples from Subdivision 22 contained variable proportions of sprat and herring as well as a small whitefish by-catch. The fishery in Subdivision 22 may thus best be characterised as a mixed pelagic fishery. The minimum legal mesh size in this area is 32 mm. This subdivision accounted for 10.3 % of the total Danish landings of herring and sprat in 2000, and 22.5% in 2001. Of the other two uncategorised samples, one came from Subdivision 25 in May 2001. Three other samples are available from Subdivision 25 in May 2001. These all indicate catches in excess of 90% sprat, indicating that "mixed" catches may be unusual from this area and season. The remaining sample came from Subdivision 28 in December 2000. No other samples are available from Subdivision 28 in December; however, all other samples from Subdivision 28 indicate catches of sprat in excess of 95%.

On average, 1.1 samples were obtained per thousand tonnes of sprat and herring landed in 2000 and 1.2 were obtained in 2001. Details of the age sampling are given in Table 11.2.2 in ICES (2003a). According to the EU data directive, a minimum of 1 sample per 2000 tonnes landed must be collected. The Danish sprat fishery in the Baltic Sea takes place over a relatively short period and a small number of large vessels take by far the largest proportion of the Danish catches.

In summary, the evidence from sampling in 2000 and 2001 indicates that most Danish pelagic catches in the Baltic are taken in targeted fisheries for sprat or herring, with relatively low by-catch of the other species. The clear exception to this is Subdivision 22, where all samples indicate that the fishery in that area should be regarded as a mixed pelagic fishery.

Table 8.1.2.1.3.1. Summary of species compositions from samples from Danish pelagic catches reported from the Baltic in 2000 and 2001.

No. samples	2000	2001
No. with 95% or more Sprat	33	22
No. with between 90 and 95% Sprat	11	5
No. with between 80 and 90% Sprat	3	6
No. with 90% or more Herring	3	1
No. with 90% or more Sandeel	5	1
Others (see Table 8.1.2.1.3 .2)	2	7
Total	57	42

Table 8.1.2.1.3.2. Details of samples described as "others" in Table 8.1.2.1.3.1.

Year	Month	SD (square)	% Sprat	% Herring	% Other
2000	4	24 (37G2)	10.6	88.7	0.6 (cod)
2000	12	28 (42G8)	74.7	25.3	-
2001	4	22 (37G1)	10.4	72.4	17.2 (whi+cod)
2001	4	22 (38G0)	17.8	65.0	17.2 (whi+cod)
2001	4	22 (37G1)	45.9	49.7	4.4 (whi+cod)
2001	5	25 (39G5)	68.5	30.3	1.2 (cod)
2001	8	22 (40G0)	78.7	16.9	4.1 (whi+cod)
2001	8	22 (39F9)	14.7	76.3	9.0 (whi+cod)
2001	10	22 (37G1)	69.6	29.0	1.3 (whi+cod)

SD = Subdivision; whi = whiting

8.1.2.1.4 Conclusions

The issue of by-catch and assessment of the species composition in the Baltic Sea is complex and, when coupled with the apparent problems with landing data and misreporting of pelagic fish catches, it is impossible at this time to quantify by-catch. For example, the detailed information supplied by DIFRES shows that in SD22, percentages of sprat in 2001 ranged from 10.4–78.75% in the samples; however, the landing data available to the Working Group were aggregated at a coarse level (for all countries) that did not allow further analysis. It cannot be stated that the conclusions presented for the short-term Danish pelagic fisheries and the report on cod by-catch in the Baltic Sea are representative for the rest of the pelagic fisheries in the Baltic Sea. Detailed data on species composition and the catches are needed in order to analyse the direct effects of fishing for industrial species in the Baltic Sea.

8.1.2.2 Indirect effects of industrial fisheries on fish

The Study Group on Multispecies Assessment in the Baltic (ICES, 2003e) has summarised the main feeding relationships in the Baltic Sea. The fish community in the open parts of the sea is dominated by just three species: cod, herring, and sprat. The abundance of the cod stock in the Main Basin is currently low, herring stocks are decreasing, and the sprat stock is at a high level. The effect of cod on prey species (herring and sprat) is therefore now at a low level. While cod biomass is low, there is the potential for herring and sprat to have an adverse effect on cod recruitment, through consumption of cod eggs and larvae. Predation mortality of sprat showed a continuous decline from the mid-1970s to the early 1990s, then levelling off. The trend in predation mortality of herring follows closely that described for sprat, but has never been as high as that for sprat. On this basis, it would seem unlikely that industrial fisheries for the prey species (sprat, herring) are negatively affecting the predator (cod); indeed, it is possible that the inverse is true, as fisheries for sprat and herring might reduce consumption of cod eggs and larvae.

8.1.2.3 Effects on seabirds

8.1.2.3.1 Direct effects of fisheries for Baltic sprat/herring on seabirds

WGSE reviewed pressures on seabirds in the Baltic in 2004 (ICES, 2004). By-catch in industrial fisheries was not viewed as a pressure on seabird populations. A number of national or regional investigations of seabird by-catch have been undertaken, focusing mainly on gillnet fisheries (Hario, 1998; Urtans and Priednieks, 2000; Dagys and Zydelis, 2002; Stempniewicz, 1994; Kirchhoff, 1982; Schirmeister, 2003). There are no records of by-catch of seabirds in any form of industrial fishery in the Baltic.

8.1.2.3.2 Indirect effects of industrial fisheries on seabirds

There have been no studies of possible indirect effects of industrial fishing in the Baltic. There are few studies of seabird diet in the Baltic, with most records being anecdotal. One study in the mid-1990s of common guillemots (*Uria aalge*) by-caught in gillnets showed that the dominant prey item was sprat (Lyngs and Durinck, 1998). It has been suggested that cod exhibit a top-down control on sprat biomass (Harvey *et al*., 2003). Given the current low biomass of cod in the Baltic, there is probably a reduction in predation on sprat; indeed, the biomass of sprat in the Baltic is high (ICES, 2003b) and is probably limited by bottom-up effects. It is likely that seabirds in the Baltic are not affected by the industrial fisheries mortality on sprat.

8.1.2.4 Effects on marine mammals

8.1.2.4.1 Direct effects of industrial fisheries on marine mammals

WGMME reviewed the status of marine mammals in the Baltic in 2003 (ICES, 2003f). No by-catch in any form of industrial fisheries has been recorded, either for seals or harbour porpoise. A further review by ASCOBANS (Kaschner, 2003) of by-catch of harbour porpoise found no records of that species being by-caught in industrial fisheries in the Baltic.

8.1.2.4.2 Indirect effects of fisheries Baltic sprat/herring on marine mammals

No studies of marine mammal diet in the Baltic were found in the time available to WGECO to allow assessment of the indirect effects of the fisheries for sprat/herring.

8.1.2.5 Effects on seabed habitats and benthos

The towed demersal gear used in the prosecution of both the mixed sprat and the herring fishery comes into contact with the seabed, and almost certainly disturbs the sub-surface sediment layers. This may have physico-chemical and biological implications likely to affect the ecosystem. However, observed changes in the benthic habitat as a result of demersal gears used in the industrial fisheries have not been demonstrated. An analogous study (Rumohr and Krost, 1991) documented the biological responses of the invertebrate community in the Kiel Bay (Western Baltic) to otter trawling activity; thin-shelled bivalves (*Abra alba*, *Mya* spp., and *Macoma calcarea*) and starfish (*Asterias rubens*) showed damage as a result of trawling, but the thick-shelled bivalves (*Astarte borealis* and *Corbula gibba*) seemed to be more resistant to mechanical stress caused by bottom-trawling. Studies are needed to quantify the impact of the industrial fisheries in the Baltic region on seabed habitats and benthos.

8.1.3 Dioxins in Baltic sprat and herring

Dioxins refer to a group of chemical compounds that share certain chemical structures and biological characteristics. Several hundred of these compounds exist and are members of three closely related families: the chlorinated dibenzo-*p*-dioxins (PCDDs), chlorinated dibenzofurans (PCDFs), and certain polychlorinated biphenyls (PCBs). The toxic effects of these substances include carcinogenic potency, immunosuppression, and reproductive toxicity (Baars *et al*., 2004). They are lipophilic compounds and accumulate in the food web (Karl *et al*., 2002).

The European Commission has endorsed the World Health Organization's initiative of establishing maximum dioxin levels (EC Directive 102/2001). The Directive contains the corresponding action values, setting limits for dioxins in fish oils, fish meal, and fish feeds (Table 8.1.3.1). The levels established apply to both feed fish and fish destined for human consumption.

Table 8.1.3.1. Limit and action values for dioxins in feedingstuff

Feedingstuff	Maximum dioxin content relative to a feedingstuff with a moisture content of 12% (in ng kg^{-1})	
	Limit value	**Action value**
Fish oil	6.00	4.50
Fish, their products and by-products	1.25	1.00
Compound feeds	0.75	0.40
Feeds for fish	2.25	1.50

Source: EC Directive 102/2001 and EC Directive 2002/32

The measured dioxin concentrations in herring older than four or five years old have been above the level allowed by the European Union for human consumption (4 pg WHO PCDD/DF TEQ per g fresh weight) (reported in ICES, 2003a). If the dioxin content remains at high levels, it is anticipated that after the year 2006, the herring of the Bothnian Sea will no longer be able to be utilized for human consumption (ICES, 2003a).

There are implications regarding utilising herring and sprat for the production of fish meal and fish oil. The limits in Table 8.1.3.1 mean that a product such as fish oil or fish meal with a contamination level above the corresponding maximum limit will not be allowed for use in the production of feedingstuffs (e.g., fish with a contamination level of above 6 ng kg^{-1} or fish and fish meal with a contamination level above 1.25 ng kg^{-1} whole weight). PCBs, PCDDs, PCDFs, and other dioxin-like compounds have been found in sprat and herring in the Baltic (Strandberg *et al*., 1998; Vuorinen *et al*., 2002). Sprat and herring from regions of the Baltic have the highest dioxin content compared to fish taken from other fishing grounds (regions of the North Sea, Ireland, and Norway) (Karl *et al*., 2002; Vuorinen *et al*., 2002). Sprat has a higher reported concentration of dioxins than herring, especially in the oil (Vuorinen *et al*., 2002). Fish oil provides an important nutritional supplement in fish pellets, and the concentrations of dioxins in lipid are greater than the fresh weight (Vuorinen *et al*., 2002). Fish oils with greater than 6 ng kg^{-1} have been reported (EP, 2004; Karl *et al*., 2002), necessitating expensive carbon filtration methods to remove the dioxins.

Kiviranta *et al.* (2003) reported that there has been no observed decrease in the dioxin content in Baltic herring, and the source of exposure of herring to PCDD/Fs and PCBs, air-zooplankton versus sediments-zooplankton or sediments-crustaceans, remains obscure.

If the long-term objective is to assure the usability of sprat and herring catches for human consumption and fish feed uses, and to reduce toxin delivery to higher predators in the ecosystem, the dioxin content should ideally be reduced in small pelagic species. The source of exposure to herring and sprat needs to be identified, and the anthropogenic source(s) of dioxins controlled. There is a HELCOM Objective (with regard to Hazardous Substances) in place to prevent pollution of the Convention Area by continuously reducing discharges, emissions, and losses of hazardous substances towards the target of their cessation by the year 2020, with the ultimate aim of achieving concentrations in the marine environment near background values for naturally occurring substances and close to zero for man-made synthetic substances.

8.2 Blue whiting

8.2.1 Description of the fishery

The 2003 report of WGECO (ICES, 2003c) contains a more complete account of the ecosystem effects of the blue whiting fisheries.

The main blue whiting fishery occurs in deeper shelf-slope waters to the west of Scotland, Ireland, around the Faroes, and towards Iceland. In the North Sea, the fishery for the most part occurs in the Norwegian Trench to the south and west of Norway. In 2003, the TAC on blue whiting in western waters was 303,000 tonnes, of which the EU share was 46%. In the North Sea, the TAC was 67,250 t, of which the EU share was 41%. Prior to 2003, the uptake has been almost 100% in both areas. As of 2003, an autonomous TAC (non-specific to nationalities) of 250,000 tonnes has been set for international waters. The fishery in the Northeast Atlantic is considered to be largely unregulated. Estimates of blue whiting by-catch/discarding in other fisheries are poor, although the species is reported as a significant constituent of catches; e.g., Pierce *et al*. (2002), in a short study in the North Atlantic, reported that approximately 10% of the overall catch in the argentine (*Argentina silus*) fishery consisted of blue whiting.

8.2.2 Direct effects: by-catch

Heino *et al*. (2003) reported average annual landings from the mixed Norway pout-blue whiting fishery in the ICES Divisions IVa,b and IIa (North Sea and Norwegian Sea) in 2000–2002 of 109,000 tonnes. Catches were dominated by the target species: blue whiting and Norway pout. Blue whiting formed an estimated 58% of this catch, whilst Norway pout formed approximately 27%. The catches were dominated by fish of age 1 year, which were almost entirely juvenile blue whiting. 0-Group blue whiting started to appear in the catches during the third and fourth quarters of the year. The estimated numbers of 0-group blue whiting in the catches of the industrial fishery were 4.5×10^7, 4.0×10^6, and 1.2×10^8 individuals in 2000, 2001, and 2002, respectively. These numbers were not considered significant in comparison to the total recruitment to the stock (ICES, 2002a). However, Heino *et al*. (2003) note that some of the blue whiting in the area fished may represent local populations, which could then be affected more strongly by the fishery. There is a need to assess the population distribution and recruitment patterns of blue whiting in EU waters. The remaining 15% of the catch, or about 16,000 tonnes, consisted of a range of fish and invertebrates. The six most important by-catch species (in terms of landed catch) were saithe, herring, haddock, horse mackerel, whiting, and mackerel, each of which represented an annual catch of at least 1000 t in this fishery. Greater numbers of saithe were caught in the third and fourth quarters, and horse mackerel in the fourth quarter. Of the by-caught species, most individuals captured were in the length range of 25–40 cm, with herring and mackerel often slightly smaller, and saithe slightly larger. This length distribution suggests that the by-catch of herring and mackerel consisted primarily of juvenile individuals. Heino *et al*. (2003) noted that this may be a significant source of mortality on the non-target species and recommended that additional research be carried out, increasing sample size and over a longer period of time.

No published information was found on the composition of the by-catch in the larger, directed blue whiting Northeast Atlantic fishery.

8.2.3 Indirect effects

There are few or no assessments on the indirect effects of fishing for blue whiting on predators and prey. The species preys on zooplankton (Plekhanova and Soboleva, 1981; Plekhanova, 1990), decapod crustaceans, including *Meganyctiphanes norvegica* (Cabral and Murta, 2002; www.fishbase.org). Blue whiting also preys on fish (Dolgov, 2001; Dumke, 1983; Zilanov, 1984). In turn, blue whiting is an important prey for many fish species (Cabral and Murta, 2002; Bjelland *et al*., 2000; Hill and Borges, 2000; www.fishbase.org; Silva *et al*., 1997) and marine mammals (Desportes and Mouritsen ,1993; Bogstad *et al*., 2002).

The lack of information on blue whiting populations and ecology when the species is heavily exploited is of concern. The current estimates of SSB and fishing mortality are considered to be uncertain and the combined stock (Sub-areas I–IX, XII, and XIV) is likely to be harvested outside safe biological limits (ICES, 2003b). Furthermore, population genetic studies have indicated that partially separated stocks exist in the Mediterranean and in the eastern Barents Sea (Giæver and Mork, 1995; Giæver and Stien, 1998; Mork and Giæver, 1993). If there are some relatively local stocks, the overall catch depletions could conceal community extirpation of a valuable prey resource to higher predators. The lag between loss of a local blue whiting stock and recruitment from another area is unknown. If the species is an important predator of zooplankton and small mesopelagics, and a prey for larger fish, considering its high abundance in some regions, it is likely to play an important role in the pelagic ecosystem (Heino and Gordoe, 2002). To reiterate the advice of WGECO in 2003 (ICES, 2003c), further investigation of these ecosystem function aspects is of paramount importance.

8.3 An evaluation of the relative benefits, in terms of ecological efficiency, of fishing "industrial" fish for fish feed for the aquaculture industry, or of not fishing these species and obtaining higher yields from "human consumption" fisheries

8.3.1 Introduction

This section addresses the question: what is the most efficient way, from an ecological perspective, of utilising lower trophic level fish resources? Is it more effective to harvest low trophic level species in industrial fisheries and convert the biomass obtained to human consumption fish protein in aquaculture systems, or is it better to leave low trophic level fish in the sea where they can be consumed by their natural predators, and then to harvest species from higher trophic levels in fisheries for human consumption?

The production of fish protein for human consumption in aquaculture systems relies heavily on feed pellets, which are generally derived from lower trophic level fish harvested from marine ecosystems by what are termed "industrial" fisheries (Naylor *et al*., 2000). In the North Sea, the exploitation of sandeels by the industrial fishery gives rise to the largest single-species annual catch, exceeding 1 million tonnes in some years, and accounting for around 40% of the total quantity of fish landed each year. Sandeels contribute approximately 80% of the total industrial fishery each year. Consequently, the remainder of this section focuses on the exploitation of sandeels by both the industrial fishery and their natural predators, and their use in the aquaculture industry.

8.3.2 Transfer efficiency in natural marine food webs

The transfer efficiency of both energy and carbon between trophic levels along a food chain is not 100% efficient. Energy is required for metabolism and maintenance and only a fraction of the food consumed by a predator is actually converted to predator biomass. Estimates of transfer efficiency vary from 10% (Lindeman, 1942; Slobodkin, 1961) to 20% or more (Greenstreet *et al*., 1997). Transfer efficiencies in the range 10% to 15% are generally accepted for predator-prey interactions involving fish predators in marine temperate shelf-sea food webs (Christensen and Pauly, 1993; Pauly *et al*., 2000; Jennings *et al*., 2002).

8.3.3 Transfer efficiency in aquaculture systems

The conversion of sandeel biomass to nutritionally complete feed pellets is generally limited by the amount of fish oil that can be derived from sandeel material. Feed pellets vary in their fish meal to fish oil ratios depending on the species for which they are intended. Feed pellets intended for carnivorous species, such as salmon, have a composition of 45% fish meal and 25% fish oil (Naylor *et al.*, 2000). One hundred tonnes of sandeel material produces 5 t of fish oil and 20 t of fish meal. From this, 20 t of nutritionally complete salmon pellet feed can be derived, utilising all the oil, but only 9 t of the fish meal leaving 11 t of meal for other purposes. Feed conversion ratios (the ratio of pellet feed consumed to fish biomass produced) vary between species, but for most farmed fish they tend to lie between 1.8 and 2.2 (Naylor *et al*., 2000). In the case of salmon feed, conversion ratios as high as 1.2:1 have been described. Under such circumstances, 20 t of feed pellets would produce 16.7 t of salmon biomass, a total conversion efficiency of sandeel biomass to salmon biomass of 16.7%, which is towards the top end of the range of fish predator trophic efficiency in the wild. However, food conversion ratios closer to 2:1 are more common, for example, in the Scottish salmon farming industry. At this food conversion ratio, a total of 10 t of salmon are produced for every 100 t of sandeels processed. This gives a transfer efficiency of 10%, which is towards the lower end of the range of transfer efficiencies observed in the wild.

In the case of aquaculture salmon production, conversion of sandeel biomass to fish flesh for human consumption appears as efficient as would be the case if the sandeel were left in the marine environment to be converted, through natural trophic interactions, to fish protein harvested in the human consumption fisheries. However, the production of pellet feed for salmon production is more demanding nutritionally than the production of pellet feed suitable for other farmed species. Pellet feed for other species requires less fish oil. Nutritionally complete pellet feed for marine finfish species, such as sole, halibut, cod, haddock, etc., for example, requires a nutritionally complete food pellet mix that consists of 50% fish meal and 15% fish oil (Naylor *et al.,* 2000). One hundred tonnes of sandeel will therefore still produce the same 5 t of fish oil, but 16.5 t of the fish meal produced can now be utilised to produce 33 t of pellet feed, leaving only 3.5 t of excess meal. When fed to these fish species at a food conversion ratio of 2.2:1, some 15 t of farmed marine finfish can be produced. This represents a trophic transfer efficiency of 15% from sandeel biomass to marine human consumption finfish flesh. Considering that these are the types of fish that would utilise sandeel biomass in the wild, this transfer efficiency is towards the top end of the range that might generally be expected in natural food webs.

8.3.4 Evaluation of the case for exploiting sandeels or leaving them in the sea

In making the comparisons of trophic transfer efficiency in natural marine food webs and aquaculture systems, only the conversion of sandeel biomass to human consumption fish protein has so far been considered. But this is by no means the only energy/material "cost" involved in the process. To produce 20 t of salmon food pellets requires 5 t of fish oil and 9 t of fish meal from 100 t of processed sandeel material. A further 6 t of other nutrient material is required. Similarly, 33 t of marine finfish feed pellets is derived from 21.5 t of sandeel material, leaving a further 11.5 t of additional material to be obtained from elsewhere. This additional material also requires processing and any energy transfer efficiencies involved have not been factored into the analysis. Furthermore, the entire process of converting sandeel biomass to feed pellets will involve a processing energy cost, and this has also not been accounted for. The unused fish meal may go some way towards redressing the balance against these additional unaccounted inputs. If the additional energy/material costs involved in the production of pellet feed for the aquaculture industry are to taken into account, then the material/energy costs necessary for the maintenance and operation of the various human consumption and industrial fishery fleets will also need to be considered.

A further assumption has also been made, that for every 100 t of sandeel left in the sea 10 t of fish flesh suitable for human consumption will be produced. This is very unlikely. Numerous other fish predators, such as grey gurnard (Hislop, 1997) and long-rough dab (H. Fraser, FRS Aberdeen, pers. comm.), also prey on sandeels and these are generally not exploited in the human consumption fisheries. In addition, many non-fish marine predators, such as seabirds and marine mammals, also rely heavily on sandeels in their diets (Pierce *et al.*, 1990, 1991; Hammond *et al.*, 1994; Tollit *et al.*, 1997; Wanless *et al.*, 1998). Thus, while trophic transfer efficiency in marine food chains may be around 10%, for every 100 t of sandeel left in the North Sea, the amount of fish flesh produced that is potentially exploitable by the human consumption fisheries is likely to be substantially less than 10 t.

In conclusion, if one is only concerned about the efficiency of converting sandeel biomass to human consumption fish biomass, then the exploitation of sandeels by industrial fisheries for the aquaculture industry is at least as efficient ecologically. At least as much fish flesh for human consumption can be produced in aquaculture systems for every 100 t of sandeel biomass processed as is likely to be harvestable from the natural marine food web of the North Sea. The question then largely becomes one of social economics. Is it of greater benefit to society to exploit lower trophic level marine fish resources in industrial fisheries and rely on an aquaculture industry to provide mankind's human consumption fish requirements, or is it better to leave these fish to be processed through the natural marine food web and then to harvest fish in the higher trophic levels in fisheries for human consumption? WGECO does not have the expertise to analyse the social and economic aspects underpinning such a question, however, examination of the underlying ecology involved may introduce some direction to this debate. WGECO also notes that there are a variety of other ecological issues relating to aquaculture, such as the impact of "escapees", which are not addressed here.

8.3.5 The relationship between industrial fisheries and human consumption landings

Implicit in the arguments underlying the comparisons made in the sections above is the belief that landings in human consumption fisheries are directly and inversely related to catches made by the industrial fishery. It is assumed that if industrial fisheries catches are reduced, gains approaching 10% of the reduction will be made in the human consumption fishery landings. Recent runs of the Multi-Species Virtual Population Analysis (MSVPA) model provide information that can be used to examine this assumption (ICES, 2003d). In addition, data collected off the east coast of Scotland are now available and can be used to assess the direct consequences of a four-year sandeel fishing closure on local gadoid (cod, haddock, and whiting) populations.

8.3.5.1 MSVPA results

Long-term trends in the utilisation of the 0-group and 1+-group sandeel resource by fish predators, seabird predators, and the industrial fishery are shown in Figure 8.3.5.1.1. Virtually no 0-group sandeels are taken by the industrial fishery. The major consumers of 0-group sandeels are fish predators and consumption has steadily increased, from around 200,000 t to approximately 500,000 t, over the period 1963 to 2001. At the start of this period, fish predators were also the principal source of mortality of 1+ aged sandeels, and relatively little sandeel biomass was removed by the industrial fishery. Over this time period, however, the annual removal of 1+ sandeels by the industrial fishery has increased to approximately one million tonnes, while consumption of 1+-group sandeels by fish predators has declined from around 1.2 million tonnes to 0.8 million tonnes. Initially therefore, these two trends would seem to suggest that the industrial fishery does compete with fish predators for the 1+ aged sandeel resource. Even if this were the case, and the sandeel fishery were to be closed immediately, these data do not suggest that we could reasonably expect a gain of 100,000 t (10% of 1 million tonnes) in human consumption landings. Consumption of 1+ sandeels by fish predators has only decreased by 400,000 t, thus the best that might reasonably be expected is a gain in human consumption landings of 40,000 t. Following the calculations in the sections above, the 1 million tonnes of sandeels taken annually by the industrial fishery might be expected to produce 100,000 t of farmed salmon or 150,000 t of farmed marine finfish. Even if one accepts that the industrial fishery has limited 1+ aged sandeel consumption by fish predators, aquaculture supported by an industrial fishery would appear to be the more ecologically efficient option.

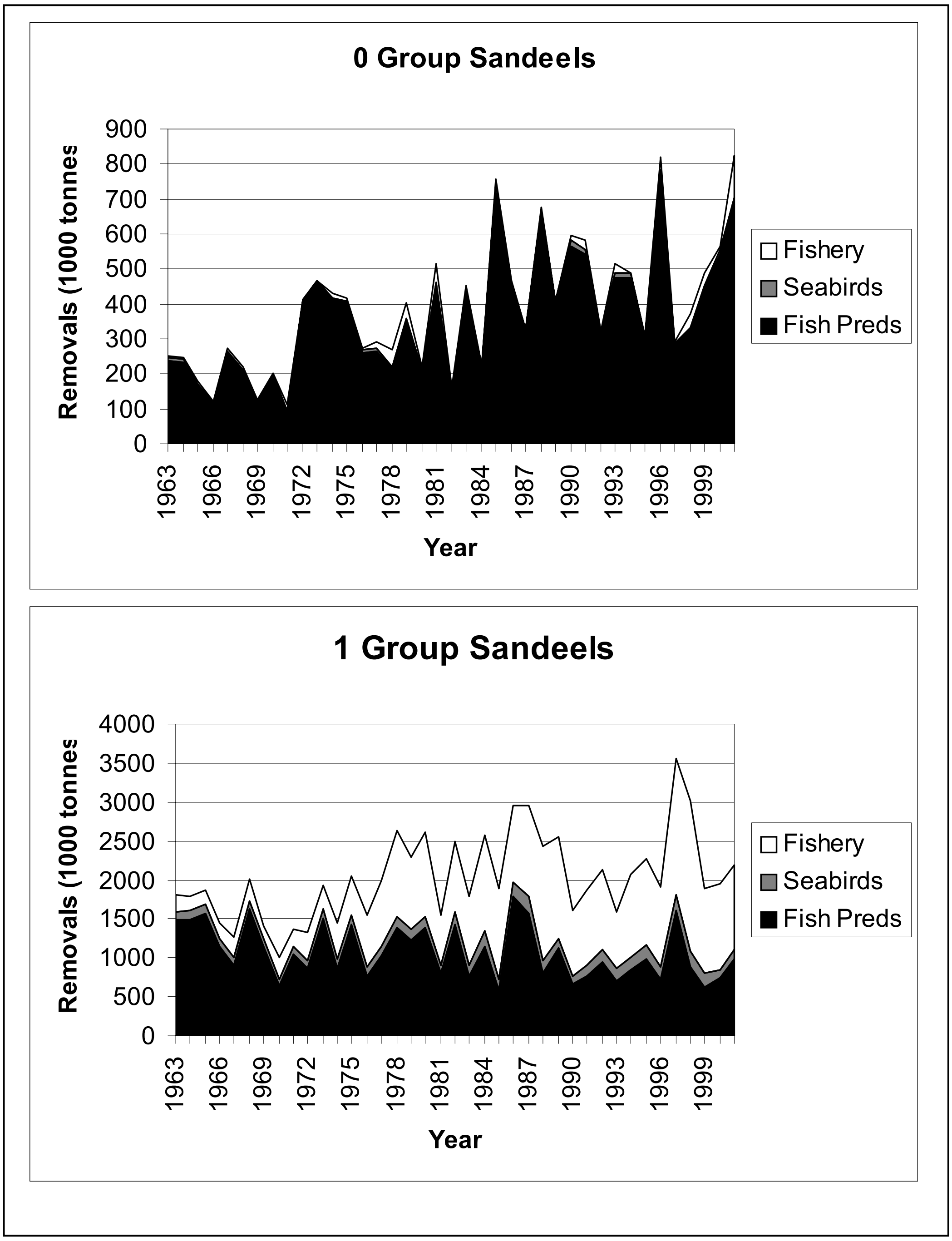


Figure 8.3.5.1.1. Long-term trends in the utilisation of the 0-group and 1+-group sandeel resource by industrial fisheries and fish and seabird predators based on data derived from the MSVPA (ICES, 2003d).

The assumption of competitive limitation of sandeel consumption by fish predators by the industrial fishery is now examined. Based on the MSVPA data, no obvious long-term trends in sandeel stock size in the North Sea are apparent (Figure 8.3.5.1.2). The increased consumption of 0-group sandeels by predatory fish does not appear to have hampered recruitment to the sandeel stock: 1-group sandeel biomass, albeit variable, shows no decline. There is certainly no indication of a decline in the overall sandeel stock size. Thus, the reduction in the consumption of 1+ sandeel by fish predators does not appear to be the result of a reduction in sandeel prey abundance caused by industrial fishing. Trends in stock biomass of the fish predators modelled by the MSVPA indicate a substantial decline in predator biomass (Figure 8.3.5.1.3). This is particularly apparent for the three main gadoid predators, cod, whiting, and haddock, as well as North Sea mackerel, which are the major piscivorous species exploited in the human consumption fisheries in the North Sea. The high levels of fishing mortality experienced by these main fish predator stocks provide the explanation for the stock declines (Figure 8.3.5.1.4). Fishing mortality on cod has continued to increase over the whole period and the stock has steadily declined. Fishing mortality on the other gadoids peaked in the mid- to late 1980s, since when declines in the stock biomass have slowed down. These data suggest that even the sandeel fishery were to stop, current levels of fishing mortality are sufficiently high as to prevent any increase in the biomass of these major fish predators. Any gains in human consumption fishery landings as a result of a sandeel closure are likely therefore to be minimal. This premise is examined in the next section where the effect of closing the sandeel fishery off the east coast of Scotland on local gadoid predators is examined.

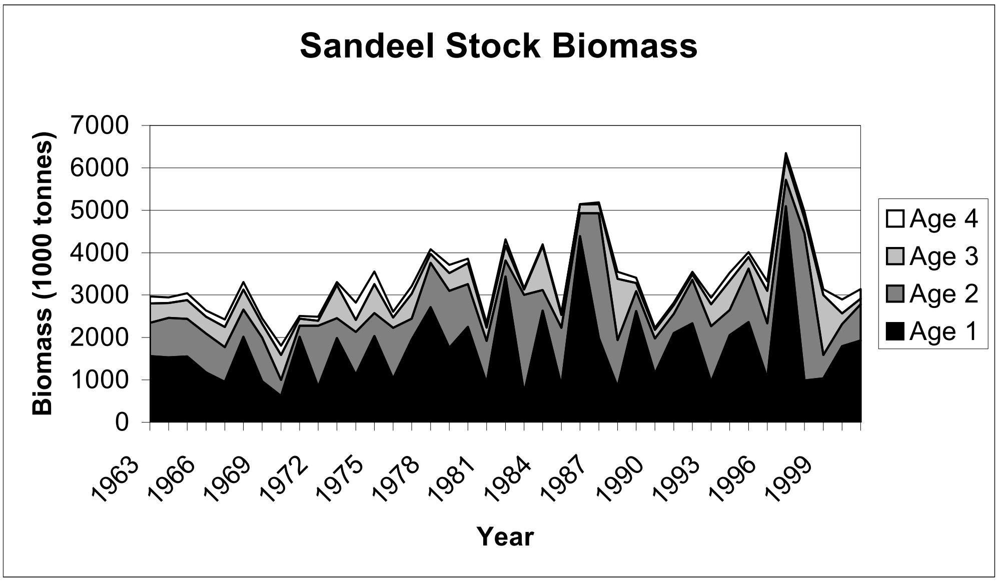


Figure 8.3.5.1.2. Long-term trend in the sandeel stock biomass of the North Sea based on data derived from the MSVPA (ICES, 2003d).

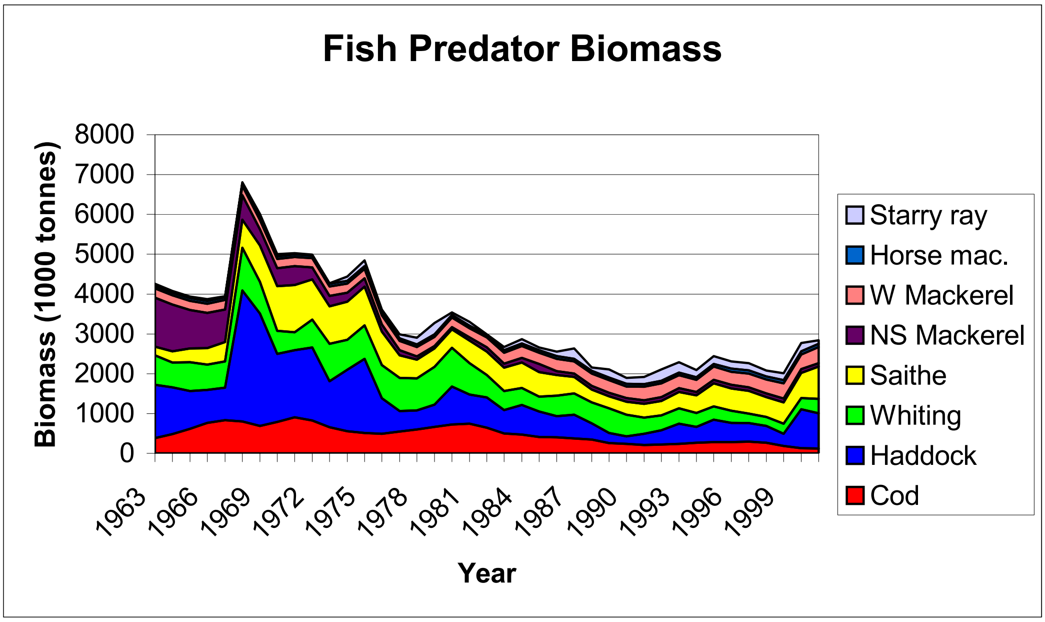


Figure 8.3.5.1.3. Long-term trends in the stock biomass of the fish predators modelled by the MSVPA (ICES, 2003d).

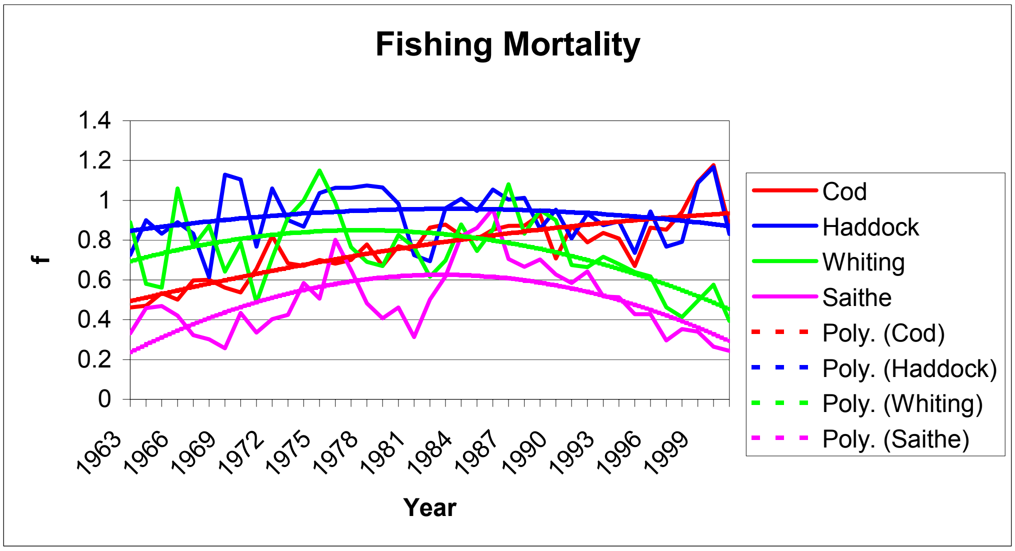


Figure 8.3.5.1.4. Long-term trends in fishing mortality for the species for which data were provided by the MSVPA model (ICES, 2003d).

8.3.5.2 Effect of closing the sandeel fishery off the east coast of Scotland on gadoid predators

In 2000, following concern over declining kittiwake breeding productivity, fishing for sandeels in an area off the east coast of Scotland was stopped (ICES, 1999). Data on gadoid biomass, diets, food consumption rates, and body condition in an area that included the main sand banks were collected over the period 1997 to 2003. Thus, three years of data were available while the sandeel fishery was in operation, and four years of data were collected subsequent to the sandeel fishery closure. Up to 100,000 t of sandeels were removed from this area by the fishery prior to the closure (Figure 8.3.5.2.1). Following the closure, sandeel abundance in the area increased markedly (Figure 8.3.5.2.2). Sandeels featured strongly in the diets of three gadoid predators: cod, haddock, and whiting (Figure 8.3.5.2.3). Over the seven-year period, the biomass of all three predators in the area declined (Figure 8.3.5.2.4); thus closure of the sandeel fishery had no beneficial effect on gadoid predator biomass. The percentage of sandeels in the diet of each of the predators was unaffected by closure of the sandeel fishery and the resultant increase in sandeel abundance (Figure 8.3.5.2.5), and no increase in food consumption rates was observed (Figure 8.3.5.2.6). Consequently predator body condition was not enhanced in the years that the closure was in force (Figure 8.3.5.2.7). Examination of the size of sandeels taken by the three gadoid predators revealed a strong dependence on 0-group sandeels (Figure 8.3.5.2.8), and this was not affected by closing the sandeel fishery (Figure 8.3.5.2.9), which primarily targeted 1+ aged sandeels. In summary, ceasing the industrial fishery for sandeels off the east coast of Scotland had no beneficial effect on the biomass, diet, feeding rate or body condition of gadoid predators in the area.

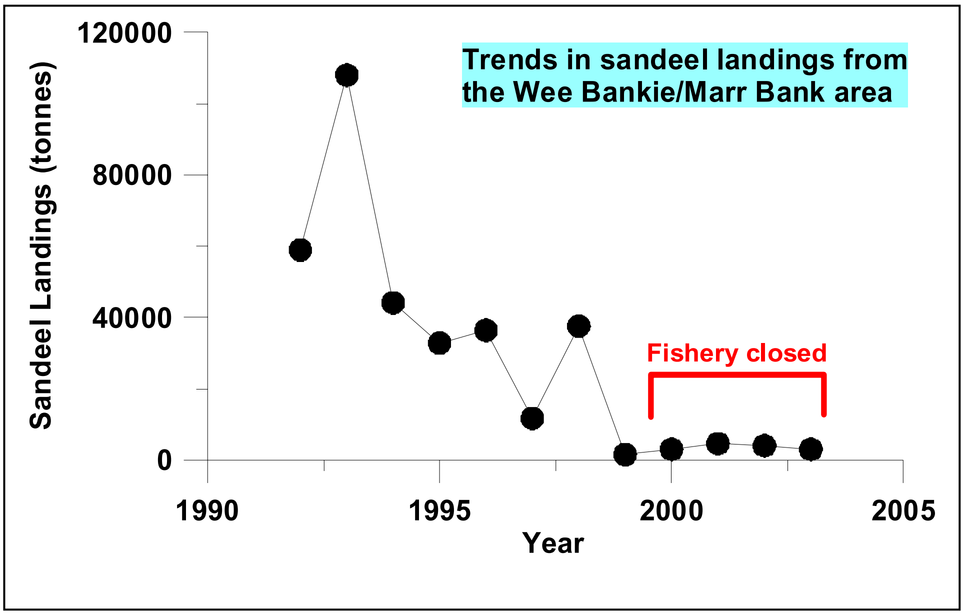


Figure 8.3.5.2.1. Trends in sandeel landings from the Wee Bankie/Marr Bank areas off the Firth of Forth, southeast Scotland.

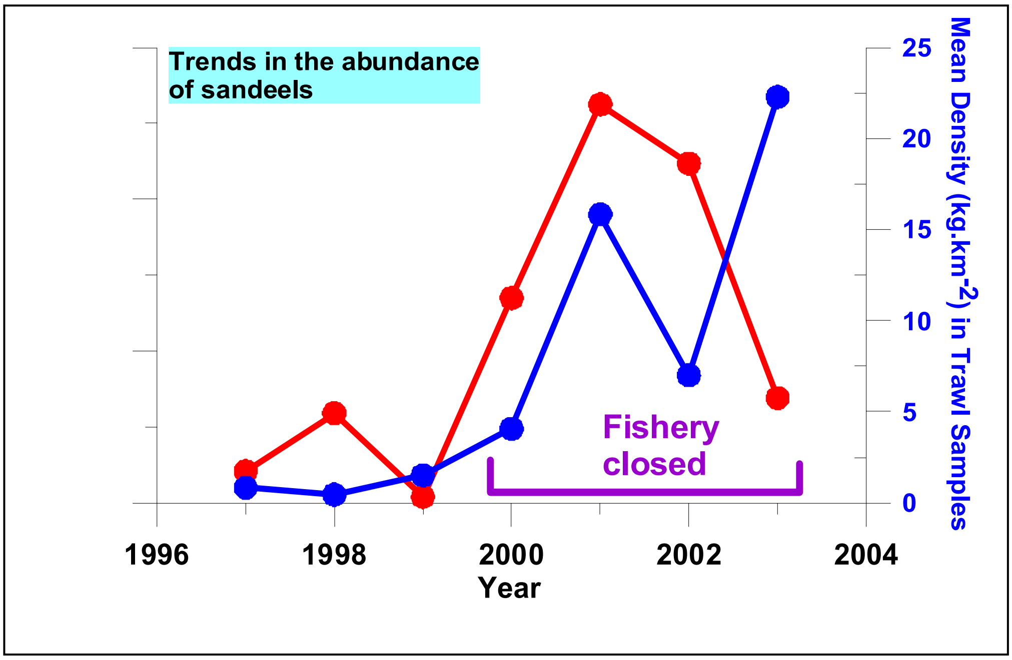


Figure 8.3.5.2.2. Trends in sandeel abundance in the Wee Bankie/Marr Bank area, off the Firth of Forth, southeast Scotland, as indicated by acoustic and demersal trawl survey indices.

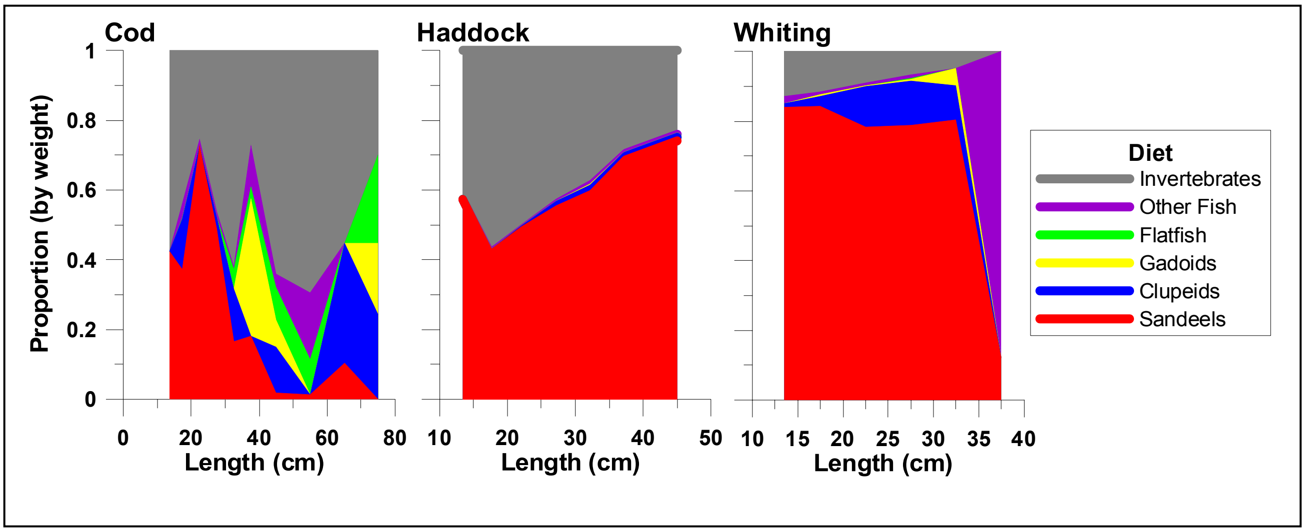


Figure 8.3.5.2.3. The effect of predator body length on the diets of cod, haddock, and whiting predators in the Wee Bankie/Marr Bank area, off the Firth of Forth, southeast Scotland.

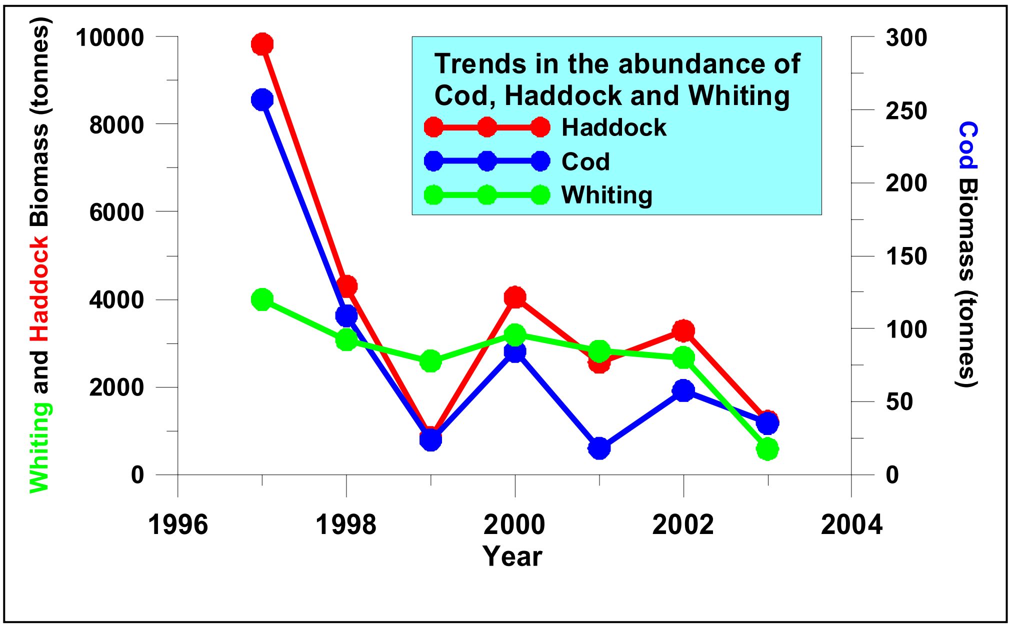


Figure 8.3.5.2.4. Trends in the biomass of cod, haddock, and whiting in the Wee Bankie/Marr Bank area, off the Firth of Forth, southeast Scotland.

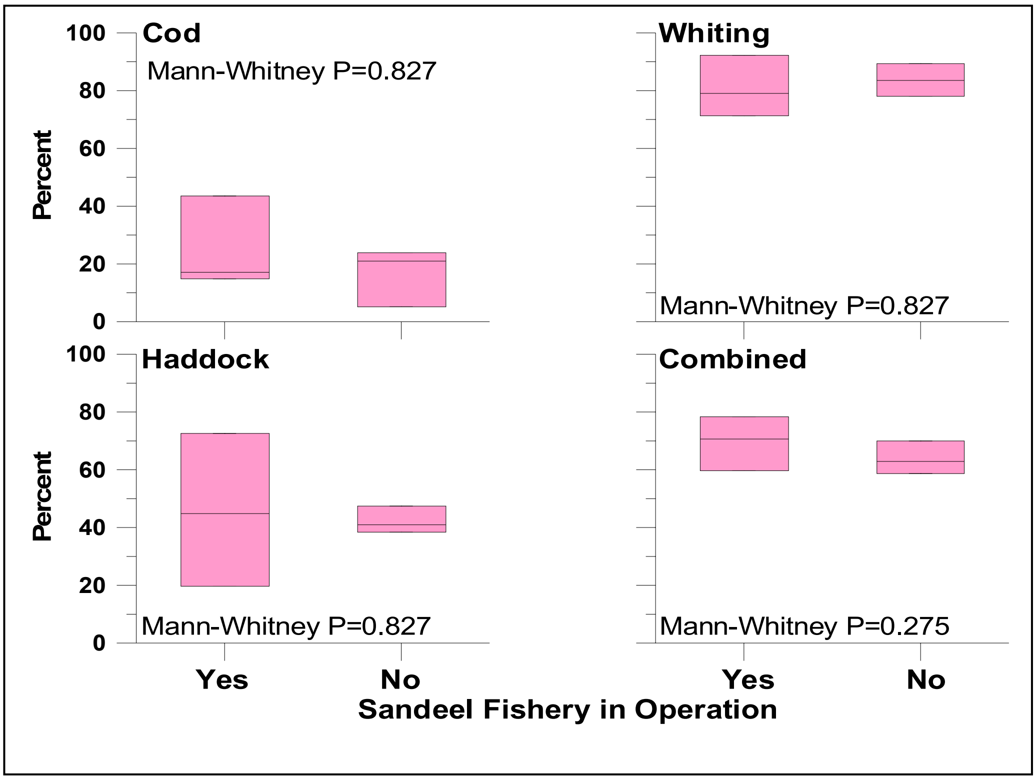


Figure 8.3.5.2.5. Variation in the percentage of sandeels in the diets of cod, haddock, and whiting in the Wee Bankie/Marr Bank area, off the Firth of Forth, southeast Scotland, in years when the sandeel fishery was in operation and years when the closure was in force.

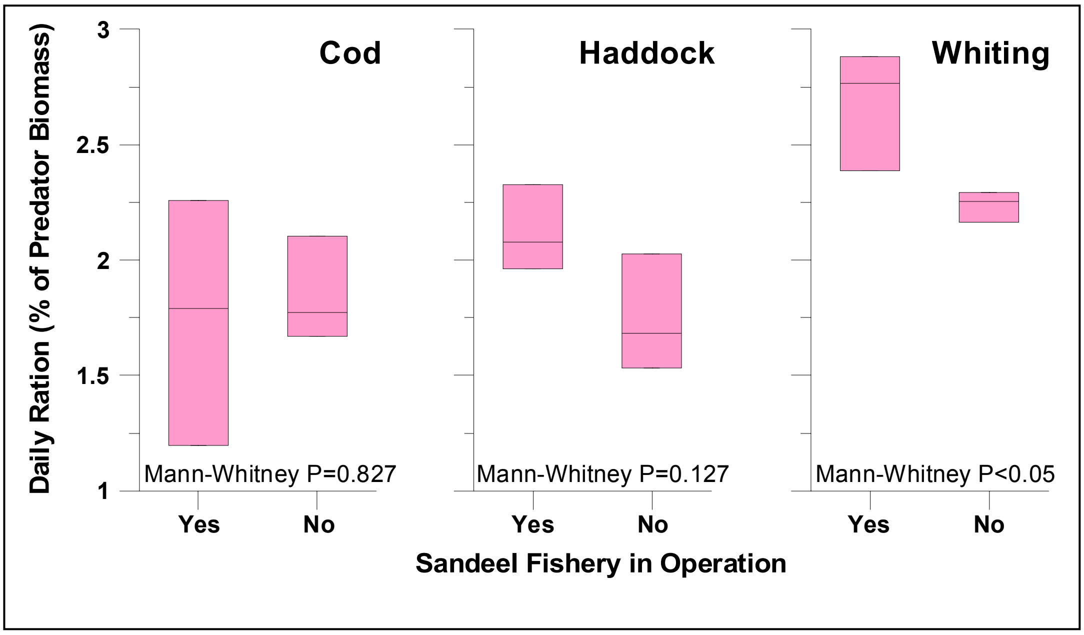


Figure 8.3.5.2.6. Variation in the daily food consumption rates of cod, haddock, and whiting in the Wee Bankie/Marr Bank area, off the Firth of Forth, southeast Scotland, in years when the sandeel fishery was in operation and years when the closure was in force.

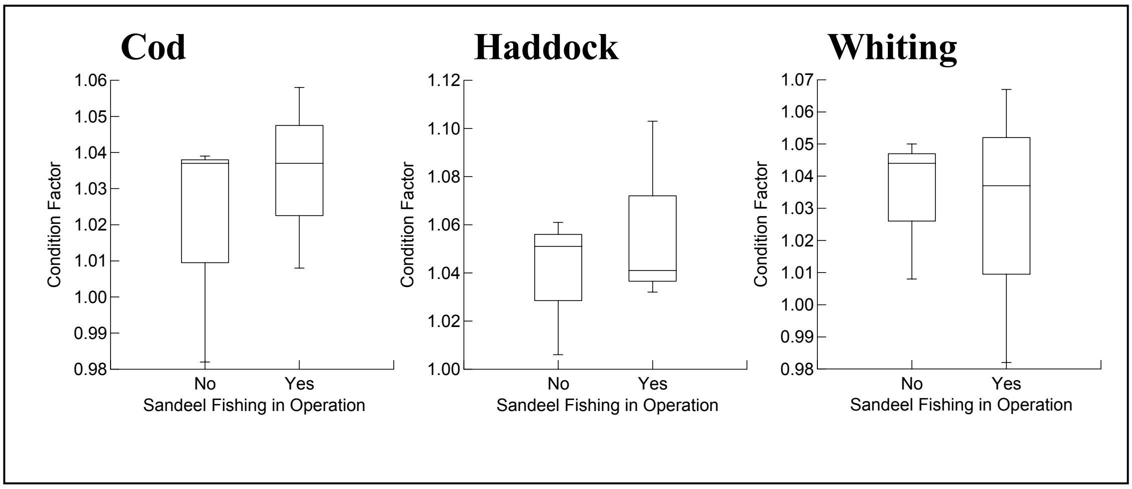


Figure 8.3.5.2.7. Variation in the body condition of cod, haddock, and whiting in the Wee Bankie/Marr Bank area, off the Firth of Forth, southeast Scotland, in years when the sandeel fishery was in operation and years when the closure was in force. None of the comparisons were statistically significant.

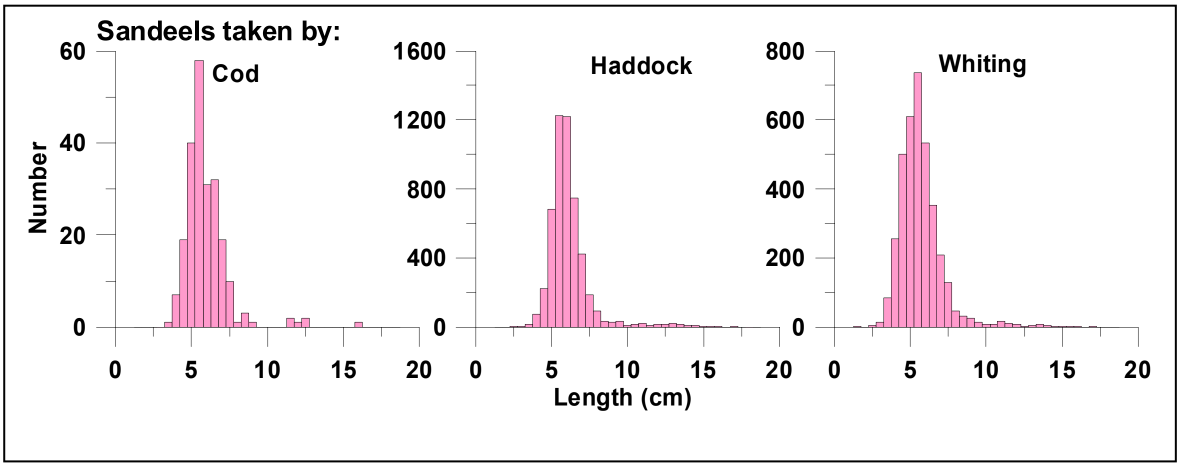


Figure 8.3.5.2.8. Length frequency distributions of sandeel prey taken by cod, haddock, and whiting in the Wee Bankie/Marr Bank area, off the Firth of Forth, southeast Scotland.

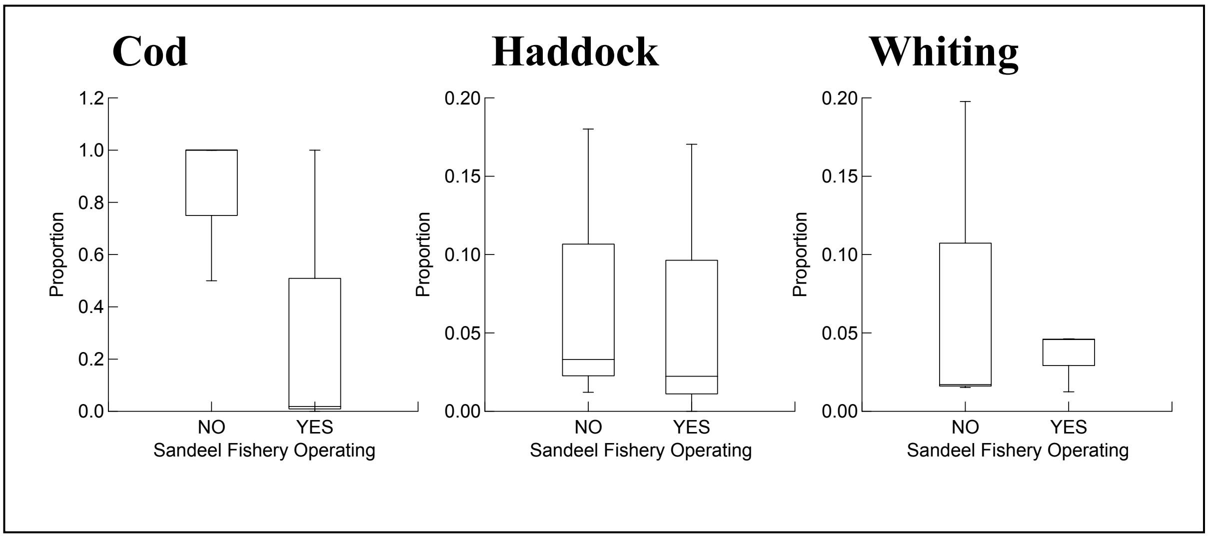


Figure 8.3.5.2.9. Variation in the proportion of sandeels taken by cod, haddock, and whiting in the Wee Bankie/Marr Bank area, off the Firth of Forth, southeast Scotland, that exceeded 8.5 cm in length in years when the sandeel fishery was in operation and years when the closure was in force. None of the comparisons were statistically significant.

8.3.5.3 Conclusions

The results of these two case studies provide no evidence to support the contention that ceasing industrial fisheries will stimulate catches in the human consumption fisheries at the current time, and under the prevailing circumstances. Instead, the data suggest that, at current population sizes, fish in the higher trophic levels in the North Sea food web are not food limited, and thus there is no reason to expect any gain in human consumption landings following a reduction in industrial fishery catches. Should fisheries management result in a recovery of the currently depleted predator stocks, such as gadoids, in the North Sea this conclusion would need revisiting. The impact of the industrial fishery on sandeel abundance should be carefully monitored as industrial fisheries continue to operate. This monitoring should be carried out at spatial and temporal scales that are relevant to the predators in question, which may or may not be recovering. Managers must ensure that food supplies for gadoids and other marine predators do not become limiting as a result of anthropogenic activities. This will ensure protection of ecologically dependent species, such as kittiwakes, and will therefore provide adequate food for recovering gadoid populations. Analysis of the MSVPA data available to date, however, suggests that, even under circumstances of increased biomass of human consumption species, a carefully managed industrial fishery should not impinge on those fisheries. Indeed, the analysis of food conversion efficiency suggests that a closely regulated combination of industrial and human consumption fisheries may provide the only solution to the long-term demand for fish protein.

8.4 Research priorities and recommendations

8.4.1 Herring and sprat

Research priorities

Sprat and herring are large, important fisheries for the Baltic States. The paucity of information on the Baltic sprat and herring fisheries, particularly in relation to geo-location of catch, and the relatively low catch sampling effort prevents complete evaluation of the ecosystem effects of these fisheries. Given that these fisheries take considerable biomass from the Baltic ecosystem, this is of major concern and needs to be addressed.

WGECO agrees with WGBFA which recommends that sampling and assessment of the species compositions of the pelagic fish caught in the Baltic should be re-evaluated/revised at national level (ICES, 2003a). The re-evaluation should take account both of samples collected by fishery inspectors as well as of biological samples collected onboard fishing vessels. When the BALTCOM database is updated with the material relevant to the quantification of by-catch in the Baltic, such as species composition in catches and landing data, WGBFA/WGECO should revisit this topic in 2005. The apparent lack of knowledge of diets of predators should be addressed.

Management recommendations

WGECO endorses the STECF (2004) recommendation that, as herring are taken with sprat in mixed pelagic fisheries, given that the status of the Baltic herring in Subdivisions 25–29 and 32 (excluding the Gulf of Riga) is uncertain, management should ensure that herring catches in the mixed pelagic fisheries do not contribute to overexploitation.

8.4.2 Blue whiting

Research priorities

Considering their abundance, and the fact that blue whiting is simultaneously both an important predator and an important prey species, it is likely that blue whiting fulfils an important structural role in the shelf-slope food web. Research into the role of blue whiting in the ecosystem is therefore necessary.

No information on the by-catch in the North Atlantic fishery was found. While this may mean that the by-catch is insignificant, without data it is not possible to confirm this. To understand the ecosystem effects of fishing, information on the full catch is needed.

8.5 References

Baars, A.J., Bakker, M.I., Baumann, R.A., Boon, P.E., Freijer, J.I., Hoogenboom, L.A.P., Hoogerbrugge, R., van Klaveren, J.D., Liem, A.K.D., Traag, W.A., and de Vries, J. 2004. Dioxins, dioxin-like PCBs and non-dioxin-like PCBs in foodstuffs: occurrence and dietary intake in The Netherlands. Toxicology Letters. In press.

Bjelland, O., and Monstad, T. 1997. Blue whiting in the Norwegian Sea, spring and summer 1995 and 1996. ICES CM 1997/CC:15.

Bjelland et al, 2000.

Bogstad, B., Haug, T., and Mehl, S. 2000. Who eats whom in the Barents Sea? NAMMCO Scientific Publications, 2: 98–119.

Cabral, H.N., and Murta, .A.G. 2002. The diet of blue whiting, hake, horse mackerel and mackerel off Portugal. Journal of Applied Ichthyology, 18(1): 14–23.

Christensen, V., and Pauly, D. 1993. Flow characteristics of aquatic ecosystems. *In* Trophic Models of Aquatic Ecosystems, pp. 338–352. Ed. by V. Christensen and D. Pauly. ICLARM Conference Proceedings.

Dagys, M., and Zydelis, R. 2002. Bird bycatch in fishing nets in Lithuanian coastal waters in wintering season 2001–2002. Acta Zoologica Lituanica, 12: 276–282.

Desportes, G., and Mouritsen, R. 1993. Preliminary results on the diet of long-finned pilot whales off the Faroe Islands. Reports of the International Whaling Commission, Special Issue, 14: 305–324.

Dolgov, A.V. 2001. Some data on the diet of blue whiting in the Barents Sea in 1998 and 2000. Working paper for the 2001 meeting of the ICES Working Group on Northern Pelagic Fisheries and Blue Whiting. 5 pp.

Du Buit, M.H. 1996. Diet of hake (*Merluccius merluccius*) in the Celtic Sea. Fisheries Research, 28: 381–394.

Dumke, A. 1983. Feeding investigations of blue whiting (*Micromesistius poutassou*) in the Norwegian Sea. ICES CM 1983/H:58. 9 pp.

EC. 2001. EC Directive 102/2001, Undesirable substances and products in animal nutrition, 27 November 2001.

EC. 2002. EC Directive 2002/32, Undesirable substances and products in animal nutrition, 21 November 2003.

EP. 2004. The fish meal and fish oil industry: its role in the Common Fisheries Policy. European Parliament, Luxembourg. FISH 113 EN. pp.148

Giæver, M., and Mork, J. 1995. Further studies on the genetic population structure of the blue whiting (*Micromesistius poutassou*) in the northeast parts of the distribution range. ICES CM 1995/H:11. 6 pp.

Giæver, M., and Stien, J. 1998. Population genetic substructure in blue whiting based on allozyme data. Journal of Fish Biology, 52(4): 782–795.

Greenstreet, S.P.R., Bryant, A.D., Broekhuizen, N., Hall, S.J., and Heath, M.R. 1997. Seasonal variation in the consumption of food by fish in the North Sea and implications for foodweb dynamics. ICES Journal of Marine Science, 54: 243–266.

Hammond, P.S., Hall, A. J., and Prime, J.H. 1994. The diet of grey seals around Orkney and other island and mainland sites in northeastern Scotland. Journal of Applied Ecology, 31: 340–350.

Hario, M. 1998. Review of incidental catches of seabirds in fisheries in Finland. CAFF Technical Report, 1: 20–24.

Harvey, C.J., Cox, S.P., Essington, T.E., Hansson, S., and Kitchell, J.F. 2003. An ecosystem model of food web and fisheries interactions in the Baltic Sea. ICES Journal of Marine Science, 60: 939–950.

Heino, M., and Godø, O.R. 2002. Blue whiting - a key species in the mid-water ecosystems of the north-eastern Atlantic. ICES CM 2002/L:28.

Heino, M., Johannessen, T., and Tangen, Ø. 2003. Norwegian mixed industrial fishery for blue whiting and Norway pout in the North Sea and Southern Norwegian Sea. Working paper to the 2003 meeting of the Northern Pelagic and Blue Whiting Fisheries Working Group.

Hill, L., and Borges, M.F. 2000. A comparison of the seasonal abundance of hake (*Merluccius merluccius*) and its main prey species off the Portuguese coast. ICES CM 2000/Q:13.

Hislop, J.R.G. 1997. Database report of the stomach sampling project 1991. ICES Cooperative Research Report, 219: 1–422.

Hopkins, C.C.E. 2000. Overview of monitoring in the Baltic Sea. Report to the Global Environment Facility/Baltic Sea Regional Project. AquaMarine Advisers. 40 pp.

ICES. 1999. Report of the Study Group on Effects of Sandeel Fishing. ICES CM 1999/ACFM:19.

ICES. 2000a. Study Group on Estimation of the Annual Amount of Discards and Fish Offal in the Baltic. ICES CM 2000/ACME:06.

ICES. 2001a. Report of the Baltic Fisheries Assessment Working Group. ICES CM 2001/ACFM:18.

ICES. 2002a. Report of the Northern Pelagic and Blue Whiting Fisheries Working Group. ICES CM 2002/ACFM:19.

ICES. 2002b. Report of the Baltic Fisheries Assessment Working Group. ICES CM 2002/ACFM:17.

ICES. 2002c. Report of the Working Group on the Assessment of Demersal Stocks in the North Sea and Skagerrak. ICES CM 2002/ACFM:01. 555 pp.

ICES (2003a). Report for the Baltic Fisheries Assessment Working Group. ICES CM 2003/ACFM:21

ICES. 2003b. Report of the ICES Advisory Committee on Fisheries Management, 2003. http://www.ices.dk/committe/acfm/comwork/report/asp/acfmrep.asp

ICES. 2003c. Report of the Working Group Ecosystem Effects of Fishing. ICES CM 2003/ACE:05.

ICES. 2003d. Report of the Study Group on Multispecies Assessments in the North Sea. ICES CM 2003/D:09. 95 pp.

ICES. 2003e. Report on the Study Group on Multispecies Assessment in the Baltic. ICES CM 2003/H:03.

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

ICES. 2004. Report of the Working Group on Seabird Ecology. ICES CM 2004/ C:05

Jennings, S., Greenstreet, S.P.R., Hill, L., Piet, G.J., Pinnegar, J., and Warr, K.J. 2002. Long-term trends in the trophic structure of the North Sea fish community: evidence from stable isotope analysis, size-spectra and community metrics. Marine Biology, 141: 1085–1097.

Karl, K., Ruoff, U., and Blüthgen, A. 2002. Levels of dioxins in fish and fishery products on the German market. Chemosphere, 49(7): 765–773.

Kaschner, K. 2003. Review of small cetacean bycatch in the ASCOBANS area and adjacent waters – current status and suggested future actions. ASCOBANS, Bonn.

Kirchhoff, K. 1982. Wasservogelverluste durch die Fischerei an der schleswig-holsteinischen Ostseeküste. Vogelwelt, 103: 81–89.

Kiviranta, H., Vartiainen, T., Parmanne, R., Hallikainen, A., and Koistinen, J. 2003. PCDD/Fs and PCBs in Baltic herring during the 1990s. Chemosphere, 50: 1201–1216.

Lindeman, R.L. 1942. The trophic-dynamic aspect of ecology. Ecology, 23(4):399-418.

Lyngs, P., and Durinck, J. 1998. Diet of guillemots *Uria aalge* in the central Baltic Sea. Dansk Ornitologisk Forenings Tidsskrift, 92: 197–200.

Mork, J., and Giaever, M. 1993. The genetic population structure of the blue whiting (*Micromesistius poutassou*). ICES CM 1993/H:5.

Naylor, R.L., Goldburg, R.J., Primavera, J.H., Kautsky, N., Beveridge, M.C.M., Clay, J., Folke, C., Lubchenco, J., Mooney, H., and Troell, M. 2000. Effect of aquaculture on world fish supplies. Nature, 405: 1017–1024.

Pauly, D., Christensen, V., and Walters, C. 2000. Ecopath, Ecosim and Ecospace as tools for evaluating ecosystem impacts of fishing. ICES Journal of Marine Science, 57: 697–706.

Pedersen, M., and Snoeijs, P. 2001. Patterns of macroalgal diversity, community composition and long-term changes along the Swedish west coast. Hydrobiologia, 459: 83–102.

Pierce, G.J., Boyle, P.R., Diack, J.S.W., and Clark, I. 1990. Sandeels in the diets of seals: applications of novel and conventional methods of analysis to faeces from seals in the Moray Firth area of Scotland. Journal of the Marine Biological Association of the United Kingdom, 70: 829–840.

Pierce, G.J., Dyson, J., Kelly, E., Eggleton, J.D., Whomersley, P., Young, I.A.G., Begoña Santos, M., Wang, J., and Spencer, N.J. 2002. Results of a short study on by-catches and discards in pelagic fisheries in Scotland (UK). Aquatic Living Resources, 15: 327–334.

Pierce, G.J., Thompsom, P.M., Miller, A., Diack, J.S.W., Miller, D., and Boyle, P.R. 1991. Seasonal variation in the diet of common seals (*Phoca vitulina*) in the Moray Firth area of Scotland. Journal of Zoology, 223: 641–646.

Plekhanova, N.V. 1990. Feeding conditions of blue whiting in the Norwegian Sea in spring and summer of 1980–1987. *In* Biology and fishery for Norwegian spring-spawning herring and blue whiting in the North-East Atlantic: Proceedings of the IV Soviet-Norwegian Symposium. Murmansk, PINRO, pp.400–419. (in Russian).

Plekhanova, N.V., and Soboleva M.S. 1981. Features of blue whiting feeding in the Norwegian Sea offshore waters in 1979 and 1980. ICES CM 1981/H:27. 9 pp.

Rahikainen, M., Kuikka, S., and Parmanne, R. 2003. Modelling the effect of ecosystem change on spawning per recruit of Baltic herring. ICES Journal of Marine Science, 60(1): 94–109.

Rumohr, H., and Krost, P. 1991. Experimental evidence of damage to benthos by bottom trawling with special reference to *Arctica islandica*. Meeresforschung, 33(4): 340–345.

Schinke, H., and Matthäus, W. 1998. On the causes of major Baltic inflows – an analyis of long time series. Continental Shelf Research, 18: 67–97.

Schirmeister, B. 2003. Verluste von Wasservögeln in Stellnetzen der Küstenfischerei – das Beispiel der Insel Usedom. Meer und Museum, 17: 160–166.

Silva, A., Azevedo, M., Cabral, H., Machado, P., Murta, A., and Silva, M.A. 1996. Blue whiting (*Micromesistius poutassou*) as a forage fish in Portuguese waters. *In* Forage Fishes in Marine Ecosystems, pp. 127–146. Lowell Wakefield Fisheries Symposium Series, No. 14.

Slobodkin, L.B. 1961. Growth and Regulation of Animal Populations. Holt, Rinehart and Winston, New York, USA.

Stempniewicz, L. 1994. Marine birds drowning in fishing nets in the Gulf of Gdansk (southern Baltic): numbers, species composition, age and sex structure. Ornis Svecica, 4: 123–132.

Strandberg, B., Strandberg, L., Bergqvist, P-A., Falandysz, J., and Rappe, C. 1998. Concentrations and biomagnification of 17 chlordane compounds and other organochlorines in harbour porpoise (*Phocoena phocoena*) and herring from the Southern Baltic Sea. Chemosphere, 37(9–12): 2513–2523.

STECF. 2004. Report of the scientific advice for 2004. SEC (2004) 372.

Tollit, D.J., Greenstreet, S.P.R., and Thompson, P.M. 1997. Prey selection by harbour seals *Phoca vitulina* in relation to variations in prey abundance. Canadian Journal of Zoology, 75: 1508–1518.

Urtans, E., and Priednieks, J. 2000. The present status of seabirds by-catch in Latvia coastal fishery of the Baltic Sea. ICES CM 2000/J:14.

Voipio, A. 1981. The Baltic Sea. Elsevier Oceanographic Series, Elsevier, Amsterdam.

Vuorinen, P.J., Parmanne, R., Vartiainen, T., Keinänen, M., Kiviranta, H., Kotovuori, O., and Halling, F. 2002. PCDD, PCDF, PCB and thiamine in Baltic herring (*Clupea harengus* L.) and sprat (*Sprattus sprattus* L.) as a background to the M74 syndrome of Baltic salmon (*Salmo salar* L.). ICES Journal of Marine Science, 59(3): 480–496

Wanless, S., Harris, M.P., and Greenstreet, S.P.R. 1998. Summer sandeel consumption by seabirds breeding in the Firth of Forth, southeast Scotland. ICES Journal of Marine Science, 55: 1141–1151.

Zilanov V.K. 1984. Blue whiting in the North Atlantic. Legkaja i pischevaja promyshlennost, Moscow. 160 pp. (in Russian).

9 A FRAMEWORK FOR MONITORING ECOSYSTEM HEALTH USING ‘SURVEILLANCE’ AND ‘PERFORMANCE’ METRICS

The terms of reference states:

g) consider a framework for the monitoring of the status of ecosystem components in the ICES area that makes use of both “descriptive surveillance metrics” and “performance metrics”. The developed framework should include a consideration of how data routinely collected as part of ICES activities can be most effectively utilised for the purpose of reporting on ecosystem status, and what measures might ultimately be added to the current incomplete suite of EcoQOs (performance metrics) currently being developed.

9.1 Introduction

The development of a management regime based on Ecosystem Quality Objectives (EcoQOs) is intended to provide an ecosystem approach to management, thus providing a higher level of environmental protection and circumventing some of the shortcomings of single-issue management approaches. This is the aim of a number of initiatives in European Seas, not least the draft European Marine Strategy (EC, 2004). The ultimate aim being to provide a “healthy ecosystem” which can sustain human demands on environmental goods and services. Many of the metrics being advanced to provide managers with an assessment of the health of ecosystem components cover the same aspects of the ecosystem (a large number of diversity measures, for example, or representations of species abundance patterns). It is difficult to see how many of these could be made operational. How does one manage the diversity or species composition of an ecosystem? Of course one cannot. It is impacting activities that can be managed and, in order to assist in that management process, the task for science is to develop the tools needed by the managers. That is to say, metrics that give a good direct measure of the extent to which an activity is impacting the system and so requires a directed management response. ICES (2001) proposed seven criteria for assessing the suitability of a metric to support the EcoQO approach, a good metric of the success of the management regime should fully meet the ICES criteria. However, metrics which are able to inform the management process directly, i.e., Decision Support (“Performance”) metrics, in many cases are likely to be more costly to measure at high frequency or over large spatial scales than the descriptive metrics. If each measure is to comply with ICES criteria ii), iii), and v), this implies that the number of metrics is also likely to be high. The number of metrics required may be at least as great as the number of human activities to be managed.

A more pragmatic approach would therefore be to develop a suite of descriptive metrics that are monitored at relatively frequent intervals over the whole area. Changes in these would initiate the assessment of the decision support or performance metrics. That is to say, changes in a group of metrics based on integrated aspects of environmental condition, such as diversity or total biomass, could then be used to trigger assessment of the performance metrics. By their integrated nature, this class of metrics is poorly linked to specific manageable activities. These integrated metrics represent Environmental Health (“Descriptive” or “Surveillance”) metrics. This approach is analogous to human health monitoring. Health professionals routinely measure pulse, respiration, temperature and blood pressure. These are simple and inexpensive to monitor, and changes in these parameters that do not directly trigger intervention but usually lead to additional diagnostic tests. The key aspect of the diagnostic test is that it establishes, at least with high likelihood, the cause so that treatment (management) measures can be introduced. Diagnostic tests are typically more technically complex and costly than the routine surveillance measures. “Performance metrics” are analogous to the diagnostic, treatment-defining tests. Once treatment is on-going, there remains a need to assess its effectiveness and this may involve a combination of “performance” and “descriptive metrics”.

We illustrate this approach more fully by considering a healthy human community as compared to a healthy fish community. We routinely measure temperature, pulse and blood pressure (fish diversity, species composition, size spectra). A problem in the system is detected by a change in one or more of these metrics, and in addition diagnostic tests are initiated, for example, examination of the blood chemistry (examination of historic and spatial patterns in fishing effort and fish size spectra). This process will start with examination of metrics related to the most likely cause of the problem and continue until the cause is established with high probability. A treatment (management) regime is then initiated. Continued monitoring of the descriptive/surveillance metric and, at least initially, the diagnostic/performance metric then assesses the success of this in correcting the problem. In addition to ensuring that the problem is remedied, this monitoring needs to assess the extent of any side effects/unwanted changes to other components of the system.

The regular environmental survey and monitoring programmes can yield metrics that can be utilised as ecosystem health surveillance metrics. Table 5.3.4.1 in the 2001 WGECO report (ICES, 2001b) provides an extensive, but not exhaustive, list of many of these. It was concluded that most of these were unsuitable as EcoQ metrics, because they were not closely linked to manageable impacting activities. It was, however, noted that for most of these proposed metrics long time series or pre-impact reference data were available and that they were responsive to changes in ecosystem status. These criteria make them acceptable as descriptive surveillance metrics.

In this section, we consider the ecosystem components, for which the status should be routinely monitored to ensure that the ecosystem is being maintained in an appropriate condition, review the extent to which data currently collected as part of ICES activities might be used to provide environmental state and performance measures specifying the need for management action, and further more identifying what measures should be added to these to provide a comprehensive suite of metrics.

9.2 Ecosystem components to be considered

In assessing the ecosystem components to be considered, we were very aware of the fact that no two experts were likely to come up with the same list, and that in part this would be influenced by their personal experience and prejudices. We have endeavoured to reduce the list to the minimum number of elements (Table 9.2.1), subscribing to the view that the lower the number of elements, the easier to obtain a coherent and integrated management regime (see Section 7, above).

Table 9.2.1 WGECO-proposed ecosystem components for which EcoQO elements should be produced in order to provide a holistic framework for ecosystem protection.

Habitats – physical and chemical attributes
Nutrients
Plankton (phytoplankton and zooplankton)
Benthos
Fish community
Commercial fish and shellfish
Marine mammals
Seabirds

WGECO recognises that this classification is artificial, but primarily reflects ecological divisions. However, while we consider commercial fish stocks to be part of the fish community, we recognise that the information needs will differ between the various groups seeking advice and support. Their needs are therefore best served by considering the fish community and commercial fish separately.

The ecosystem components identified in Table 9.2.1 were selected to provide the minimum number of components that need to be managed for while providing adequate coverage of the entire system. "Habitats" is taken to refer to the physical and chemical environment and, hence, includes water quality and the physical (substratum) aspects of the environment. Nutrients include the essential biological nutrients and consideration of their sources, fluxes, and biogeochemical transformations. Plankton (phytoplankton and zooplankton) and benthos both provide food resources, while the former is also comprise parts of nursery environments for benthos and fish. The benthos element also includes their role as structural habitat agents. The fish community includes the whole fish assemblage. Commercial fish and shellfish may be aggregated into five. The top predators, marine mammals and seabirds are, at least for the public, the most conspicuous elements of the marine ecosystem and are often regarded as environmental sentinels.

9.3 Consideration of how data routinely collected as part of ICES activities can be most effectively utilised for the purpose of reporting on ecosystem status

9.3.1 Introduction

WGECO considered that using only "routine" ICES activities limited the scope of reporting on ecosystem status too severely. Instead this term of reference was considered within the broader context of considering any routine data gathering activities that were routinely made use of in ICES work. Thus, seabird colony surveys, seabirds at-sea surveys, and observer schemes to monitor cetacean by-catch in fisheries, although not part of ICES coordinated data-gathering activities, nevertheless are integral to the work carried out by ICES working groups, such as WGSE and WGMME. Recent working group and study group reports were therefore reviewed in an attempt to identify as many such routine data-gathering activities as possible. Because of time constraints, WGECO limited this task to the North Sea only.

9.3.2 Components of the marine ecosystem covered by routine data gathering activities

Table 9.3.2.1 provides a list of routine data-gathering activities that are instrumental to the work of ICES. These activities are divided into broad categories and each activity is identified by survey name where possible. The seasonality of each activity, and its longevity, is indicated. For each data-gathering activity, the table indicates to which component of the ecosystem the activity is primarily directed, which components are addressed by secondary or ancillary data collection, and which components are touched on by occasional data collection. Finally, Table 9.3.2.1 gives a “Data Collected Code”. This links to Table 9.3.2.2, which provides as detailed a description of the gear or samplers deployed during each activity, along with as precise a description of the actual data recorded as possible. Table 9.3.2.2 has a final column labelled “Additional Data Code”. This links to Table 9.3.2.3 where suggestions for the possible recording of additional information from the samples currently collected are made. Finally, Table 9.3.2.4 indicates which sets of data might be used to support a variety of different metrics that could be applied to monitor change in the eight components of the ecosystem under consideration.

Table 9.3.2.1 suggests that the data accrued by routine gathering activities address all ecosystem components. Some components are particularly well covered, for example, commercial fish and fish communities. This is not surprising given that many coordinated surveys are undertaken in support of fisheries management. However, this finding may also reflect the bias of expertise within WGECO, in that the group contains few oceanographers and planktologists. Thus, some routine data gathering activities that are incorporated in the work of ICES that are associated with these areas of marine science may have inadvertently been excluded.

Table 9.3.2.4 again underlines the preponderance of current routine data-gathering activities to support metrics concerned with monitoring change in fish communities and commercial fish populations. The data collected appear particularly suited to metrics that rely on species relative abundance data, and for each of these types of metrics, a number of different data sets can be used. If the additional information suggested in Table 9.3.2.3 were to be collected, then the number of metrics that could be applied to other components of the ecosystem that are currently less well covered is increased. Table 9.3.2.4 also again stresses a point made elsewhere in this report, that while we have plenty of data available to support surveillance metrics, and could easily collect more data to support others, we have little scope to determine many performance metrics. Indeed this table would imply that we have no adequate performance metrics for the habitats, nutrients, and plankton components. We have suggested elsewhere that community structure metrics may provide only weak performance metrics for fish communities, so the situation regarding the availability of performance indicators is perhaps not even as strong as this table might imply. WGECO believes that the paucity of performance metrics indicated by Table 9.3.2.4 reflects the actual situation, although this table may in fact be incomplete. More work is necessary to further develop this set of tables.

In Section 14, WGECO stresses the importance of cross-calibrating the different surveys currently in use, and where possible extending this to calibrate gears currently used with gears that have ceased to be used in survey work. Doing this would help to improve the usefulness of our current surveillance metrics, and could perhaps help to convert some of these metrics into performance metrics.

Table 9.3.2.1. Catalogue of routine surveys currently undertaken in the North Sea and used in ICES work and indications of the components of the ecosystem that can be monitored using the data collected (P = primary objective, S = secondary objective, O = occasional objective). Data collected codes cross link to Table 9.3.2.2 where a detailed list of the sampling equipment used and data collected is provided.

Survey Type	Area	Name	Season	Period	Ecosystem Components								Data Collected Codes
					Habitats	Nutrients	Plankton	Benthos	Fish Community	Fish Commercial	Marine Mammals	Seabirds	
Groundfish	IV	Q1 IBTS	Winter	1971–2004	S,O		S		P	P			1, 6, 7
	IV	Q3 IBTS	Summer	1991–2004	S,O			O	P	P			1, 6, 16
	IV	DBTS	Summer	1985–2004				P	P	P			2
	IV	SNS	Summer	1969–2003				P	P	P			3
	IV	DFS	Summer	1970–2004				P	P	P			4
	IV	EBTS	Summer	1988–2004				S,O	P	P			5
Acoustic	IV	Herring	Summer	1992–2004	S,O		S,O		S	P	O	O	8, 6
	IV	Mackerel	Autumn	c2000–2004	S,O		S,O		S	P			8, 6
Ichthyoplankton	IV	Cod/Plaice	Spring	- 2004	S		S		S	P			9, 6
	IV	Mackerel	Spr/Sum	- 2004	S		S		S	P			10, 6
Landings	IV	Markets	All Year	1963–2004					S	P			11
ByCatch	IV	Discards	All Year	1976–2004					S	P			12
Remote Sensing	IV	MERIS	All Year										
	IV	ENVISAT	All Year										
Oceanographic	IV	Ferry Box	All Year		P	P	P						13
	IV	CPR	All Year	c1930–2004	S,O	S,O	P						14
Benthic Grab	IV	Infauna	Summer	1986 & 2000				P					15
Seabird	IV	Colonies	Spring	1970, 1985–2004								P	17
	IV	Beached	All Year	c1975–2004								P	18
	IV	At Sea	All Year	c1980–2004								P	19
Seal Survey	IV	Grey	Summer								P		20
	IV	Common	Summer								P		21
	IV	Beached	All year								P		18
Cetaceans	IV	Beached	All year								P		18
	IV	Observer	All Year								P		22
	IV	SCANS	Summer	1994 & 2005							P		23

Table 9.3.2.2. Sampling equipment used and data collected in each of the surveys listed in Table 9.3.2.1. Metrics Codes and Additional Data Codes link to Tables 9.3.2.3 and 9.3.2.4, respectively.

Data Collected Codes	Sampling Equipment	Data Collected	Additional Data Codes
1	GOV demersal trawl, 20mm codend	Numbers fish at length in catch, weight- and age-length relationships	A
2	8m beam trawl, 40mm codend	Numbers fish at length in catch, weight- and age-length relationships, no. and weight benthos	A, D
3	6m sole beam trawl, 40mm codend	Numbers fish at length in catch, weight- and age-length relationships, no. and weight benthos	A, D
4	3m beam trawl, 20mm codend	Numbers fish at length in catch, weight- and age-length relationships, no. and weight benthos	A, D
5	4m beam trawl, 40mm codend	Numbers fish at length in catch, weight- and age-length relationships, no. and weight benthos	A, D
6	CTD profiler	Temperature and salinity at 0.5m or 1m depth intervals, calibration salinity samples	B
7	Plankton sampler, MIK or Isaak Kidd	Numbers of herring and sprat larvae (at length)	C
8	EK500, Pelagic Trawl	Acoustic echo integration, numbers fish at length in catch, weight- and age-length relationships	A
9	Gulf VII (Gulf III/Bongo) 40cm diameter 270µm mesh, 100-200m^3 filtered	Number of cod and plaice eggs per volume filtered	E
10	Gulf III (20cm)/Bongo (60cm), 250-280µm mesh, water column oblique tow	Number of mackerel and horse mackerel eggs per volume filtered	E
11	Landed catch sampled	Numbers commercial fish at length in landings, weight- and age-length relationships	F
12	Catch brought aboard sampled	Effort and Catch per trawl quantified, related to vessel and location. Numbers commercial fish at length in catch, age-length relationships. Proportion catch at length discarded recorded	G
13	CTD profiler, water sampler on 9 ferry routes	Temperature and salinity at 0.5m or 1m depth intervals, calibration salinity samples, fluorometry and chlorophyll calibration samples, transmissometry, nutrients, pH, O_2, algal composition	
14	Standard CPR body with Autonomous Plankton Sampler	Geo-referenced “snapshots” of filtered plankton collected on silk band filter, “greenness” index	
15	0.1m^2 van Veen/Day Grabs/0.1 m^2 or 0.25 m^2 Box Core	Abundance of benthic invertebrate species. Sediment particle size distribution, organic content.	H
16	0.1m^2 van Veen/Day Grabs/0.1 m^2 or 0.25 m^2 Box Core/2m Epibenthic beam trawl	Numbers at length benthic invertebrate infauna and epibenthos, weight at length relationships, mean individual weight for productivity. Meiofauna analysis. Sediment particle size distribution. Samples are associated with GFS trawl sample and CTD sampling	H
17	Seabird colony visits, Binoculars/ Telescope	Counts of breeding seabirds/occupied nests, records of egg/chick production	
18	Coastline survey	Location ands counts of dead seabirds/seals/cetaceans by species. Recovery of bird leg-rings.	
19	Vessel or Aircraft based survey	Geo-referenced counts of seabirds in transects of fixed width, abundance/distribution estimates.	
20	Aerial Survey/Population Modelling	Counts of grey seal pup production, models used to estimate population size that would give rise to observed number of pups	
21	Aerial Survey	Count of numbers of moulting harbour seals – minimum population size estimate	
22	Fisheries Bycatch Observer schemes	Counts of numbers of marine mammals taken by different fisheries related to effort.	
23	Shipbased and aerial survey	Geo-referenced counts of cetaceans in transects of fixed width, abundance/distribution estimates.	

Table 9.3.2.3. Additional data that could be collected from the sampling already undertaken during "routine" survey work in the North Sea. Additional Data Codes link to Table 9.3.2.2.

Additional Data Code	**Additional information that could be recorded from the samples collected**
A	Stomach samples for diet/consumption analysis, tissue samples for genetics/contaminants/disease, otoliths for growth analysis. Extension of weight-lenghth and age-length relationships to include more/all species.
B	Additional instruments fitted to CTD (fluorometry/transmissometry), calibration samples (chlorophyll/nutrients), primary productivity
C	Analysis of other fish larvae (sandeels, etc) and other larger zooplankton (euphausids, etc.) in samples
D	Size frequency distributions and weight at length distributions of benthos
E	Identification of all fish eggs and larvae, and other large zooplankton (eg Euphausids). Genetic analysis of eggs.
F	Otoliths for growth rate analysis
G	Numbers at Length for all species in the catch. Proportion of catch at length of all species discarded
H	Full suite of seabed habitat measurements, eg organics, O_2, Chemistry, Nutrients. Meiofauna samples, Numbers of each species at length, weight-length relationships

Table 9.3.2.4. Indication of the type of data collected during "routine" survey work in the North Sea needed to support different types of surveillance or performance metric for each ecosystem component. Figures in the table are the data codes indicated in Table 9.3.2.1 and described in detail in Table 9.3.2.2. Figures in bold are the additional data codes indicated in Table 9.3.2.2 and described in detail in Table 9.3.2.3.

Metric Types	Surveillance Metrics								Performance Metrics							
	Habs.	**Nutrs.**	**Plank.**	**Benth.**	**Fish Commer.**	**Fish Commun.**	**Mar Mamms.**	**Seabirds**	**Habs.**	**Nutrs.**	**Plank.**	**Benth.**	**Fish Commer.**	**Fish Commun.**	**Mar Mamms.**	**Seabirds**
Size Spectra				16, **D**	1, 2, 3, 4, 5,	1, 2, 3, 4, 5, **A**							1, 2, 3, 4, 5			
Communty Structure			14, **C, E**	14, 16, **C, E**	1, 2, 3, 4, 5,	1, 2, 3, 4, 5,	23,	17, 19								
Life History Composition					1, 2, 3, 4, 5,	1, 2, 3, 4, 5,										
Indicator Species	1, 2, 3, 4, 5, 15, 16		14, **C, E**	14, 15, 16, **C, E**	1, 2, 3, 4, 5,	1, 2, 3, 4, 5,		17, 19			14, **C, E**	16				
Rare Species			14, **C, E**	15,												
Threatened/Declin-ing spp.				15, 16	1, 2, 3, 4, 5,	1, 2, 3, 4, 5,	23,	17,								
Abundance/Biomass/ Concentration		H	B	15, 16	1, 2, 3, 4, 5, 7, 8, 9, 10	1, 2, 3, 4, 5,	20, 21, 23	17, 19		H	B					
Spawning Stock Biomass					11, 12								11, 12			
Fishing Mortality					11, 12, **G**	G							11, 12			
Kittiwake Breeding Succ.								17,								17
Contaminant Loads	**H**	**H**				**A**	18,	18,							18	18
Habitat Quality	6, 16,															
Water Temp./Salinity	6, 13,	13,	13, 14,	14,												
Marine Mammal By-catch							22,								22	
Productivity			**B**		1, 2, 3, 4, 5, **A**	**A**					**B**					

9.4 A framework for monitoring the ecosystem: A three-stage approach

It is not for science to set objectives, but science has a role in advising society of achievable and verifiable objectives and considerable effort will need to be invested by scientists in working with society/stakeholders in formulating the actual objective. For the purposes of this discussion, we will consider the desired objective of the management regime to be a "healthy system".

We propose the application of a three-stage approach. Firstly, there should be development of a programme of "health" monitoring covering the condition of each component. Where this programme identifies deviations from the normal range, implementation is set in place of a second phase of measurements designed to inform managers of the cause of the change and hence the measures to be taken to return the system to health. As management measures are introduced, then the system needs to be monitored to ensure that the management measures deliver the desired changes and not any unwanted side effects.

9.4.1 Ecosystem health monitoring – Descriptive surveillance metrics

A large amount of the data routinely collected by survey and monitoring programmes can yield metrics which can be utilised as ecosystem health surveillance metrics. WGECO 2001 Table 5.3.4.1 (ICES, 2001) provides an extensive, but not exhaustive, list of many of these. WGECO concluded that most of these were unsuitable as EcoQ metrics, because they were not closely linked to manageable impacting activities; however, it was recognised that for most proposed metrics, long time series or pre-impact reference data were available and they were responsive to changes in ecosystem status. These criteria make them acceptable as descriptive surveillance metrics.

The first element of the monitoring framework should comprise regular monitoring of a suite of descriptive surveillance metrics for each ecosystem component (Table 9.2.1). WGECO considers that working groups with detailed knowledge of each of the components should now be tasked with reviewing existing date series and metrics and proposing appropriate monitoring regimes including details of data to be collected, spatial scale and location of sampling, timing and frequency of monitoring.

In developing their proposed monitoring scheme, specialist working groups should be specifically asked to identify the criteria or range of values, deviations which would trigger diagnostic investigations.

9.4.2 Diagnosing causes of deteriorations in health – Performance diagnostic metrics

When the behaviour of any SINGLE surveillance metric triggers the criteria for diagnostic investigations, managers should initiate a review of the likely causes. The values/conditions that initiate management action should be prescribed in the decision control rules set out in the management plan.

The diagnostic investigations are a technical exercise and will make recommendations as to the follow-up investigations that need to be initiated. In many cases, a management scheme could be developed where these investigations are prescribed in advance for various scenarios of behaviour of the surveillance metrics.

The purpose of the diagnostic studies is to establish, with high likelihood, the cause of the deterioration in the health of an ecosystem component. When this "causality" is established, appropriate management actions can be initiated.

It should be recognised, however, that in many cases, especially when operating in a precautionary framework, management action to control the most likely causes of the change will need to be initiated even before the diagnostic investigations have run their course.

9.4.3 Monitoring the effectiveness of the management regime – a combination of metrics

WGECO acknowledges that many management measures will have social or economic costs. It is therefore important that it can be quickly established whether the corrective management measures taken are delivering the desired output. If not, management measures will need to be adjusted.

For most management measures, there is no guarantee of success and there will be a risk of unwanted impacts on the system caused by the management measure (i.e., side effects). Monitoring of the performance of the management regime will need to be developed to ensure that (i) the management changes have the desired effect on the ecosystem component whose status triggered action, and (ii) there are no unintended effects on other ecosystem components. This will probably involve a limited period in which both the diagnostic/performance metrics are measured in addition to the suite of descriptive surveillance metrics. In due course, if the management measure(s) are effective and there are no adverse effects, the monitoring programme will be able to revert to the base surveillance regime.

9.5 References

EC. 2004. Guidance on how to develop the ecosystem approach to managing human activities in the marine environment. (Draft) EAM(2) 04/3/1 Brussels.

ICES. 2001. Report of the ICES Advisory Committee on Ecosystems, 2001. ICES Cooperative Research Report, No. 249, 94 pp.

ICES. 2001. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 2001/ACME:09, 106 pp.

10 TOR H, REVIEW ECOSYSTEM RESPONSES TO SPATIAL REDUCTIONS IN FISHING ACTIVITIES, AND DESCRIBE BIOLOGICAL DEVELOPMENT IN THESE AREAS; REVIEW GUIDELINES FOR ESTABLISHMENT OF MPA AND RECOMMEND REVISIONS

10.1 Introduction

In 2002 the World Summit on Sustainable Development (WSSD) reaffirmed sustainable development as a central element of the international agenda, and Governments agreed to a wide range of commitments and targets for action to achieve more effective implementation of sustainable development objectives. Such commitments for the marine environment included encouraging the application of the ecosystem approach for the sustainable development of the oceans by 2010, achieving by 2010 a significant reduction in the current rate of loss of biological diversity, and establishing representative networks of Marine Protected Areas by 2012 (WSSD, 2002). In the expectation that this final obligation to establish MPA becomes an increasingly important focus of international activity, WGECO has begun an assessment of ecosystem responses to MPA, and of existing guidelines to aid their development.

The Term of Reference was to "review data on ecosystem responses to spatial reductions in fishing activities in temperate freshwater and marine areas and describe similarities and differences in the biological development of these areas. Particular considerations should be given to differences in ecosystem development in response to the geographical position/scale of the studied areas and our understanding of meta-population dynamics. Review published guidelines for the establishment of marine protected areas and recommend revisions".

This term of reference was broadly divided into two sections, one dealing with case studies of closed areas and their results (Sections 10.2–10.7) and the other dealing with the published guidelines (Section 10.9). Both the case studies and the guidelines have been evaluated (Sections 10.8 and 10.10) and an evaluation of the guidelines with respect to the case studies has been carried out (Section 10.11). Given the time available, this has necessarily been a limited review of the data that were readily available and the choice of cases and guidelines does not necessarily reflect a value judgement on the part of the WG.

10.1.1 Evaluation criteria for Closed Areas / Marine Protected Areas

In order to be able to compare the different case studies, a set of criteria was chosen. Information for the different sites was then tabulated according to the criteria, and for each site a more detailed description of the observed changes is given in the text.

10.2 Georges Bank

10.2.1 Area information

We are very grateful to Steve Murawski for providing us with unpublished data to enable us to successfully complete the section on Georges Bank.

- Purpose of closure: To provide protection for fishery stocks (Atlantic cod, haddock, and yellowtail flounder) within the Northeast Multispecies Fishery Management Plan.

- Site: Georges Bank Closed Area I, Closed Area II, and Nantucket Lightship. See map for locations (Figure 10.2.1.1). The area provides a suitable spawning habitat for groundfish stocks. The bottom substrate is a gravel-sand mixture that provides shelter for fish and invertebrates. CA-II comprises a seabed substrate of gravel and sand with extensive sand waves to the south. Adult and juvenile cod and haddock are present in both CA-I and CA-II in winter and spring, but in summer and autumn adults and larger juveniles move out of CA-II to cooler, deeper waters, north and east of the area. CA-I retains significant numbers of cod and haddock in all seasons. The Nantucket Lightship area comprises gravel, sand boulders in the northeast and sand, silt and clay further south and west. Yellowtail, winter and windowpane flounders, little skate and winter skate occupy the areas virtually all year round. The area is also home to large numbers of shrimp, polychaetes, brittle stars and mussels. The activities and issues of concern are: overfishing/overexploitation of resources, taking the broodstock before spawning; concerns of fishing gear destroying the habitat, and fishing-related habitat impacts.

- Type of closure:

    i. Gear/fleet type: All gear capable of catching groundfish are banned (trawls, scallop dredges, gillnets, hook fishing). Pelagic gears (longline, hook and line or harpoon) are allowed as long as the regulated species are not retained, if they are caught. Vessels in the area may not have other gear on board capable of catching the regulated species. Pelagic midwater trawl gear is also allowed, under strict conditions. In CA-II, all pot gear for lobsters or hagfish are allowed, provided that none of the regulated species are taken, and that there is no other gear on board capable of catching the relevant species. Activities allowed with restrictions or permits are the following: commercial bottom trawling, commercial use of traps, consumptive recreational fishing, extractive research, and other commercial fishing.

    ii. Absolute size: CA-I = 3,960 km^2 ; CA-II = 6,927 km^2; Nantucket Lightship (NLS) = 6,275 km^2.

    iii. Spatial scale (e.g. % of area): CA-I and CA-II represent 21%, 17%, and 29%, respectively, of the area occupied on Georges Bank for cod, haddock, and yellowtail flounder. The Nantucket Lightship area represents about 22% of the New England yellowtail flounder's range (Murawski *et al.*, 2000).

    iv. Time scale: December 1994 emergency closure was extended until April 1995, and is a year-round closure from 18 April 1995 to present. The vicinity of the area has been subject to seasonal closures since 1982. In 1999 it was decided to reopen a portion of the closed area, south of 41° 30' N, for a limited scallop fishery (Rago and McSherry, 2001). This covers the southern part of CA-II. TAC for scallops was 4,257 t, TAC for yellowtail flounder was 4.54 mt/trip ,and there was a restriction on the total number of trips per vessel. Total scallop landings and yellowtail flounder by-catch were monitored on a daily basis, and the area would be closed whenever the set limits were attained. A 10 nm buffer was closed around CA-II to improve enforcement of the closed area, and there was also 25% observer coverage for the closed area.

- Did closure meet the objectives: Yes. Fishing mortality was reduced and SSB increased for cod, haddock, and yellowtail flounder in CA-I and CA-II and to a lesser extent for New England (NLS) yellowtail flounder. The mean density of haddock has increased significantly, especially in CA-I, since the closure. Recent reports speak of 'a particularly compelling demonstration of reserve siting (CAI) that is in concordance with the life-history, habitat preferences and movement patterns of one of the primary resource species the reserve was intended to help conserve' (Murawski *et al.*, in press).

- What (other) unexpected factors influenced the outcome: The success of the closed areas is also due to the wider-spread management measures imposed by Canada and the U.S. These include reduced effort, trip limits, increased mesh size and a reduction in TAC in Canada.

- How long did it take for implementation: Following recommendations from scientists for immediate reductions in fishing mortality, the Secretary of Commerce closed Areas I and II and NLS in December 1994. This emergency authority was only valid for 180 days and in April 1995 the closure became permanent following an amendment by the New England Fishery Management Council. On 30 March 1982 a different configuration of the area was first closed on a seasonal basis through a Secretarial Emergency Action.

- Enforcement

    i. Type: US coastguard and National Marine Fisheries Service are responsible for enforcement of laws and regulations. VMS data have been made available since 1998.

    ii. Effectiveness: High levels of dedicated ship and aircraft patrol, significant and increasing penalties for violators and the introduction of VMS acting as a deterrent to incursions have all led to a high level of observance of the measures.

- Is the closed area part of a network of closed areas: Yes. There is an area of more than 17,000 km^2 that is closed to fisheries year round. A total of 50,000 km^2 are closed on a seasonal basis, the so-called 'rolling closures' (Figure 10.2.1.1).

10.2.2 Observed changes - within closed area

- Effort: Levels of fishing effort have declined.

- Focal species: In CA-I, only haddock and yellowtail flounder were caught at consistently and significantly higher catch rates (kg/h) within the closed area than outside, for both seasonal surveys (spring and autumn) (Murawski *et al*., in press). The mean density of haddock expressed as mean weight per tow increased significantly following closure, with a ratio of 'inside versus outside' of more than 10:1 (Murawski *et al*., in press). It can therefore be concluded that the reserve has been successful. Absolute catch rates of Atlantic cod were low and highly variable, although the catch rates within the closed area were 2–3 times higher inside than outside (Murawski *et al*., in press). In CA-II there were elevated catch rates (kg/h) for yellowtail flounder in spring and autumn, and elevated catch rates for Atlantic cod and haddock in spring. Exploitation rates decreased and SBB increased for four stocks following closure. The stocks are: Georges Bank cod, haddock and yellowtail flounder and, to a lesser extent, S New England (area NLS) yellowtail flounder. Increase in SSB was due to increased survival of adults and relatively high rates of somatic growth (NEFSC, 1999, in Murawski *et al*., 2000). Fishing mortality was effectively reduced on the western spawning components of haddock, cod and yellowtail flounder. This area (CA-I) comprises a large amount of the stock, which has a limited dispersal. For the eastern spawning stock (CA-II), protection is high in the first half of the year. The measures taken outside the area have also contributed to the reduction in fishing mortality for these stocks. On Georges Bank, the instantaneous fishing mortality rates decreased between 1994 and 1997 by 66%, 62%, and 88%, respectively, for haddock, cod, and yellowtail flounder. CA-I is essential for protection of haddock spawning stock. The NLS area protects only a small part of the yellowtail flounder resource, and although SSB increased and exploitation decreased, depleted stock sizes and poor year class, combined with fishery management outside the area, meant that the stocks did not significantly increase (Murawski *et al*., 2000).

- Commercial fish species and fish community: Based on CPUE values winter flounder, spiny dogfish, and pollock were more abundant, within the closed area than outside in the autumn survey, but not in spring (Murawski *et al*., in press). For American plaice the opposite was the case, with higher catch rates (kg/h) inside the closed area than outside in spring, but not in autumn. There were elevated catch rates for winter flounder in spring and autumn and elevated catch rates for pollock in spring. Despite this, total catch numbers and weights did not show a strong density differential between open and closed areas (Murawski *et al*., in press). All areas have sedentary groundfish species (e.g., winter flounder, windowpane flounder, winter skate and little skate), which have benefited from the closure, although exact details are not available at the moment.

- Benthos: Atlantic sea scallop (*Placopecten magellanicus*) biomass increased fourteen-fold in the closed areas during 1994-1998. In July 1998 the total and harvestable biomasses of scallops were nine and fourteen times higher, respectively, in the closed as compared to the open areas. The results from the limited reopening of the scallop fishery have led managers to review a formal 'area rotation' scheme for scallop fisheries with the intention of increasing yield per recruit (Murawski *et al*., 2000).

- Habitats: Not evaluated.

- Marine mammals: There is an active marine mammal monitoring programme in the region, focusing primarily on the great whales and seals and harbour porpoise. These species are assessed about biannually. Much of the by-catch monitoring is driven by the need to document deaths of harbour porpoise. These data are not evaluated here.

- Birds: Routine survey data onboard trawl survey research cruises were collected, but this has been discontinued. The data are not evaluated here.

- Other: Sea turtles are relatively common and there are a number of ongoing survey efforts and by-catch monitoring of fisheries in which they are encountered. The data are not evaluated here.

10.2.3 Observed changes – directly adjacent to closed area

- Focal species: There is a substantial year-round stock density differential, although enough haddock moves up to 20 km outside the closed area to increase catch rates there (Murawski *et al*., in press). Movement of haddock to the open area surrounding CA-I is so localised that abundance indices outside have not improved. It is likely that the high catches at the boundary of the closed area are responsible for preventing higher resource abundance further away (Murawski *et al.,* in press). The haddock movements are probably related to spawning, and CPUE adjacent to the closed area is lowest in March/April when spawning peaks, and highest (> 3,000 kg/h) in June-August following spawning and coinciding with higher water temperatures (Murawski *et al.*, in press). The relationship between CPUE and minimum distance, based on sea sampling and survey data, shows two peaks, one at about 20 km from the area, the other at around 100 km. A similar pattern was seen for CA-II (Murawski *et al*., in press).

- Benthos: Although there are no data directly from adjacent areas, surveys have shown that the increase in numbers of large individuals and increase in biomass occurring in the closed area were not mirrored outside. Absolute numbers of scallops increased both inside and outside the closed area, although more in the former, but the size distribution remained heavily biased to small individuals outside the closed area (Murawski *et al*., 2000).

- Other aspects (commercial species, fish community, habitats, birds, marine mammals): Data were available, as described above, but not evaluated here.

10.2.4 Observed changes – away from closed area

- There were peaks of haddock CPUE at discrete intervals (20 km and > 100 km) away from the closed areas (Murawski *et al*, in press). The authors analysed the data to determine spill over effects, which will be discussed in the evaluation of case studies in Section 10.8 below.

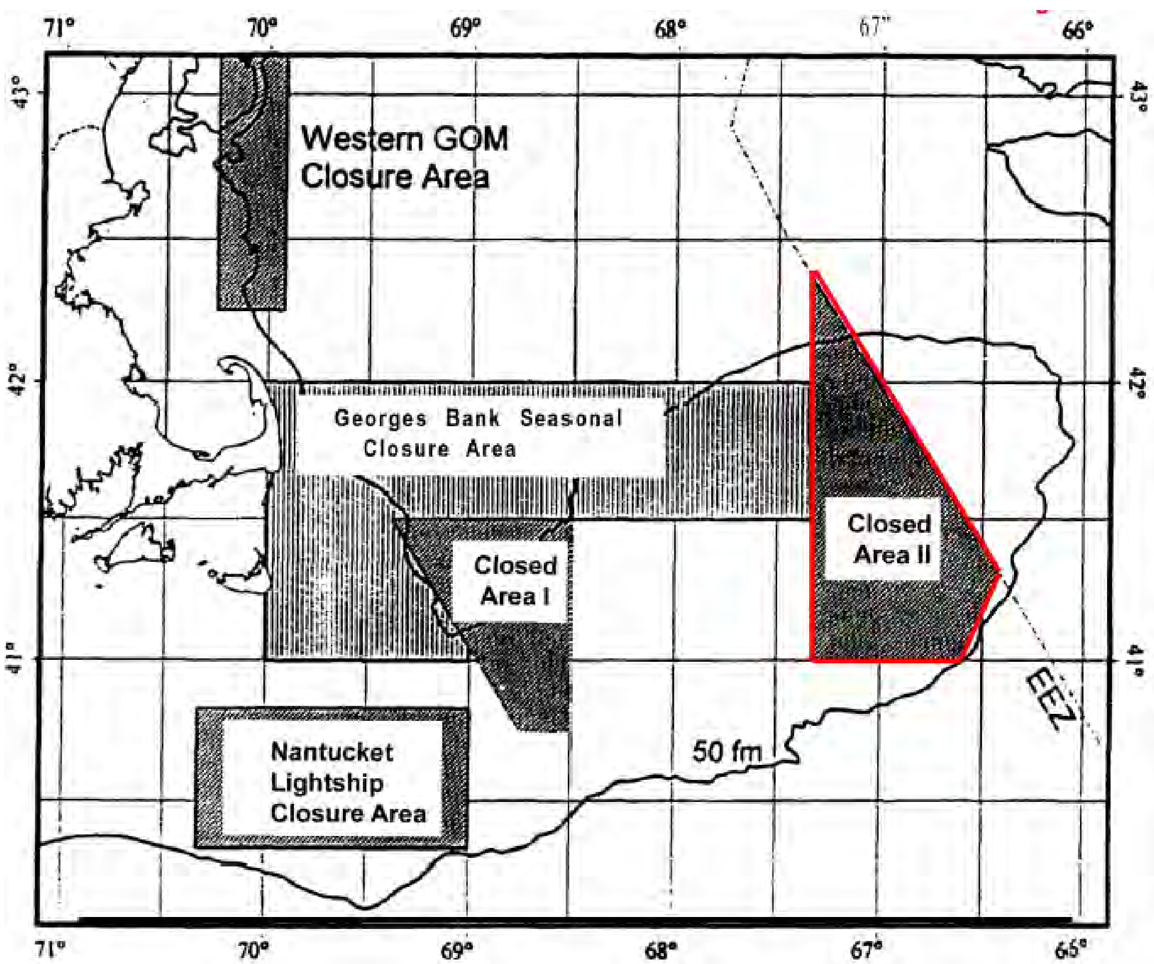


Figure 10.2.1.1 Map of the Georges Bank closed areas.

10.3 Inshore Potting Agreement (IPA) area, Devon

10.3.1 Area information

- Purpose of closure: In this area, towed gears occupied more inshore waters in the mid-1970s leading to strong conflicts with the pot and gillnet fishery. An area was therefore closed for towed gears to reduce the conflict with fishermen using static gears (Figure 10.3.1.1).

- Site: The shallower, mostly hard-bottom areas outside Devon from the coast to 6 n.m. was affected. Information was collected from Kaiser *et al*. (2000), and although no formal study of changes to the fish stocks has been performed, data on fish abundance were gathered through interviews with fishermen (Blyth *et al*., 2002). However, one study of effects on epibenthic and habitat changes has been performed and these data are summarised below.

- Type of closure:

    i. Gear/fleet type: The pot and gillnet fisheries were given exclusive access to some areas and the towed gear fleets were excluded from these areas and moved more offshore. This increased the pot fisheries inside the closure, due to decreased risk of losing gears and probably also increased effort of towed gears outside the closure.

    ii. Absolute size: 478.4 km^2

    iii. Spatial scale: 73% was permanently allocated to static gear and 27% allocated on a seasonal basis.

    iv. Time scale: 1978 ongoing.

- Did closure meet objectives: Yes, conflicts have decreased.

- What unexpected factors influenced the outcome: None.

- How long did it take for implementation: Towed gears expanded their area in the mid-1970s at the expense of the pot fisheries and in 1978 the first agreement was in place.

- Enforcement:

    i. Type: Initially it was voluntary but it was legislated for in 2002.

    ii. Effectiveness: Good for the whole period except on the edges of the area.

- Is the closed area part of a network: No.

10.3.2 Observed changes – within closed area

- Effort: Probably decreased for demersal fish, but there is still gillnet fishing within the area. Pot fisheries have increased.

- Focal species or habitat: There were no observations of abundance changes for either the crab or the demersal stocks. However, there were no formal studies to assess this.

- Commercial species: No obvious abundance changes, but there are larger fishes (rays included) inside the area according to those fishermen interviewed.

- Fish community: There are anecdotal records of larger individuals within the area according to those fishermen interviewed, but no other data.

- Benthos: More large-bodied, fragile epibenthic species occur inside. The infauna communities seem to be less stressed, and commercially exploited crustaceans are protected.

- Habitats: Habitat-forming species are protected as well as the three-dimensional structure of the seabed. The protected area could serve as a seeding area for scallops that are commercially overexploited in the area.

- Marine mammals: No data.

- Birds: No data.

10.3.3 Observed changes – directly adjacent to closed area

- Effort: The fact that the trawl fisheries were moved out of the closed area suggests that effort may have increased adjacent to the closed area.

- Focal species or habitat: There are no data covering these criteria.

- Commercial species: There are some indications of smaller-sized fishes occurring, however, the closed area is probably too small to provide spill-over effects to adjacent areas of demersal fish species.

- Fish community: No data.

- Benthos: The epibenthos is dominated by small, mobile, and resilient fauna. Infauna is more stressed compared to those communities inside the closure.

- Habitats: Habitats are flattened out and "less three-dimensional".

- Marine mammals: No data.

- Birds: No data.

10.3.4 Observed changes – away from closed area

- There are no data to evaluate these criteria.

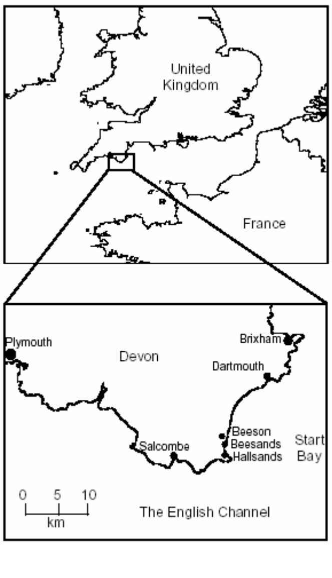


Figure 10.3.1.1. Map showing the location of the Inshore Potting Agreement area, Devon, UK.

10.4 Trawl exclusion area, Öresund

10.4.1 Area information

- Purpose of closure: Use of all towed gears was banned in 1932 due to the danger of using these in this area, which is heavily used by shipping traffic (Figure 10.4.1.1).

- Site: The area is the narrow sound between Sweden and Denmark, 8–40 m deep. It is approximately the same area as the ICES 'area 23' except for a small northern part. Information was collected from an unpublished manuscript, based on IBTS and local coastal surveys (Svedäng *et al*., unpublished).

- Type of closure

  i. Gear: All towed gear (trawls, purse seines, etc.).

  ii. Absolute size: approximately 1700km^2.

  iii. Spatial scale: Not applicable but the area includes local spawning areas for cod and probably many other species as well.

  iv. Time scale: The ban was started in 1932 and has continued since then.

- Did closure meet objectives: Yes, but objectives were not biological, so biological consequences are just a side effect of the closure.

- What unexpected factors influenced the outcome: The fact that there is local spawning within the area and most probably a local population of at least cod is likely to have enhanced the effect of the closure on the cod population.

- How long did it take for implementation: This information is hard to get since it was in 1932.

- Enforcement

  i. Type: The coastguard is responsible for the enforcement. There is cooperation between the Danish and Swedish coastguards.

  ii. Effectiveness: There is ongoing trawling in the area, both from boats passing through and from trawlers (mostly Danish) in the northern part. The coastguard stops a couple of boats each year, but fines are too low (only trawl gears are expropriated) in comparison to the very high profits. The fishermen are also cooperating, using look outs and a collective "fine-fund" to pay for expropriated gears.

- Is the closed area part of a network: No.

10.4.2 Observed changes – within closed area

- Effort: It is much lower than in the surrounding areas, but has increased since the implementation of the ban since fishing with gillnets and recreational fishing is still allowed. Reduction of effort compared to before the ban is hard to evaluate and not relevant due to the large efficiency increase of the gears since 1932.

- Focal species or habitat: There is none due to the purpose for the closure.

- Commercial species: The catches of gadoids are generally much larger in the Öresund, especially for large individuals (see below for details). Figures are based on mean CPUE in 2001–2002 and compare data from Öresund with the Kattegat. (Percent caught in Öresund compared to the Kattegat);

Species	Length Range (cm)	% Change
Cod	> 50	10700
Cod	30><50	2300
Cod	<30	48
Haddock	>30	52500
Haddock	<30	107
Whiting	>30	512
Whiting	<30	22
Plaice	>30	1260
Plaice	<30	16
Lemon sole	>30	2330
Lemon sole	<30	840

The net fishing that now dominates does not catch as large individuals as a trawl and is not as effective. Tagging studies indicate a northward migration of spawning cod during the spawning season but not during the rest of the year. The cod population has been genetically differentiated from other populations.

- <u>Fish community:</u> Changes in the community are hard to assess because there are no data available from the time period before the ban. Since there is a pronounced salinity gradient from the Baltic, through the Öresund and up through the Kattegat, there will be a natural increase in the number of species going north. A comparison with adjacent areas is therefore not appropriate as a background for changes in the community composition.

- <u>Benthos:</u> See fish community above.

- <u>Habitats:</u> See fish community above.

- <u>Marine mammals:</u> no data.

- <u>Birds:</u> no data.

10.4.3 Observed changes – directly <u>adjacent</u> to closed area

- There are no data to evaluate these criteria.

10.4.4 Observed changes – <u>away</u> from closed area

- There are no data to evaluate these criteria.

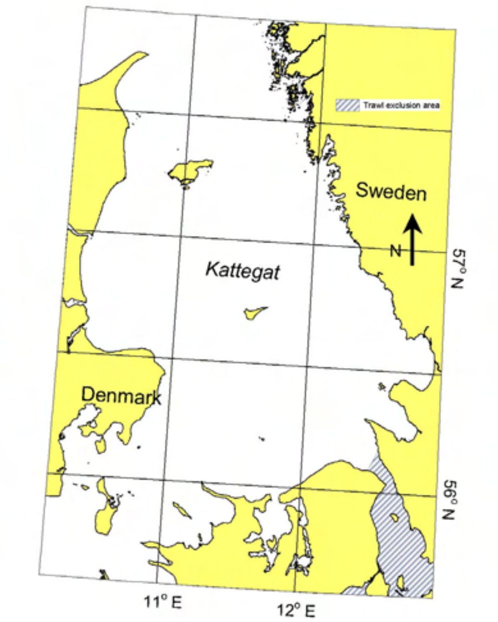


Figure 10.4.1.1 Map of the trawl exclusion area, Öresund closed area.

10.5 Scottish east coast sandeel closure

10.5.1 Area information

- Purpose of closure: The European Council justification reads: "Recent scientific advice indicates that quantities of sand eels within an area off the northeast coast of England and the east coast of Scotland are currently insufficient to support both fisheries upon them and the requirements of various species for which sand eels are a major component of their diet and that a closure of fisheries for sandeels in this area is therefore required" (Figure 10.5.1.1).

- Site: The closed area (ICES rectangles: 40E8–44E8; 44E7–44E8) surrounding the Firth of Forth region with major sandeel fishing banks Wee Bankie and Marr Bank. Sandeel aggregations in the Firth of Forth appear to be reproductively isolated from the major fishing areas at Dogger, Klondyke, and Fisher banks, although they may receive an influx of young of the year sandeels (0-group) from more northerly grounds. Evidence for this comes from the spatial and temporal distribution of sandeels before and after settlement (Wright *et al*., 1998), the distribution of fished areas (Jensen *et al*., 2001), tagging studies (Kunzlik *et al*., 1986) and simulations of larval transport (Proctor *et al*., 1998). Black-legged kittiwake diets on the Isle of May, the largest black-legged kittiwake colony in the Firth of Forth and the closest to the Wee Bankie, consists of 1+ aged sandeels in May, but in June and July they switch almost entirely to 0-group sandeels (Wanless *et al*., 2002). The fishery in the area was prosecuted in June in most years and almost exclusively targeted 1+ sandeels (Pedersen *et al*., 1999).

- Type of closure

    i. Gear/fleet type: Sandeel fishery prohibited. The sandeel fishery is a small-meshed <8mm trawl fishery with a 20m high and 100m wide trawl with very large front-end mesh size (25m) gradually narrowing down to the small meshed cod end. The gear is provided with large light bobbins and trawls with minimum contact with the seabed. The vessel size ranges between 50 and 900 GT with an average GT of about 400 t.

    ii. Absolute size: ca. 21,000 km^2

    iii. Spatial scale (e.g. % of area): The regional sandeel area 3 (Wright et al. 1998) contains about 8–10 sandeel banks and the closed area covers about 90% of that.

iv. Time scale: Restrictions on fishing for sandeels. (1). During the years 2000, 2001 and 2002, it shall be prohibited to land or retain on board sandeels caught within the geographical area bounded by the east coast of England and Scotland, and a line sequentially joining the following coordinates:

-the east coast of England at latitude 55° 30'N,
- latitude 55° 30'N, longitude 1° 00'W,
- latitude 58° 00'N, longitude 1° 00'W,
- latitude 58° 00'N, longitude 2° 00'W,
- the east coast of Scotland at longitude 2° 00'W.

(2). Before 1 March 2001 and again before 1 March 2002, the Commission will report to the Council on the effects of the provision contained in paragraph 1. On the basis of the said reports, the Commission may propose appropriate amendments to the conditions indicated in paragraph 1."

The closure was prolonged by the EC in 2002 for the period 2003-2005, allowing a limited monitoring fishery of 40 fishing days per year.

- Did closure meet the objectives: Sandeel abundance in the area has increased but consequences for top predators have not been fully assessed as yet.

- What (other) unexpected factors influenced the outcome: None

- How long did it take for implementation: The UK called for a moratorium on sandeel fishing adjacent to seabird colonies along the UK coast and, in response, the EU requested advice from ICES. An ICES Study Group was convened in 1999 in response to this request with two terms of reference (ICES, 1999):

- Enforcement

  i. Type: Sandeel vessels carry satellite monitoring equipment .

  ii. Efficiency: The VMS data has not been reviewed scientifically but it would be about 100% efficient.

- Is the closed area part of a network of closed areas: There are a number of separate sandeel banks within the area, but the nearest other closed sandeel area is around Shetland.

10.5.2 Observed changes - within closed area

- Effort: Has been reduced to 10% in years 2000–2003 compared to 1991–1999.

- Focal species or habitat: Kittiwake breeding success does not show a clear signal of improvement in the years when the closure has been in effect, although trawl surveys indicate local sandeel abundance has increased.

- Commercial fish species: Monitoring fishery on sandeels does not show any consistent changes in CPUE. Trawl surveys indicate a decline in cod, haddock and whiting.

- Fish community: no data available.

- Benthos: no data available.

- Habitats: no data available.

- Marine mammals: some evidence of increased usage of area by some marine mammals.

- Birds: some evidence of increased usage of area by some seabirds.

- Other: no data available.

10.5.3 Observed changes – directly adjacent to closed area

- Effort: Has increased to 145% in years 2000–2003 compared to 1991–1999 (in the nine ICES rectangles to the east of closed area in region 3, see Figure 10.5.1.1)

- Commercial fish species: Sandeel catches have increased. Other fish species are not evaluated.

- Fish community: Benthos, Habitats, Marine mammals, Birds, Other: There are no data to evaluate these criteria.

10.5.4 Observed changes – away from closed area

- There are no data to evaluate these criteria.

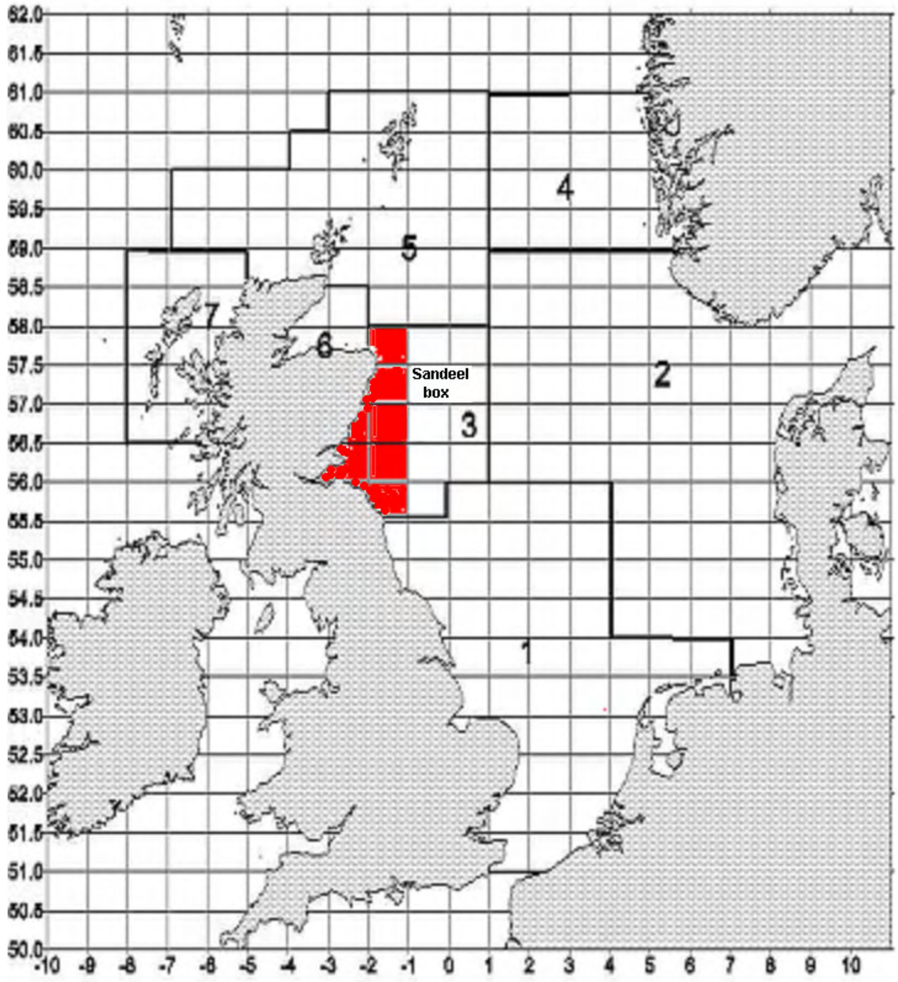


Figure 10.5.1.1 The area closed for sandeel fishing off the east coast of Scotland from 2000 onwards.

10.6 Plaice box

10.6.1 Area information

- Purpose of closure: To reduce the discarding of undersized plaice (*Pleuronectes platessa*) in the main nursery areas, and thereby to enhance recruitment to the fishery (Figure 10.6.1.1).

- Site: The ‘‘plaice box’’ is a partially closed area along the Dutch, German and Danish coast in the North Sea. Information was collected from Piet and Rijnsdorp (1998), Smit *et al.* (1997), Van Beek *et al.* (1998), Pastoors *et al*. (2000).

- Type of closure

  i. Gear/fleet type: It was closed to trawlers with engine power >300 hp.

  ii. Absolute size: not estimated.

  iii. Spacial scale (e.g., % of area): The whole area was covered.

  iv. Time scale: Established in 1989. Initially it was closed in the 2nd and 3rd quarter. From 1994 it was closed in the 2nd to 4th quarters and for the whole year since 1995.

- Did closure meet objectives: No, the closure did succeed in reducing the amount of fishing effort in the box but probably did not succeed in reducing discarding of undersized plaice. The recruitment to the fishery did not increase and has only decreased after the area was closed

- What unexpected factors influenced the outcome: Possibly due to a climatically induced increase in water temperature, juveniles moved to deeper waters and thereby decreased the effect.

- How long did it take for implementation: Implementation took two years as the scientific basis was in 1987 by the ICES North Sea Flatfish WG. Note however, that a similar proposal was already made in 1921 (!) by the ICES Plaice Committee.

- Enforcement

  i. Type: 1993-present black box and VMS data could be used to study the micro-distribution of the Dutch fleet.

  ii. Efficiency: close to 100%.

- Is the closed area part of a network: No.

10.6.2 Observed changes – within closed area

- Effort: The effort data showed that the temporary closure resulted in reduction of effort by about 60%, while the year-round closure resulted in a 94% effort reduction. Most of the effort was displaced to the areas just outside the plaice box.

- Focal species or habitat: It was difficult to contribute any of the changes to the effect of the closure of the box as there were confounding effects such as a marked increase in temperature in the south-astern North Sea that more or less coincided with the closure of the box. Other relevant observations were that juvenile plaice was observed to move to deeper water (outside the box), thereby reducing the effect of the box. Inside the box, growth rate declined and natural mortality went up (Pastoors *et al*., 2000). Discard information shows that plaice discards are about 78% in the box area.

- Commercial species: There were changes observed inside the closed area, but these hardly differed from those observed outside the box and could not unambiguously be attributed to the closure (Piet and Rijnsdorp, 1998).

- Fish community: There were changes observed inside the closed area, but these hardly differed from those observed outside the box and could not unambiguously be attributed to the closure (Piet and Rijnsdorp, 1998).

- Benthos: There were changes observed inside the closed area, but these hardly differed from those observed outside the box and could not unambiguously be attributed to the closure (Piet and Rijnsdorp, 1998).

- Habitats: not evaluated.

- Marine mammals: not evaluated.

- Birds: not evaluated.

- **Observed changes – directly adjacent to closed area**

- Effort: Displaced from inside the box, e.g., increased.

- Focal species: Outside the box area, discards of juvenile place were 31% before the closure of the box and increased to 74% in the period 1999–2000.

- Commercial species: Outside the box area sole discards only showed a minor increase from 12% to 19%

10.6.3 Observed changes – away from closed area

- There are no data to evaluate these criteria.

Figure 10.6.1.1 Map showing the location of the plaice box.

10.7 Cod box

10.7.1 Area information

- Purpose of closure: The area was closed to protect the spawning population of cod.

- Site: In 2001 a large area in the North Sea was closed from 15/2 – 30/4 2001 for all cod-related fishing fleets to protect the spawning population of cod. Fishing activities were monitored using VMS, the biota (demersal fish and benthos) by several bottom surveys (Figure 10.7.1.1).

- Type of closure

    i. Gear/fleet type: All cod-catching gear.

    ii. Absolute size: not estimated.

    iii. Spatial scale (e.g. % of area):

    iv. Time scale: 15/2 – 30/4, 2001.

- Did closure meet objectives: No.

- What unexpected factors influenced the outcome: Bad timing and positioning of the area resulted in that no positive effects of the closure were achieved.

- How long did it take for implementation: Months.

- Enforcement

    i. Type: VMS

    ii. Effectiveness: Very good. During that period target effort was reduced by (probably) 100%.

- Is the closed area part of a network of closed areas: No.

10.7.2 Observed changes – within closed area

- Effort: During the period target effort was reduced by (probably) 100% and displaced to the outside boundary of the box and fishing grounds outside the North Sea.

- Focal species: No beneficial effects of the closure were observed for the cod population as both the positioning and timing of the closed area were not well chosen. The closed areas only partially overlapped with known spawning grounds. In the southern grounds, peak spawning takes place from weeks 4–7 and probably somewhat later further north. The box was closed weeks 8–17 so it probably only protected the second part of the spawning season.

- Commercial fish species: Catches of commercial species were higher after re-opening but returned to normal after 2–3 weeks.

- Benthos: no data.

- Habitats: no data.

- Marine mammals: no data.

- Birds: no data.

10.7.3 Observed changes – directly adjacent to closed area

- Effort: Increased, displaced from the closure.
- Focal fish species: Probably the closure had a negative impact on the rate of discarding of demersal species due to an increase in trawling activities in areas that are normally not fished.
- Commercial fish species: Probably the closure had a negative impact on the rate of discarding of demersal species due to an increase in trawling activities in areas that are normally not fished.
- Benthos: Probably the closure had a negative impact on the rate of discarding of vulnerable components of the ecosystem (e.g., elasmobranchs or long-lived benthic species) due to an increase in trawling activities in areas that are normally not fished.
- Habitats: no data.
- Marine mammals: no data.
- Birds: no data.

10.7.4 Observed changes – away from closed area

- There are no data to evaluate these criteria.

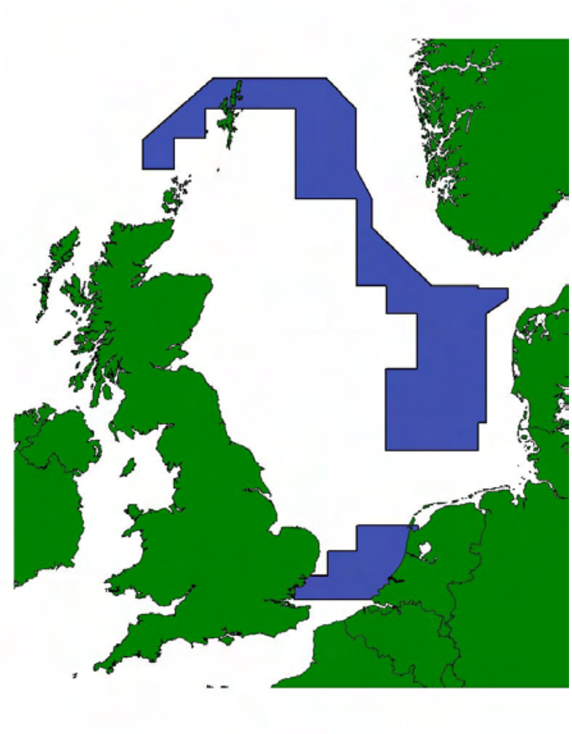


Figure 10.7.1.1. Map of the cod closure area.

10.8 Summary table of criteria for the case studies

Table 10.7.1. General information about six case studies on MPAs in temperate waters.

		Type							Enforcement		
Site	**Purpose**	**Gear/fleet type**	**Abs. size (km^2)**	**Spatial scale (% of area)**	**Time scale**	**Did closure meet objectives**	**Unexpected factors**	**Time for implementation**	**Type**	**Effectiveness**	**Is the closed area part of a network**
Georges Bank	To protect and help restore overfished groundfish resoures	All fishing gears capable of retaining groundfish	CA-I 3,960 CA-II 6,927 NLS 6,275	20–30% of range	Dec. 1994 – present year-round	Yes, for haddock and yellowtail flounder	Wider scale management in US and Canadian waters	Months	US Coast Guard and VMS (from 1998)	Good	Yes, CA I, CA II and NLS = > 17,000 km^2
IPA, Devon	Reduce conflict between static and towed gears	pots, gillnets, dredges, trawls	478,4	73% devoted to static gear all year, 27% seasonally	1978-	Yes (to 90 %)	No	2-3 years	Volunteer, changed to coast guard (legislation in 2002)	Good	No
Öresund	Prevent accidents	towed gear	~1700	NA	1932-	Yes (it prevents accidents)	includes local cod spawning	No data	Coast guard	Bad	No
Southeast Scottish Coast	Conserve sandeels serving as seabird prey	Industrial Fleet using Commercial Sandeel Trawl	21,000	NA	2000-2005	Sandeel abundance increased. Effects on seabirds not clear as yet	Climate/hydrographical factors determining sandeel availability to seabirds	1 year	Monitoring fishery permitted. VMS and SFPA overflight	Good	No
Plaice box	Protect juveniles in nursery areas	All beam trawlers > 300 Hp	no data	NA	1989-1994 seasonal, 1995-present whole year	No, plaice SSB decreased	Juveniles moved out of area due to temp. change	2 yrs	Coast guard and VMS from 2000 onwards	Good	No
Cod box	Protect spawning population of cod	All cod-related fishing fleets	no data	NA	Closed 15/2 – 30/4 2001	No	Bad timing and positioning of closure	Months	Coast guard and VMS	Good	No

Purpose: Reason for closure of the area.
Gear/fleet type: What gears are affected by the closure.
Abs. size: Absolute size of area
Spatial scale: Part of the area covered by closure
Timescale: Time of closure
NA = Not Applicable

Table 10.7.2. Information on changes **within** the closed area of six MPAs in temperate waters.

	Observed changes **within** closed area								
Site	Effort	Focal species or habitat (abundance/ dimensionality)	Commercial fish species (abundance)	Fish community (abundance)	Benthos (abundance)	Habitats (dimensionality)	Marine mammals	Birds	Other
Georges Bank	Decrease	Increase*	Increase*	better*	better	not evaluated	not evaluated	not evaluated	Sea turtles, not evaluated
IPA, Devon	Decrease/ increase	no data	No change (More large individuals)	better	better	better	no data	no data	no
Öresund	Decrease	irrelevant	Increase	no data	no data	no data	no data	no data	no
Southeast Scottish Coast	Decrease	Increase (More larger/older individuals)	Decrease	no data	no data	no data	Possible Increase	Variable depending on species	no
Plaice box	Decrease	Decrease	no change	no change	no change	no data	no data	no data	no
Cod box	Decrease	no change	no change	not evaluated	Not evaluated	no data	no data	no data	no

Effort: Changes in fishing/dredging effort after closure.
Focal species or habitat: Changes of the focal species/habitat after closure.
Commercial species: Changes in abundance of commercial fish species after closure.
Fish community: Changes in the fish community diversity after closure.
Benthos: Changes in the benthic community diversity after closure.
Habitats: Changes of the habitats after closure.
Marine mammals: Changes of abundance of marine mammals after closure.
Birds: Changes in the abundance of birds after closure
* measured as biomass
Not evaluated = data exist but have not been evaluated

Table 10.7.3. Information on changes **adjacent** to the closed area of six MPAs in temperate waters.

	Observed changes – directly **adjacent to** closed area								
Site	Effort	Focal species or habitat (abundance/ dimensionality)	Commercial fish species (abundance)	Fish community (abundance)	Benthos (abundance)	Habitats (dimensionality)	Marine mammals	Birds	Other
Georges Bank	Increase	Increase	not evaluated	not evaluated	no change	no data	no data	no data	Sea turtles, not evaluated
IPA, Devon	no data	no data	no data. (Smaller individuals though).	no data	worse	worse	no data	no data	no
Öresund (Kattegat)	no data	irrelevant	no data	no data	no data	no data	no data	no data	no
Southeast Scottish Coast	Increase	Increased (sandeel)	not evaluated	no data	no data	no data	no data	no data	no
Plaice box	Increase (due to displacement)	Increase	no change	not evaluated	Not evaluated	no data	no data	no data	no
Cod box	Increase (due to displacement)	Decrease	Decrease	not evaluated	worse	no data	no data	no data	no

Effort: Changes in fishing/dredging effort after closure as an effect of the closure.
Focal species or habitat: Changes of the focal species/habitat after closure.
Commercial species: Changes in abundance of commercial fish species after closure.
Fish community: Changes in the fish community diversity after closure.
Benthos: Changes in the benthic community diversity after closure.
Habitats: Changes of the habitats after closure.
Marine mammals: Changes of abundance of marine mammals after closure.
Birds: Changes in the abundance of birds after closure
* measured as biomass
Not evaluated = data exist but have not been evaluated

Table 10.7.4. Information on changes further **away from** the closed area of six MPAs in temperate waters.

	Observed changes - **away** from closed area								
Site	Effort	Focal species or habitat (abundance/dimensionality)	Commercial fish species (abundance)	Fish community (abundance)	Benthos (abundance)	Habitats (dimensionality)	Marine mammals	Birds	Other
Georges Bank	no data	no data	no data	no data	no data	no data	no data	no data	no
IPA, Devon	Not evaluated	Not evaluated	Not evaluated	no data	no data	no data	no data	no data	no
Öresund	no data	no data	no data	no data	no data	no data	no data	no data	no
Southeast Scottish Coast	no change	no data	no data	no data	no data	no data	no data	no data	no
Plaice box	no data	Not evaluated	Not evaluated	Not evaluated	Not evaluated	no data	no data	no data	no
Cod box	no data	Not evaluated	Not evaluated	Not evaluated	Not evaluated	no data	no data	no data	no

Effort: Changes in fishing/dredging effort after closure as an effect of the closure.
Focal species or habitat: Changes of the focal species/habitat after closure.
Commercial species: Changes in abundance of commercial fish species after closure.
Fish community: Changes in the fish community diversity after closure.
Benthos: Changes in the benthic communities after closure.
Habitats: Changes of the habitats after closure.
Marine mammals: Changes of abundance of marine mammals after closure.
Birds: Changes in the abundance of birds after closure
* measured as biomass
Not evaluated = data exist but have not been evaluated

10.9 Evaluation of case studies with reference to the geographical position / scale of the areas and of meta-population dynamics

In the time allowed for this ToR, we have looked at a small number of closed areas, and have tried to select a cross-section that reflects different temperate areas and conditions. In doing so, we recognise that the examples used are the minimum required for a full meta-analysis. Nevertheless, we have used a comprehensive set of evaluation criteria and identified some common themes that appear to have wider application.

We deliberately chose to describe the observed changes in ecosystem components within and away from the boundaries of the closed area. The expected spill-over effects of such areas are well known, yet it is apparent from our initial analysis that fewer data are available with increasing distance from the site to evaluate this. The exception is the Georges Bank closure (Murawski *et al.,* 2000, in press). Density-related spill-over is characterized by a gradient in biomass or abundance from the boundaries of the closed area that declines over distance away from the area. This results in negative density-distance relationships, which were found for haddock for CA-I, yellowtail flounder and winter flounder for CA-II and yellowtail flounder for the NLS area (Murawski *et al*., in press). The authors were not able to attribute the enhanced occurrence of fish (specifically haddock) to spill-over effects alone, because the results were confounded by seasonal migrations and optimal habitat preferences. Survival of recruits (indexed as recruits per unit of spawning stock biomass) improved from the mid-1990s on and for cod and haddock 1998 recruitment survival was above the long-term mean (Murawski *et al*., 2000).

In most cases there appears not to have been any targeted data collection of any ecosystem components outside the area. The documented example where the condition of benthic communities and habitats deteriorated outside the boundaries of a closed area was at the inshore potting area in Devon, UK, where the exclusion of mobile gears resulted in impacts along the boundary. This sampling design did not provide evidence of augmented populations of target stocks. Sampling strategies for any of the ecosystem components at distance from the closed areas were absent from nearly all examples chosen in this analysis.

The exception to this is the Georges Bank closure, which is extensive and well documented. The differences between the success of each of the individual areas can be mostly attributed to the placement of the closed area in relation to the seasonal use of that area by the fish stock in question. The significant increase in scallop biomass seen in the closed area was not reflected outside (Murawski *et al*., 2000), highlighting the importance of effects on fishing on this species.

The duration of closure will need to consider management objectives and the dynamics of the fishery, but it is now recognized that temporary area closures lead to effort displacement if they are not accompanied by catch or effort controls (Rijnsdorp *et al*., 2001). This was also seen in Georges Bank, where the closures have caused allocation changes among gear types and fleet sectors, whereby the large trawlers and other mobile gear fleets excluded from the closed areas fished further away and caused higher mortality on some species than would otherwise have occurred (Murawski *et al*., 2000). Moreover, there was more competition with more localized fisheries elsewhere. The duration of closure in all the examples selected in Table 10.7.1 was at least four years, and one (Øresund), extended back to 1932, but no recurrent seasonal closures were examined. In an analysis of response of the North Sea beam trawl fleet to the closure of the “cod box”, Dinmore *et al*. (2003) suggested that repeated seasonal area closures would lead to a more homogeneous distribution of annual trawling activity, which was thought to have slightly greater cumulative impacts on total benthic invertebrate production and led to localized reductions in benthic biomass for several years. They recommended that, under such circumstances, effort reductions or permanent area closures should be considered, leading to a single but permanent redistribution of fishing disturbance (Dinmore *et al*., 2003). In that context Murawski *et al.* (2000) concluded that year-round large closed areas have been easier to enforce than seasonal small closures.

At least two cases that did not succeed in meeting their objectives support the contention that science should be used to the fullest extent possible in the establishment and monitoring of closed areas. The example of the plaice box shows that environmental factors that were not incorporated in the decision-making beforehand, turned out to be very important in determining the subsequent effect of the closure. For the closure of the cod box, it can only be concluded that scientific information that was available at the time was apparently not used in the decision-making. The plaice box was designed and implemented in collaboration with the fishermen, which is in principle an advantage in ensuring stakeholder acceptance of the process. The failure of this closure to meet its objectives, however, has affected relationships between science and the fishermen to date.

10.10 Review of published guidelines for the establishment of Marine Protected Areas

The development of meaningful guidelines for the establishment of closed areas is an important process to guide managers and scientists in site selection. The following section deals with the final part of the ToR, to 'review published guidelines for the establishment of marine protected areas and recommend revisions'. In addressing this part of the ToR, we have compared well-known guidelines from OSPAR, IUCN, and WWF.

10.10.1 Guidelines for the Identification and Selection of Marine Protected Areas in the OSPAR Maritime Area

The OSPAR Commission has agreed to promote the establishment of a network of marine protected areas (MPAs) to ensure the sustainable use, protection, and conservation of marine biological diversity and ecosystems, as part of the Annex V 'On the Protection and Conservation of the Ecosystems and Biological Diversity of the Maritime Area'. The objective of the Commission is to take the necessary measures to protect and conserve the ecosystems and the biological diversity of the maritime area which are, or could be, affected as a result of human activities, and to restore where practicable marine areas which have been adversely affected (OSPAR BDC 03/10/1).

The aim of OSPAR MPAs is to:

- protect, conserve and restore species, habitats and ecological processes which are adversely affected as a result of human activities;
- prevent degradation of and damage to species, habitats and ecological processes, following the precautionary approach;
- protect and conserve areas that best represent the range of species, habitats and ecological processes in the OSPAR area.

The purpose of OSPAR MPAs is also that they should form an ecologically coherent network of well-managed MPAs. This is considered particularly important for highly mobile species, such as certain birds, mammals and fish, to safeguard the critical stages and areas of their life cycle (such as breeding, nursery and feeding areas).

OSPAR has identified a two-stage process for the identification and selection of MPAs, focusing on the identification and prioritisation of sites.

10.10.1.1 Stage 1 – the identification of possible sites

In the first stage of the MPA selection criteria, seven ecological criteria/considerations are to be applied:

(1) Threatened or declining species and habitats/biotopes.

The area is important for species, habitats/biotopes and ecological processes that appear to be under immediate threat or subject to rapid decline as identified by the ongoing OSPAR (Texel-Faial) selection process.

(2) Important species and habitats/biotopes.

The area is important for other species and habitats/biotopes as identified by the ongoing OSPAR (Texel-Faial) selection process.

(3) Ecological significance

The area has:

- a high proportion of a habitat/biotope type or a biogeographic population of a species at any stage in its life cycle;
- important feeding, breeding, moulting, wintering or resting areas;
- important nursery, juvenile or spawning areas, or
- a high natural biological productivity of the species or features being represented.

(4) High natural biological diversity

The area has a naturally high variety of species (in comparison to similar habitat/biotope features elsewhere) or includes a wide variety of habitats/biotopes (in comparison to similar habitat/biotope complexes elsewhere).

(5) Representativity

The area contains a number of habitat/biotope types, habitat/biotope complexes, species, ecological processes or other natural characteristics that are representative for the OSPAR Area as a whole or for its different biogeographic regions.

(6) Sensitivity

The area contains a high proportion of very sensitive or sensitive habitats/biotopes or species.

(7) Naturalness

The area has a high degree of naturalness, with species and habitats/biotope types still in a very natural state as a result of the lack of human-induced disturbance or degradation.

There is some guidance provided by OSPAR on how to use these criteria. The Guidance states that an area qualifies for selection as an MPA if it meets several but not necessarily all of these criteria. The consideration and assessment of these criteria should be based on best available scientific expertise and knowledge.

It is inevitable that this process will identify a number of sites which meet some of the criteria. For this reason a second stage has been included which allows further prioritisation.

10.10.1.2 Stage 2 - prioritisation of sites for designation

In this second stage of the process, it is recommended by OSPAR that the ecological criteria/considerations applied in stage 1 should be reapplied to help prioritise the identified sites. For example, an area that holds a higher population of the species concerned or that meets additional ecological criteria may warrant a higher priority. In addition, the following six practical criteria/considerations should be taken into account in developing a prioritised list of sites. For instance an area with a comparatively higher level of support from stakeholders and political acceptability will be more suitable to be established as an MPA.

(1) Size

The size of the area should be suitable for the particular aim of designating the area, including maintaining its integrity, and should enable the effective management of that area.

(2) Potential for restoration

The area has a high potential to return to a more natural state under appropriate management.

(3) Degree of acceptance

The establishment of the MPA has a comparatively high potential level of support from stakeholders and political acceptability.

(4) Potential for success of management measures

There is a high probability that management measures and the ability to implement them (such as legislation, relevant authorities, funding, and scientific knowledge) will meet the aims for designation.

(5) Potential damage to the area by human activities

It is an area where significant damage by human activity may happen in the short term.

(6) Scientific value

The area has a high value for scientific research and monitoring.

10.10.1.3 Use of the criteria to meet the aims of OSPAR MPAs

Concluding guidance relates each of these sets of criteria (ecological – stage 1, practical – stage 2) to the objectives of the OSPAR MPA strategy. The following table is provided by OSPAR as guidance on using the criteria to select MPA which achieve the major aims of the MPA Strategy.

Aims of OSPAR MPAs	**Protect, conserve and restore species, habitats and ecological processes which are adversely affected as a result of human activities**	**Prevent degradation of and damage to species, habitats and ecological processes following the precautionary approach**	**Protect and conserve areas which best represent the range of species, habitats and ecological processes in the OSPAR area**
Ecological considerations	(A1) High priority habitats and species which meet the Texel-Faial criteria of 'Decline'	(A1) High priority habitats and species which meet the Texel-Faial criteria of 'high probability of a significant decline' (A2) Important habitats & species which meet the other Faial criteria (global importance, local (species)/regional (habitats) importance, rarity, sensitivity, keystone species, ecological significance) (A6) Sensitivity	(A3) Ecological significance (A4) High natural biological diversity (of species within a habitat and of habitats in an area) (A5) Representativity, including of the biogeographic regions (A7) Naturalness
Practical considerations	(B1) Size (B2) Potential for restoration (B3) Degree of acceptance (B4) Potential for success of management measures (B6) Scientific value	(B1) Size (B3) Degree of acceptance (B4) Potential for success of management measures (B6) Scientific value (B5) Potential damage to the area by human activities	(B1) Size (B3) Degree of acceptance (B4) Potential for success of management measures (B6) Scientific value

Note: Numbers in brackets refer to the specific criteria in the Guidelines for the Identification and Selection of MPAs in the OSPAR Maritime Area.

10.10.1.4 Guidelines for management of OSPAR MPAs

OSPAR sees management plans as valuable tools to help achieve the objectives of their MPAs. There is guidance for the development of a management plan, including an outline structure, based on that of IUCN (OSPAR, 2003). Special note is made of the identification and regulation of important human activities and how international and European Community legislation may assist with the implementation of management measures. The latter are listed in the OSPAR legal study (OSPAR 02/2/4). National legislation will be required to support management of OSPAR MPAs within EEZs. The effectiveness of the management measures will need to be evaluated and the management plan will need to be adapted as necessary and appropriate on a regular basis. The management plan should be developed with the active involvement of relevant stakeholders from the earliest stages.

10.10.2 Guidelines for the Identification and Selection of Marine Protected Areas by the IUCN.

IUCN has been active in the development of Marine Protected Areas (MPAs) since the 1970s and a number of important documents have been produced by the Union, most notably the 1991 "Guidelines for Establishing Marine Protected Areas", the "Guidelines for Protected Area Management Categories (IUCN, 1994) and in 1995 "A Global

Representative System of Marine Protected Areas". In 1999 new guidelines were published, replacing those written in 1991, and based on the advances made in the intervening period (Kelleher, 1999).

Six categories were defined in *Guidelines for Protected Area Management Categories* (IUCN 1994), according to different management objectives These are also relevant to Marine Protected Areas;

- I Strict protection (i.e., Strict Nature Reserve / Wilderness Area)
- II Ecosystem conservation and recreation (i.e., National Park)
- III Conservation of natural features (i.e., Natural Monument)
- IV Conservation through active management (i.e., Habitat/Species Management Area)
- V Landscape/seascape conservation and recreation (i.e., Protected Landscape/Seascape)
- VI Sustainable use of natural ecosystems (i.e., Managed Resource Protected Area)

IUCN (Kelleher, 1999) defines a Marine Protected Area (MPA) as:

> "Any area of intertidal or subtidal terrain, together with its overlying water and associated flora, fauna, historical and cultural features, which has been reserved by law or other effective means to protect part or all of the enclosed environment" (IUCN, 1994).

The goal of MPAs according to IUCN is to "conserve the biological diversity and productivity (including ecological life support systems) of the oceans." IUCN has also defined a goal for a global network of MPAs (Kelleher, 1999):

> "To provide for the protection, restorations, wise use, understanding and enjoyment of the marine heritage of the world in perpetuity through the creation of a global, representative system of marine protected areas and through the management in accordance with the principles of the World Conservation Strategy of human activities that use or affect the marine environment."

10.10.2.1 Steps in the IUCN guidelines for identification and selection of marine protected areas

The 1999 IUCN guidelines set out the following steps (IUCN, 1999):

1. Placing MPAs in their wider context.
MPAs should be integrated with other policies for land use and use of the sea. It is also desirable for countries to make use of international agreements, notably UNCLOS and CBD.

2. Developing the legal framework.
In most countries, a key step will be to establish the legislation needed. This may either be enabling legislation, which allows the administration or communities to establish individual MPAs, or specific legislation establishing an MPA, usually as a large multiple-use area.

3. Working with relevant sectors.
Many sectors of human activity affect the coast and the sea, and it is vital for those planning an MPA to work with these sectors from the earliest opportunity. Tourism often has most to gain from an MPA and can generate the greatest economic activity from it. Fisheries is the other key sector, and one with which it is most important to cooperate. Other relevant sectors include aquaculture, coastal development, agriculture, forestry, industry, defence and science.

4. Making partnerships with communities and other stakeholders.
MPA management should understand the local communities that will be affected by the MPA and identify potential partners. It must listen to the many interests and seek ways to involve them as participants in resource management

5. Selecting the sites for MPAs.
The dependence on marine areas tends to be even greater than on terrestrial areas. Some forms of fishing can occur in large areas without threatening the conservation objectives of the MPA because they do not involve habitat modification. This makes it feasible to balance conservation and the needs of local people. Weight needs to be given to events outside the MPA that might affect it, such as pollution.

6. Planning and managing the MPA.
Management should be responsive and adaptive, working with local interests in a way that builds support for the conservation objectives. Most MPA management is about managing human activities, so this must be at the heart of the approach.

7. Zoning, in which various areas are allocated for various uses.
This is usually the best way of ensuring strict protection of a core zone as part of a larger, multiple-use area.

8. Planning for financial sustainability. Lack of funds is a critical problem for many MPAs. Managers therefore need the freedom to raise funds in as many ways as possible, such as user fees, donations and environment funds, and to retain those funds for management of the MPA. External donors are advised to extend the aid period for protected area projects, so as to help achieve financial sustainability.

9. Ensuring research, monitoring, evaluation and review
Research and monitoring should be firmly orientated to solving management issues. Guidance is given on the planning and development of a monitoring and research programme, with its different emphases in the planning and the implementation phase of the MPA. Most important of all is to use the results of research and monitoring to evaluate and if necessary reorient management.

10.10.2.2 IUCN criteria for the selection of MPAs

The criteria (Step 5) for selecting MPAs cover a wide range of issues, as seen below.

Biogeographic criteria	Presence of rare biogeographic qualities or representative of a biogeographic "type" or types Existence of unique or unusual geological features
Ecological criteria	Ecological processes or life-support systems (e.g. as a source for larvae for downstream areas) Integrity, or the degree to which the area, either alone or in association with other protected areas, encompasses a complete ecosystem The variety of habitats Presence of habitat for rare or endangered species Presence of nursery or juvenile areas Presence of feeding, breeding or rest areas Existence of rare or unique habitat for any species Degree of genetic diversity within species
Naturalness	Extent to which the area has been protected from, or has not been subject to, human-induced change
Economic importance	Existing or potential economic contribution due to protection (e.g. protection of an area for recreation, subsistence, use by traditional inhabitants, appreciation by tourists and others, or as a refuge nursery area or source of economically important species)
Social importance	Existing or potential value to local, national or international communities because of its heritage, historical, cultural, traditional, aesthetic, educational or recreational qualities
Scientific importance	Value for research and monitoring
International or national significance	Existence of any national or international designation Potential for listing on a national or international system
Practicality or feasibility	Degree of insulation from external destructive influences Social and political acceptability, degree of community support Accessibility for education, tourism, recreation Compatibility with existing uses, particularly by locals Ease of management or compatibility with existing management regimes
Duality or Replication	MPAs, particularly when small, can be subject to devastating destructive influences, either from humans or from nature, such as cyclones on coral reefs. It is therefore desirable that there should be more than one sample of every major ecosystem type in a representative system.

10.10.2.3 IUCN Research and Performance indicators

Research and monitoring within the IUCN framework is suggested as being directed at pressures (natural and human induced), state (biota, processes and ecological) and response of management. Moreover the implementation of measures, compliance, and meeting of objectives should also be monitored.

IUCN is quite clear on the value of performance indicators (Kelleher, 1999).

"The step to reconsider the management programme on the basis of the results of monitoring has been omitted or performed superficially in most MPAs. Yet, if MPAs are to be ecologically and socially sustainable, almost continuous evaluation and learning is essential. Evaluation must address two broad questions:

a) What has been accomplished by the MPA and learned from its successes and failures?
b) How has the context (e.g. environment, governance) changed since the programme was initiated?

The answers to these questions can be used to re-focus management in future. A meaningful evaluation can be conducted only if the MPA objectives were stated in clear terms and if **indicators** for assessing progress were identified in the planning phase, and monitored afterwards. Baseline data are essential. Many evaluations yield ambiguous results because these preconditions for assessing performance do not exist."

Extensive examples are given in the Guidelines on the type of research and monitoring necessary to evaluate the success of an MPA (Table 10.5.2.3.1)

Table 10.5.2.3.1 Examples of research and monitoring for evaluating MPAs (GESAMP, 1996, in Kelleher, 1999).

Topic	Examples of research	Examples of monitoring
Pollution Contaminant inputs	Identify **major sources** (industry, agriculture, fisheries, sewage, shipping etc.) and **pathways** (pipes/sewers, rivers, atmosphere, discards from ships etc.); developing suitable **sampling** and **analytical** methods	Quantify **loads** of priority contaminants (e.g. heavy metals, nutrients, organochlorines, TBT, oil, faecal coliform bacteria)
Fishery management Stock depletion – causes and solutions	Investigate life-cycles, reproductive features, feeding requirements and habitats of affected species; identify factors (climatic, trophic, human etc.) controlling **inter-annual variations** in these characteristics; determine local factors limiting **recruitment**, such as fishing methods and intensity, predation, disease, poor water quality, reduced spawning habitat etc.	Implement a schedule of measurements to obtain **more reliable data on temporal variations** in key parameters as identified from prior research (e.g. numbers and age-classes of fish or shellfish harvested by different methods, availability of prey species, variations in water and prey quality, rates of habitat loss, incidence of disease)
Conservation of habitats and biodiversity Impacts of development/use of coastal areas and resources	Identify, classify and map **remaining natural (undeveloped) habitats** and compare with any historical records; characterize associated biotic communities and exploitable living resources; evaluate their **inter-dependencies, ecological importance** and **sensitivities** to human activities; identify factors that may determine habitat **sustainability** and appropriate measurable indicators of these factors; quantify relative extents of modified habitats and areas reclaimed for housing, industry, agriculture, aquaculture, forestry, tourism and recreation, transport, harbours and marinas; develop an interactive, computerized database to hold all such records.	Implement a long-term programme to quantify physical, biological and **ecological changes** in habitats with a particular focus on more **sensitive species**, communities and processes; develop indicators of long-term **sustainability** derived from prior research; maintain up-to-date records on rates of physical development and changes in patterns and intensities of human activities; record changes in demography, tourist numbers, aquaculture, fishery production, port traffic, offshore aggregate extraction, sewage and waste generation and other factors that may increase pressures on habitats and resources, or reduce biodiversity.

10.11 Review of published guidelines for the establishment of marine protected areas

10.11.1 Published examples

There are a number of international activities that aim to evaluate the effectiveness of MPAs. Most recent is the IUCN Management Effectiveness Initiative to developed new, practical methods for assessing the effectiveness of MPAs, and examining their capacity for adaptive management by providing a step-by-step process for planning and evaluating the management effectiveness of MPAs. The results are to be published in an IUCN report during 2004 entitled 'How Is Your MPA Doing? A Guidebook of Natural and Social Indicators for Evaluating Marine Protected Area Management Effectiveness'. http://ipo.nos.noaa.gov/mgmteffect/guidebook.html. This increased international activity is to be encouraged, and WEGCO welcomes the move towards more rigorous scientific assessment of the impact of closed areas on biota. A compilation of views from across a range of published sources, and which were relevant to the WGECO meta-analysis that follows, are:

- Socio-economic considerations usually determine the success or failure of MPAs. In addition to biophysical factors, these considerations should be addressed from the outset in identifying sites for MPAs, and in selecting and managing them;

- It is better to have an MPA which is not ideal in the ecological sense but which meets the primary objective than to strive vainly to create the 'perfect MPA';

- It is usually a mistake to postpone action on the establishment of an MPA because biophysical information is incomplete. There will usually be sufficient information to indicate whether the MPA is justified ecologically and to set reasonable boundaries;

- An MPA must have clearly defined objectives against which its performance is regularly checked, and a monitoring programme to assess management effectiveness. Management should be adaptive, meaning that it is periodically reviewed and revised as dictated by the results of monitoring;

- There is a global debate about the merits of small, highly protected MPAs and large, multiple use MPAs. Much of this debate arises from the misconception that it must be one or the other. In fact, nearly all large, multiple use MPAs encapsulate highly protected zones, which can function in the same way as individual highly protected MPAs. Conversely, a small, highly protected MPA in a larger area subject to integrated management can be as effective as a large, multiple use MPA;

- Because of the highly connected nature of the sea, which efficiently transmits substances and forcing factors, an MPA will rarely succeed unless it is embedded in, or is so large that it constitutes, an integrated ecosystem management regime.

Alder et al., (2002) have developed a method for evaluating MPA management. It is based on a multidimensional scaling model for the rapid appraisal of fisheries (Rapfish) and uses six evaluation fields (living resources, non-living resources, economic, social, ecosystem functions and management), each with 6-10 attributes. The authors report that managers and researchers are positive about the first results. More information about the necessary software, is available at www.fisheries.ubc.ca/Projects/MPAEM.htm

The two reviewed sets of guidelines were set against a number of possible criteria and compared. This exercise preliminary exercise might structure thinking ahead of a more extensive review and a substantiated recommendation of revisions.

Guidelines	**Guidelines for selection of sites**	**Stepwise guide to management plan**	**Guidelines for research and monitoring**	**Performance indicators**
OSPAR	Primarily ecological and practical (size, potential for management success)	Yes	Yes	Not explicit
IUCN	Ecological, but also emphasis on socio-economics	Yes	Yes	Yes

The IUCN Guidelines form a comprehensive framework for the implementation of MPAs. Many specific details are treated in the Guidelines and clear step-by-step guides are given in the Appendices for the development of a co-management partnership, the contents of an MPA Management Plan and how to make a zoning plan. IUCN has explicit involvement of public and possible candidates for co-management at an early stage of the designation process. The selection of MPAs comes at Step 5 of the IUCN Guidelines, whereas, for example, for OSPAR, this step is part of the initial stages in the process, which has two stages for the identification and selection of MPAs, focussing on the identification and prioritisation of sites. However, in the Guidelines for the management of MPAs in the OSPAR Maritime Area (an Annex of the OSPAR report) all the steps described in the IUCN Guidelines are stated as being essential for the success of MPA implementation.

10.11.2 Present and future work on guidelines

The initiation of Marine Protected Areas is a very relevant and increasingly used management tool. There are many other initiatives world-wide for the design and implementation of MPAs each with their own set of Guidelines, which have not been discussed here. It would be timely and relevant to take note of these and to carry out a more explicit review of the available guidelines. ICES itself is holding a Theme Session on the issue of MPAs in 2004.

10.12 Conclusions

10.12.1 Case-studies

In the time allowed for this ToR we have looked at a small number of closed areas, and have tried to select a cross-section that reflects different temperate areas and conditions. We deliberately chose to describe the observed changes in ecosystem components within and away from the boundaries of the closed area. The expected spill-over effects of such areas are well known, yet it is apparent from our initial analysis that fewer data are available with increasing distance from the site to evaluate this. In most cases, there appears not to have been any targeted data collection of any ecosystem components outside the area, although the potential benefits to commercial fish species was assessed outside the areas established at the Øresund and at the IPA, Devon, UK. The exception to the above is the Georges Bank closure, which is extensive and well documented.

The duration of closure will depend on the biology of the target species and the dynamics of the fishery, but it is now recognized that temporary area closures lead to effort displacement if they are not accompanied by catch or effort controls. Under conditions with repeated, seasonal area closures, effort reductions or permanent area closures should be considered, leading to a single but permanent redistribution of fishing disturbance. In that context, Murawski *et al.* (2000) concluded that year-round large closed areas have been easier to enforce than seasonal small closures.

At least two cases that did not succeed in meeting their objectives support the contention that science should be used to the fullest extent possible in the establishment and monitoring of closed areas. The example of the plaice box shows that environmental factors that were not incorporated in the decision-making beforehand, turned out to be very important in determining the effect of the closure. For the closure of the cod box, it can only be concluded that scientific information that was available at the time was apparently not used in the decision-making. The plaice box was designed and implemented in collaboration with the fishermen, which is in principle an advantage in ensuring stakeholder acceptance of the process. The failure of this closure to meet its objectives, however, has affected relationships between science and the fishermen to date.

The analysis of the case-studies has provided valuable information for addressing the term of reference, but two issues were only briefly discussed in this section: geographical position/scale and meta-population dynamics. These are issues that must be addressed in order to be able to make well-founded decisions about the implementation of MPAs on fish and shellfish stocks. Expanding on the work that was done at this meeting by taking a wider range of case studies and a greater representation with regard to geography and scale, and incorporating the knowledge of meta-population dynamics, will provide the background for ICES (WGECO) to advise on MPAs in the future.

10.12.2 Guidelines

The IUCN Guidelines form a comprehensive framework for the implementation of MPAs, including guides for the development of a co-management partnership, the contents of an MPA Management Plan, and how to make a zoning plan. They have explicit involvement of public and possible candidates for co-management at an early stage of the designation process and, in consequence, the selection of MPAs comes at Step 5 of the Guidelines. In contrast, the OSPAR guidance has this stage as part of the initial part of the process, focusing on the identification and prioritisation of sites. A re-evaluation of these criteria in OSPAR to bring them into line with those of the international community could bring a range of benefits.

The group carried out an exercise to relate the two sets of selection criteria to the case-studies. Even though the case-studies were not set up as fully fledged MPAs, the exercise served to highlight the freedom given in the selection criteria and the difficulty in quantification of the conservation objectives. Moreover, there is a wide range of issues that need to be addressed for which ICES is not ideally in a position to advise (economic, social, cultural).

ICES is well placed to advise on the science and, as our case studies illustrated, quality science is important in ensuring that MPAs are able to meet their objectives. ICES is, therefore, competent to contribute to the scientific elements of any framework for the establishment of MPA, however, the development of guidelines must also include appropriate social cultural and economic considerations and these are beyond ICES expertise.

10.13 References

Blyth, R.E. Kaiser, M.J. Edwards-Jones, G. and Hart, P.J.B. 2002. Voluntary management in an inshore fishery has conservation benefits. Environmental Conservation, 29: 493-508.

Dinmore, T. A., Duplisea, D. E., Rackham, B. D., Maxwell, D. L., and Jennings, S. 2003. Impact of a large-scale area closure on patterns of .fishing disturbance and the consequences for benthic communities. ICES Journal of Marine Science, 60: 371–380.

ICES. 1999. Report of the Study Group on Effects of Sandeel Fishing. ICES CM 1999/ACFM:19.

IUCN. 1994. Guidelines for Protected Area Management Categories. CNPPA with the assistance of WCMC. IUCN, Gland, Switzerland, and Cambridge, UK. x + 261 pp.]

Jensen H.; Rindorf A.; Horsten M.B.; Mosegaard H.; Brogaard P.; Lewy P.; Wright P.J.; Kennedy F.M.; Gibb I.M.; Ruxton G.; Arnott S.A. and Leth J.O. 2001. Modelling the population dynamics of sandeel (Ammodytes marinus) populations in the North Sea on a spatial resolved level. DG XIV no. 98/025.

Kaiser, M.J. Spence, F.E. and Hart P.J.B. 2000. Fishing gear restrictions and conservation of benthic habitat complexity. Conservation Biology, 14(5): 1512-1525.

Kelleher, G. (1999). Guidelines for Marine Protected Areas. IUCN, Gland, Switzerland and Cambridge, UK. xxiv +107pp.

Kunzlik, P.A. Gauld, J.A. and Hutcheon, J. R. (1986). Preliminary results of the Scottish sandeel tagging project. ICES CM 1986/G:7.

Murawski, S.A., Brown, R. Lai, H.-L., Rago, P. and Hendrickson, L. 2000. Large-scale closed area as a fishery-management tool in temperate marine systems: the Georges Bank experience. Bulletin of Marine Science 66(3):775-798.

Murawski, S.A., Rago, P. and Fogarty, M. in press. Spillover effects from temperate marine protected areas. American Fisheries Society Symposium, in press.

Pastoors, M. A., A. D. Rijnsdorp, *et al.* (2000). Effects of a partially closed area in the North Sea (“plaice box”) on stock development of plaice. ICES Journal of Marine Science, 57: 1014-1022.

Pedersen, S.A., Lewy, P. and Wright, P., 1999. Assessment of the lesser sandeel (Ammodytes marinus) in the North Sea based on revised stock divisions. Fisheries Research, 41: 221-241. Pedersen, S.A., Lewy, P. and Wright, P., 1999. Assessment of the lesser sandeel (Ammodytes marinus) in the North Sea based on revised stock divisions. Fisheries Research, 41: 221-241.

Piet, G. J. and A. D. Rijnsdorp (1998). "Changes in the demersal fish assemblage in the south-eastern North Sea following the establishment of a protected area ("plaice box")." Ices Journal of Marine Science, 55(3): 420-429.

Proctor, R., Wright, P.J. and Everitt, A. (1998). Modelling the transport of larval sandeels on the north west European shelf. Fisheries Oceanography.7, 347-354.

Rago, P. and McSherry, 2001. Spatial distribution of fishing effort for sea scallops: 1998-2000. Appendix G. Prepared for Effects of Fishing Gear on Fish Habitat in the Northeastern U.S. 6 pp.

Rijnsdorp, A. D., Piet, G.J. and Poos, J.J. (2001). Effort allocation of the Dutch beam trawl fleet in response to a temporarily closed area in the North Sea. ICES CM 2001/N:01

Smit, W., Davidse, W. P., De Jager, J., De Ruijter, C., Smit, M. H., Taal, C., and Van Wijk, M. O. 1997. Visserij in cijfers 1997. LEI PR No. 31-97 (in Dutch).

Svedäng, H. Svensson, A. and Hagberg, J. Unpublished. Differences in technical fishing regulations between the Sound and Kattegat: marked effects on abundance and size distribution of dominant demersal fish species. Henrik Svedäng, Institute of Marine Research, Turistgatan 5, 453 30 Lysekil, Sweden.

Van Beek, F. A., De Wilde, J. W., Dol, W., and Blom, W. C. 1998. Creation of database of fishery statistics of the Dutch fleet (DAFIST). RIVO-DLO Report C014/98.

Wanless, S., Daunt, F., Wilson, L., Hamer, K., Humphreys, L., Hennicke, J., and Garthe, S. 2002. Seabird foraging performance and diet (colony based studies) summer 2002. Report 2002-006 of the IMPRESS Project, 23pp.

Wright P., Verspoor E., Andersen C., Donald L., Kennedy F., Mitchell A., Munk P., Pedersen S.A., Jensen H., Gislason H. and Lewy P. 1998. Population structure in the lesser sandeel (Ammodytes marinus) and its implications for fishery-predator interactions. DG XIV no. 94/071.

Wright, P.J., Jensen, H., Mosegaard, H., Dalskov, J., and Wanless, S., 2002. European Commission's annual report on the impact of the Northeast sandeel fishery closure and status report on the monitoring fishery in 2000 and 2001.

11 MEASURES OF HABITAT QUALITY

Terms of Reference: i) Consider the existing frameworks for assessing the role of habitats in support of biological diversity and the provision of ‘essential’ habitat elements for key life stages and review any existing measures of ‘habitat quality’. Based on these analyses consider how this EcoQO element can be advanced.

11.1 Existing international frameworks to assess the role of habitats in supporting biological diversity

The biodiversity of communities in natural habitats is reliant upon the physical, chemical and biological characteristics of the area. Natural habitats are defined here as the ‘terrestrial or aquatic areas distinguished by geographic, abiotic and biotic features, entirely natural or semi-natural’ (EC, 1992). Assessing the role of habitats in supporting biological diversity will rely on the amount and the scale of the disturbance the habitat encounters, whether from natural or anthropogenic causes.

As part of its obligations towards the United Nations Convention on Biological Diversity (1992), the European Community has adopted a biodiversity strategy which aims to:

“…reverse present trends in biodiversity reduction or losses and to place species and ecosystems… at a satisfactory conservation status, both within and beyond the territory of the Union” (EC, 2001).

The approach developed by the European Community Biodiversity Strategy recognises that protection of key species and habitats is essential to reduce biodiversity loss, and suggests that more than 10–20% of the world's territory could potentially be designated as a protected area (EC, 2001). However, it recognises that this is a limited means of protecting biodiversity and is unlikely to result in a satisfactory response to the problem of biodiversity loss.

The European Biodiversity Action Plan (BAP) on Fisheries aims to preserve or rehabilitate biodiversity where it is under threat due to fisheries and aquaculture activities. One of the measures established under this BAP is the protection of non-target, species, habitats, and ecosystems from fishing activities. The BAP on the Conservation of Natural Resources is a non-statutory mechanism that aims to provide legal protection of flora and fauna and/or the places where they occur and aims to bring habitats and species of Community interest to a satisfactory conservation status by fully implementing the Birds and Habitats Directives and by providing adequate financial and technical support for the conservation and sustainable use of areas designated under this legislation.

UK Biodiversity Action Plan

In the UK, the Lead Partners reported 18 (42%) habitats and 53 (15%) species associated with coastal and marine habitats. The factors causing loss or decline of habitats identified coastal and marine-related issues as important for the conservation of 32 habitats and 72 species (UKBAP, 2002). Lead Partners frequently identify impacts associated with climate change / sea level rise (28% of factors), coastal defence works (15%), damage from bottom-trawling fishing gear, by-catch of marine species, coastal erosion and overfishing of stocks. These factors account for 66% of all marine and coastal related threats. Other factors include pollution from commercial and domestic sources, dredging and fish farming.

11.2 Existing international frameworks for the provision of ‘essential’ habitat elements for key life stages

Both the U.S. and Canada have legislation which requires the identification of “essential” (U.S.) or “critical” (Canada) habitat for species, as a precondition for providing protection to such habitats. In the U.S., the legislation is the Magnuson-Stevens Act (1996), which requires specification of essential fish habitat (EFH in U.S. literature) for all managed fish species. The U.S. Endangered Species Act (1973) also includes habitat provisions; specifically Critical Habitat. This is defined as: habitat necessary to avoid extinction, the bare minimum or core habitat. WGECO choose to deal with the “essential habitat” designations under the Magnuson- Stevens Act, and not the more restrictive ESA concept of “critical” habitat, because the former was more directly related to marine issues.

In Canada, the Fisheries Act has provisions to protect fish habitat, and a no-net-loss policy for fish habitat has been in place since 1986. However, it was the Species at Risk Act (2002) which included a legal requirement that “critical” habitat for species designated as “threatened” or “endangered” under the Act be identified and protected from damage or disturbance.

In Europe, protected habitats are listed under Annex I of the Habitat and Species Directive (EC, 1992) and the species whose habitats require protection are listed under Annex II. The Water Framework Directive (EC, 2000) assesses the quality of water bodies and regulates to improve their status. Specific objectives of the Directive are to prevent further deterioration and protect and enhance the status of the aquatic ecosystems.

In response to the legislation, the U.S. has a developed a formal set of guidelines for identification of EFH (NOAA website). In Canada, science-based workshops are in the process of developing a framework of determining critical habitat, and guidelines for its application. These are detailed in Appendix 11.1 and 11.2.

In Europe, protection of the species and habitats protected under the Habitats and Species Directive (1992) has led to the development of protected areas. Currently, most implementation of the Habitats and Birds Directives is on shore and within territorial waters. Further work is underway to implement the Directives out to the limits of the EEZ or continental shelf extensions.

A comparison of the US Magnuson-Stevens Act (1996), the Canadian Species at Risk Act (2002) and the EU Habitats and Species Directive (1992) is shown in Table 11.1.

Essential Habitats: Atlantic cod

Several sources document the importance of gravel/cobble substrate to the survival of newly settled juvenile cod (Lough *et al*., 1989; Gotceitas and Brown, 1993; Tupper and Boutilier, 1995). A substrate of gravel or cobble allows sufficient space for newly settled juvenile cod to find shelter and avoid predation (Lough *et al*., 1989; Valentine and Lough, 1991; Gotceitas and Brown, 1993; Tupper and Boutilier, 1995; Valentine and Schmuck, 1995). Particular life history stages or transitions are sometimes considered "ecological bottlenecks" if there are extremely high levels of mortality associated with the life history stage or transition. Extremely high mortality rates attendant to post-settlement juvenile cod are attributed to high levels of predation (Tupper and Boutilier ,1995).

Increasing the availability of suitable habitat for post-settlement juvenile cod could ease the bottleneck, increasing juvenile survivorship and recruitment into the fishery. For these reasons, areas with a gravel/cobble substrate meet the first criterion for habitat areas of particular concern. Specific areas on the northern edge of Georges Bank have been extensively studied and identified as important areas for the survival of juvenile cod (Lough *et al*., 1989; Valentine and Lough, 1991; Valentine and Schmuc,k 1995). These studies provide reliable information on the location of the areas most important to juvenile cod and the type of substrate found in those areas. These areas have also been studied to determine the effects of bottom fishing on the benthic megafauna (Collie *et al*., 1997). Gravel/cobble substrates not subject to fishing pressure support thick colonies of emergent epifauna, but bottom fishing, especially scallop dredging, reduces habitat complexity and removes much of the emergent epifauna (Collie *et al*., 1997).

Acknowledging that a single tow of a dredge across pristine habitat will have few long-term effects, Collie *et al.* (1997) focus on the cumulative effects and intensity of trawling and dredging as responsible for potential long-term changes in benthic communities. For these reasons, the identified area on the northern edge of Georges Bank meets the second criterion, as well as the cumulative effects consideration, for designation as a habitat area of particular concern. Collie *et al.* (1997) also describe the relative abundance of several other species such as shrimps, polychaetes, brittle stars, and mussels in the undisturbed sites. These species are found in association with the emergent epifauna (bryozoans, hydroids, worm tubes) prevalent in the undisturbed areas.

Several studies of the food habits of juvenile cod identify these associated species as important prey items (Hacunda, 1981; Lilly and Parsons, 1991; Witman and Sebens, 1992). These areas provide two important ecological functions for post-settlement juvenile cod relative to other areas: increased survivability and readily available prey. These areas are also particularly vulnerable to adverse impacts from mobile fishing gear.

http://www.nero.noaa.gov/ro/doc/sec_3.pdf

Table 11.2.1. A comparison of U.S., Canadian and European habitat legislation.

	Europe: Habitat and Species Directive (1992)	**U.S.: Magnuson-Stevens Act (1996)**	**Canada: Species at Risk Act (2003)**
How are habitats selected?	Decided by political process but generally: Annex I: Habitats tend to be rare and/or endangered. Annex II: Habitats support species considered important for conservation reasons	For every species for which a Fisheries Management Plan is developed, essential habitat must be specified (by location and on maps) in the FMP (See Annex 11.1). "Identifications must be based on the best available science regarding the habitat requirements of each managed species"	For every species listed as Threatened, Endangered, or Extirpated critical habitat must be identified in recovery plans and action plans. Detailed analytical methods are specified for making such identifications, with the methods to use depending on the level of information available. (see Annex 11.2)
What % spatial area must be protected?	The guidelines vary with the protection perceived as necessary	100% of "essential habitat" for each exploited species must have its quality preserved/	100% of "critical habitat" – unless an "incidental Harm Permit" has been issued. These can only be issued if the harm will not "jeopardize survival or recovery of the species.
How is 'quality' assessed / incorporated?	According to favourable conservation status	Guidelines do not specify, beyond using "best available science"	Analytical scientific methods must be used, and acceptable options are included in the guidelines. If data are insufficient to allow quantification of "quality" the recovery plan must include provisions to acquire the necessary data and knowledge.
How are life-history stages incorporated?	Annex I: n/a Annex II: habitats of protected species must be protected for all life history stages, especially key stages e.g. breeding or nursery grounds.	The Act requires that specification of Essential habitat in the FMP must include the habitat needed by all life-history stages	The technical guidelines specify that a "comprehensive summary of the species essential life history stages and key habitat needs" must be developed.
Is there provision for networks of MPAs?	Yes – Natura 2000	Covered in other legislation	Canada's Oceans Act includes provisions that MPAs can be established to protect important habitat features. No guidelines are in place to determine what is an important habitat feature.
Ease of implementation?	Resource intensive	To ensure that the science used in determining the essential habitat was the "best available" requires substantial scientific work. The specified processes also require extensive consultation.	Analytical tasks to be completed are demanding, although the guidance is clear. The requirement to increase knowledge when it is weak is also demanding on the science support for SARA
What are the science/ data requirements?	Surveyed every six years after designation	The Act only specifies that geographic positions of essential habitat must be given. The data requirements to determine what habitats are essential are left to the "best available science" advisors (and judges) to evaluate.	Characterizing the relationship between essential life history stages/key activities and habitat features;. Habitat attributes … may include biotic (e.g., vegetation) and abiotic (e.g., soil) elements, as well as ecological processes (e.g., pollination, disturbance). Evaluate and describe any potential biases associated with different types of species occurrence data – biases that could skew an understanding of current and historic distribution and status. Identify and document areas that have not been adequately surveyed but which could potentially support populations of the species.

The U.S. and Canadian frameworks are highly detailed. WGECO often criticises programmes for being long on conceptual terms and short on detail; that is not the case here. The details differ. The U.S. framework has many definitional details, and then most of the framework and guidelines are about process and communication. It references "best science" but gives little guidance on what science will be the best under what conditions.

The Canadian framework has lots of definitional detail, and few major inconsistencies with the U.S. definitions. (Appendix 11.3). Otherwise the Canadian framework is largely about what scientific activities should be the basis for the designation under what circumstances. Taken together it presents a set of difficult choices:

• If they conclude that they are data and knowledge poor for a particular species, then some very superficial things are done. We have concerns that the results would give weak justification to be confident that critical habitat was, in fact, being protected – or that management might be highly intrusive on human activities that were not harming recovery.

• If they conclude that there are some data, or can collect it, the analytical demands are very high and require knowledge of species abundance – habitat feature relationships which rarely are well documented.

11.3 Existing measures of "habitat quality"

Several indices of habitat quality have been developed which assess the physical, chemical and biological attributes of habitats. The indices differ in the weight placed on the attributes to assess quality; some indices favour the physical structure of the habitat over the biological communities present.

11.3.1 Index of biotic integrity (IBI) (Karr *et al*., 1986)

The IBI assesses habitats based on the biological communities they support, with an emphasis on species richness and indicator species. The index compares the assessment sites to a reference site which is considered relatively un-impacted.

The IBI was developed originally for freshwater systems but has evolved to include marine systems. A NOAA workshop was held to develop an IBI for marine benthic and pelagic habitats for the purposes of assessing essential fish habitat and concluded that although IBI was a suitable index to measure benthic habitat quality, it was not suitable for measuring the quality of water column habitats as they were too variable, too dynamic and too transient in quality (Hartwell, 1998).

The method devised for measuring marine benthic habitat quality was: 1) categorise the benthic habitat as having soft bottom, hard bottom or live bottom substrates, 2) categorise the area as estuarine (submerged or intertidal), coastal shore zone or offshore, 3) divide the assessment area according to geographical boundaries based on large scale oceanographic and geological features, 4) measure the 'health' of the biological community. The latter was to be assessed by:

- Infauna community structure, composition, number of organisms and biomass by taxa
- Shellfish, epibenthic fish, benthic foraging fish community structure, composition, number of organisms and biomass by taxa
- Percent spatial extent of 3-D refugia
- Percent spatial extent of living refugia verses total refugia
- Dominance by selected species (opportunistic verses equilibrium)
- Changes in dominance
- Biomass of fish food
- Contaminant impact (e.g. incidence of disease, dominance of pollution tolerant species)
- The age structure of selected species (as a measure of physical disturbance/chemical impact)

Specific to estuaries:

- Measures of resident verses migratory species
- Functional parameters of selected species (e.g., filtration capacity)

The characteristics of healthy and degraded habitat were identified as:

Degraded	Healthy
• Low diversity	• High diversity
• High dominance by selected species	• Low dominance
• High proportion of immature individuals	• Stable age structure
• High proportion of tolerant species	• Low proportion of tolerant species
• High proportion of r selected species	• High proportion of K selected species
• High chemical body burdens	• Low chemical body burdens
• High disease/lesion incidence	• Low disease/lesion incidence
• Low coverage by biological refugia	• High coverage by biological refugia

11.3.2 Organism-sediment index (OSI) (Rhoads and Germano, 1986)

The OSI is more process orientated than the IBI and uses images to record the end products of biological and physical processes that structure benthos (Diaz *et al*., 2003). Data are collected by sediment profile images to estimate the depth of the apparent colour redox potential (RPD) layer, the successional stages of the macrofauna, the presence of gas bubbles in the sediment (an indication of high rates of methanogenisis), and the presence of reduced sediment at the sediment water interface that would indicate current or recent low dissolved oxygen conditions to assess the quality of the benthic habitat (Diaz *et al*., 2003).

Comparisons of the IBI and OSI (Diaz *et al*., 2003)

A comparison of the IBI and the OSI was conducted in Chesapeake Bay, USA (Diaz *et al*., 2003). The results showed significant differences in the assessment of habitats as stressed or of good quality. When IBI indicated poor conditions, the OSI tended to indicate good quality habitat. The authors argued that this result was to be expected as the benthic habitat quality (as measured by the OSI) would improve before biotic integrity (as measured IBI).

11.3.3 Benthic Habitat Quality Index (BHQ) (Nilsson and Rosenberg, 1997)

The BHQ uses sediment surface and sediment profile images to assess sediment characteristics (texture, oxic/anoxic conditions, lamination) which can be related to functional properties of macrofauna (burrows, tubes, feeding voids, reworked sediments) which will give an indication of habitat quality. The BHQ was developed in relation to benthic faunal successional models developed by Pearson and Rosenberg (1976) and OSI (Rhoads and Germano (1986).

11.3.4 Habitat Affinity Indices (HAI) (Nelson and Monaco, 1999)

HAI defines habitat affinity based on the relative concentration of a species in a particular habitat compared with the availability of that habitat in the study area. Measurements include dissolved oxygen, temperature, salinity, depth, substrate type, sediment contaminants and toxicity and the size and species present in that area.

11.3.5 Submerged Aquatic Vegetation (SAV) Habitat Quality Index

(http://www.epa.gov/bioindicators/html/marinetidal.html)

The U.S. Chesapeake Bay restoration program has focused on SAV for the Bay grasses, as they require light and suitably low nutrient levels in the water. They have set a goal of providing adequate habitat to 1m depth for SAV. To develop this indicator, Chesapeake Bay Program Bay segments were assessed using 1994 to 1996 data and were scored as passing, failing or borderline for SAV habitat requirements: Secchi depth (a measure of water clarity), dissolved inorganic nitrogen, dissolved inorganic phosphorus, chlorophyll a (a measure of algae), and suspended solids. In some areas only four habitat requirements apply; dissolved inorganic nitrogen habitat requirements do not apply in tidal fresh and oligohaline, or very low salinity, areas. Scores for each segment are a composite based on all applicable habitat requirements.

Scores are adjusted to range between 1 and 10 (1 being most degraded, 10 representing the best condition). No area in Maryland is considered to be pristine in terms of SAV habitat quality. For the Unified Watershed Assessment, developed in 1998 under the Clean Water Action Plan, watersheds were identified as needing restoration if they are scored lower than 7. Watersheds scored 7 or higher were considered to justify preventive measures in order to maintain their relatively good condition. To score this high, all parameters for a segment had to be assessed as at least borderline in quality.

11.3.6 EC Water Framework Directive (WFD) (EC, 2000)

The WFD is included in this section as it the first EC Directive to recognise the importance of aquatic biota in assessing the quality of European fresh and coastal marine waters. Specific objectives of the Directive are to prevent further deterioration and protect and enhance the status of the aquatic ecosystems.

The Directive is unique in setting ecological targets ('high and good ecological status') for surface waters and in doing so, the Commission has recognised the need for an integrated approach to managing three of the components of aquatic habitats: water quality, water quantity, and physical structure. In this regard, the Directive is consistent with and complimentary to the EcoQO concept.

Implicit in the Directive are the establishment of programmes of measures designed to bring those water bodies not at good status up to required levels. It is assumed that the classification of physio-chemical conditions and hydromorphological elements (habitat) will give guidance to the appropriateness of these programmes of measures for achieving improvements in habitat status.

Logan and Furse (2002) give a good overview of the Directive in the way that habitat variables and biota are interlinked to give rise to ecological targets and measures to achieve them. Although in this paper consideration is restricted to the link between habitat and biota for rivers, many of the principles discussed will also apply to lakes, transitional waters, and coastal waters. Open water marine environments are not considered in the WFD. This does not suppose that open marine environments will not be included at some time in the future.

They conclude that to design suitable programmes of remedial measures, it will be important to diagnose the cause of the failure to achieve good status. Providing the right diagnosis will lead to the most appropriate and cost effective programme of measures. This diagnosis will be an integration of the information from the identification of pressures and from the monitoring of biological, physiochemical and hydromorphological elements. It is as this stage that the clearest link between habitat factors and biological elements will be required. All of the biological metrics already developed are specific to particular pressures for example Biological Monitoring Working Party (DoE, 1978) scores relate macro-invertebrate communities to organic pollution.

Implementation of the WFD:

Future monitoring programmes must comply with a number of mandatory elements identified in the Directive document (REF).

Guidance documents (Common Implementation Strategy documents: Commission's website) have been ratified by 'Water Directors' as an interpretation of the statute and include a number of additional or optional recommended quality elements.

National governments and their agencies have undertaken to consider the guidance documentation in the development of methodologies capable of acquiring the desired information on identified quality elements, assessing those quality elements and complying with parameters necessary for the integration of that data at a European level. This is a programme of work that is currently ongoing in many jurisdictions.

The general criteria for assessment in the WFD are:

- an assessment of the deviation of observed conditions to those that would normally be found under reference conditions;
- an assessment that provides for natural and artificial habitat variation;
- a protocol that accounts for the range of natural variability and variability arising from anthropogenic activities of all quality elements in all water body types; and

- a scheme that provides for the detection of the full range of potential impacts (including hydro-morphological elements) to enable robust classification of ecological status.

The River Invertebrate Prediction and Classification System (RIVPACS) (Wright, 2000) influenced the drafting of the Directive, with the concept of a reference state and selection of river typology variables. RIVPACS argue that there is an ecologically robust link between habitat variables and biological elements (Logan and Furse, 2002).

As an assessment of the deviation of observed conditions to those that would normally be found under reference conditions forms the basis for any assessment, considerable effort will need to be expended in building appropriate reference models. This process is still in early stages. Recent calls for research and development of both methodological approaches and model construction and suggest that this process has still quite a bit to go (www.sniffer.org.uk/). The guidance documentation for the WFD recognises that even for transitional and coastal marine environments that models are generally not well developed or validated for the marine environment and given the problems with using historical data, a reference network of high status sites is likely to be the preferred approach for deriving reference conditions for transitional and coastal waters. The development of monitoring programmes and an evaluation of the capacity of applied methodologies is required before the Directive can be implemented.

Habitat is not defined in the WFD documentation as it is in U.S. and Canadian literature (Annex 11.3) as a discrete element. Although the classification of ecological status is based upon the status of the biological, hydromorphological and physico-chemical quality elements, the hydromorphological and physico-chemical elements are referred to as the supporting elements.

Supporting, as interpreted in the CIS Guidance documents, means that the values of the hydro-morphological quality elements are such as to support a biological community of a certain ecological status. This recognises the fact that biological communities are products of their physical environment. The Directive does not intend that supporting elements, in this instance, hydromorphology, can be used as surrogates for the biological elements in surveillance and operational monitoring. Instead, the monitoring and assessment of the physical quality elements will support the interpretation, assessment and classification of the results arising from the monitoring of the biological quality elements.

11.3.7 Summary of existing indices and the WFD

All the indices and the WFD measure the physical, chemical and biological characteristics of habitats and use indicator species, in some part, to assess quality. The use of indicator species is much debated and inconsistencies in the response of species to stressors is well documented (Jones and Kaly, 1996; Linke-Gamerick *et al*., 2000; Mendez *et al*., 2000; Forbes *et al*., 2001; Bustos-Baez and Frid, 2003). The IBI and HAI are reliant on comparisons to control sites and therefore may not be the most suitable index for areas impacted by human disturbance for several centuries.

11.4 How can this EcoQO element be advanced?

Before any progress with this EcoQO can be made, the habitat types to which it is to be applied and their distribution will need to be determined. European habitat types have been classified by EUNIS, which is a hierarchical system which uses both physical descriptors and characterising species to identify habitat types. In the Northeast Atlantic, a project to map seabed habitats (Mapping European Seabed Habitats (MESH)) has been established. The area to be mapped is shown in Figure 11.4.2.

Frameworks which identify habitats of high concern (e.g., rare, endangered or considered “essential” for life history stages of species considered important) will identify those areas whose protection is a high priority. Within Europe, the Habitat and Species Directive (1992) identifies those habitats in need of protection (Annex I) and some species whose protection needs to be applied through the protection of their habitat (Annex II).

Current measures of habitat quality use indicator species and reference sites to assess the status of habitats. Although the use of indicator species and communities is controversial (see Section 11.3.7), and reference sites may not be suitable for areas which have been impacted for centuries, they are a central component of several indices of habitat quality which are currently in use.

The EcoQ for habitats is to “restore and/or maintain habitat quality”. Figure 11.4.1 presents a framework that can distinguish whether a particular habitat requires restoration or maintenance.

For those habitat types which receive statutory protection, the requirement is that their quality is maintained. This pre-supposes that they are high quality on designation (i.e., favourable conservation status under the Habitats Directive).

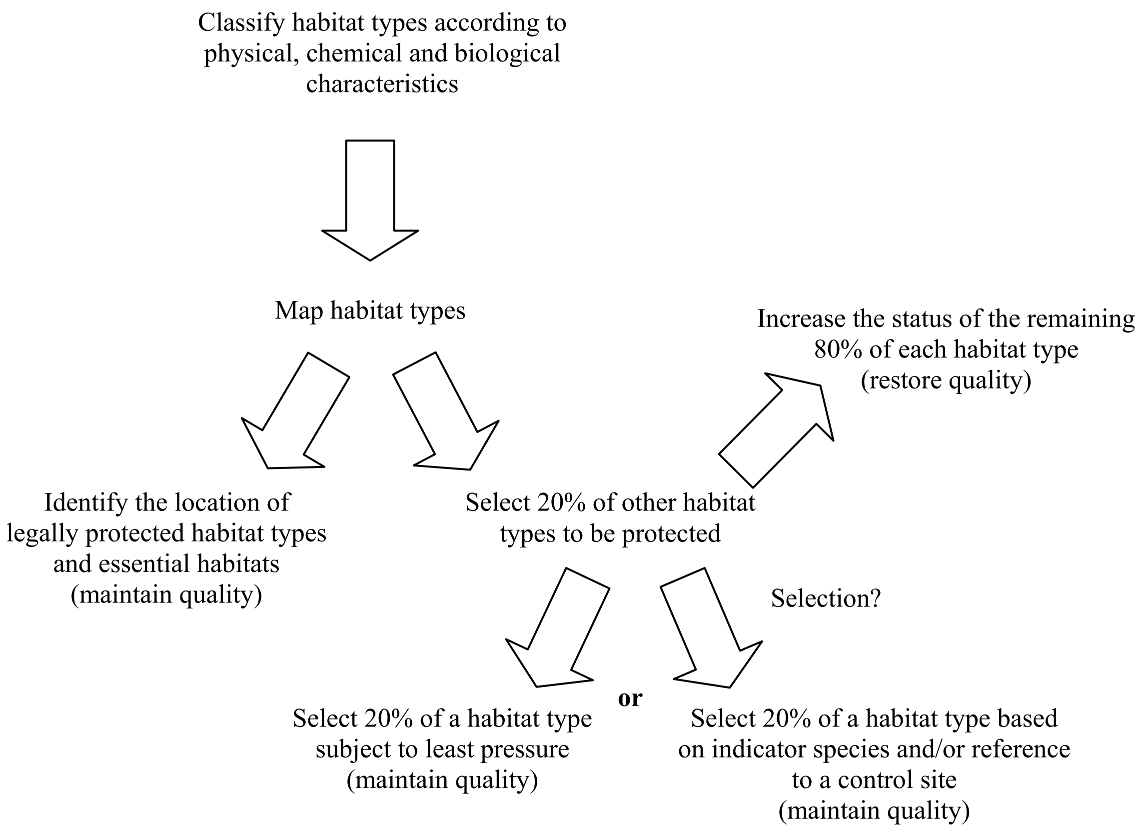
Of those habitat types which are not protected specifically by legislation, the IUCN recommendations imply that 20% of each habitat type be protected. The logical development of this line is that 20% of the highest quality areas of each habitat type should be protected and hence maintained.

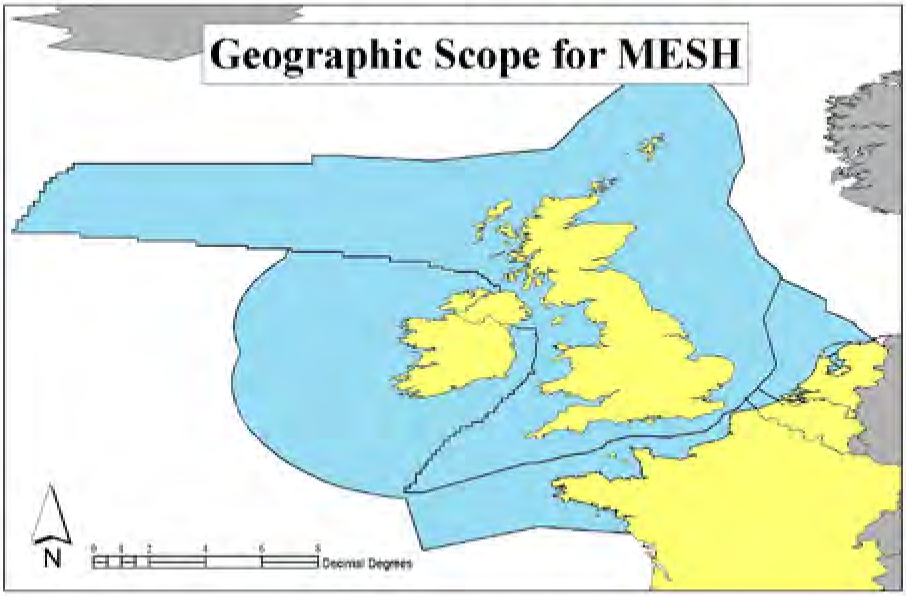
The implication of this then is that the remaining 80% of each habitat type will need to be maintained if in favourable conservation status or improved if not.

Thus, this EcoQO will be met by the protection of all designated habitats types and for the 20% (if the IUCN recommendations are followed) of all other types. To achieve the assessment of quality and, if necessary, achieve improvement of the remainder will require development of a means of assessing habitat quality in a rigorous and ecologically relevant way. As this is made operational, parallels may emerge with the WFD requirement to improve the ecological status of water bodies. WGECO is not convinced that the indicator species-based indices used in fresh and transitional waters will be applicable in the more geographically extensive and ecologically dynamic seas and oceans.

The protection of 20% of all habitat types has been recommended by the IUCN (2003), whilst the European Community Biodiversity Strategy suggests that more than 10–20% of the world's territory could potentially be designated as a protected area (EC, 2001). The selection of these areas should be based on the highest quality example of those habitat types. One method of achieving this would be by identifying the level of human-mediated disturbance on a habitat type and select those areas of that habitat type exposed to the least stress for protection. The assumption here is that since these sites are the least stressed, they are most likely to be in the best condition for that habitat type. This would avoid reliance on indicator species and control reference sites. For the remaining 80% of the habitat area, which will need to improve (restore) quality, the 20% which are protected can be used as reference sites.

Figure 11.4.1 A framework for developing habitat quality as an EcoQO.





(http://www.jncc.gov.uk/marine/mesh/default.htm)

Figure 11.2 The geographic scope of the "Mapping European Seabed Habitats" project (MESH). The project will cover the sea areas mapped in blue. Boundaries are country EEZs (or equivalent), except France, where the southern boundary relates to southern limit of the Interreg North-West Europe area.

11.5 References

Bustos-Baez S. and Frid, C.L.J. (2003) Using indicator species to assess the state of macrobenthic communities. *Hydrobiologica* **496** 299-309

Collie, JS; Escanero, GA; Valentine, PC (1997) Effects of bottom fishing on the benthic megafauna of Georges Bank *Marine Ecology Progress Series* **155** 159-172.

Diaz, R.J., Cutter, G.R. and Dauer, D.M. (2003) A comparison of two methods for estimating the status of benthic habitat quality in the Virginia Chesapeake Bay. *Journal of Experimental Marine Biology and Ecology* **285-286** 371-381

DoE (1978). Biological Monitoring Working Party. Final report: assessment and presentation of the quality of rivers in Great Britain. Unpublished report, Department of the Environment, Water Data Unit, UK.

EC (1992) Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora. (Habitats Directive)

EC (2000) Council Directive 2000/60/EC of the European Parliament and of the Council establishing a framework for the Community action in the field of water policy. (Water Framework Directive)

EC (2001) Communication from the Commission to the Council and the European Parliament. Biodiversity action plans in the areas of Conservation of Natural Resources, Agriculture, Fisheries, and Development and Economic Co-operation. Brussels, 27.3.2001 COM(2001)162 final.

Endangered Species Act (1973). US Public Law

Forbes, V. Andreasson MSH and Christensen, L. (2001) Metabolism of the polycyclic aromatic hydrocarbon fluoranthene by the polychaete *Capitella capitata* species. *Environ. Tox. Chem*. **20** 1012-1021

Gotceitas, V; Brown, JA (1993) Risk of predation to fish larvae in the presence of alternative prey: Effects of prey size and number. *Marine Ecology Progress Series* **98** (3) 215-222

Hacunda, JS (1981) Trophic Relationships Among Demersal Fishes in a Coastal Area of the Gulf of Maine. *hery Bulletin*. **79** (4) 775-788

Jones, GR and Kaly UI (1996) Criteria for selecting marine organisms in biomonitoring studies. In: Schmidt, R.J. and Osenberg CW (eds) Detecting ecological impacts: Concepts and applications in coastal habitats. Academic Press, San Diego, California. pp.29-48

Karr, J.R., K.D. Fausch, P.L. Angermeier, P.R. Yant, and I.J. Schlosser. 1986. *Assessing biological integrity in running waters, a method and its rational*. Illinois Natural History Survey, Special Publication 5.

Lilly, GR; Parsons, DG (1991) Distributional patterns of the northern shrimp (*Pandalus borealis*) in the Northwest Atlantic as inferred from stomach contents of cod (*Gadus morhua*). ICES COUNCIL MEETING PAPERS., ICES, COPENHAGEN (DENMARK), 15 pp

Linke-Gamerick, I, Vismann, B. and Forbes, V.E. (2000) Effects of fluoranthene and ambient oxygen on survival and metabolism in three sibling species of *Capitella* (Polychaeta). *Marine Ecology Progress Series* **194** 169-177

Logan, P. and Furse, M. (2002). Preparing for the Water Framework Directive – making the links between habitat and aquatic biota. *Aquatic Conservation* **12** 425-437.

Lough, RG; Valentine, PC; Potter, DC;Auditore, PJ; Bolz, GR; Neilson, JD; Perry, RI (1989). Ecology and distribution of juvenile cod and haddock in relation to sediment type and bottom currents on eastern Georges Bank. *Marine Ecology Progress Series*. **56** (1-2) 1-12.

Magnuson-Stevens Fishery Conservation and Management Act (1996). US Public Law 94-265.

Mendez, N. Linke-Gamenick, I., and Forbes V.E. (2000) Variability in reproductive mode and larval development within the *Capitella capitata* species complex. *Invertebr. Reprodu. Dev*. **38** 131-142

Nilsson HC and Rosenberg R. (1997). Benthic habitat quality assessment of an oxygen stressed fjord by surface and sediment profile images. *Journal of Marine Systems* **11** 249-264

Nelson, DM and Monaco, ME (1999). http://www.gap.uidaho.edu/Meetings/Posters/99Posters/nelson.htm

Species at Risk Act (2002). Canada.

Tupper, M; Boutilier, RG (1995) Size and priority at settlement determine growth and competitive success of newly settled Atlantic cod. *Marine Ecology Progress Series*. **118** (1-3) 295-300

Rhoads, D.C., and J.D. Germano (1986) Interpreting Long-Term Changes in Benthic Community Structure: A New Protocol. *Hydrobiologia* **142** 291-308

UK Biodiversity Action Plan (2002) http://www.ukbap.org.uk/2002OnlineReport/mainframe.htm

Witman, J. D. and Sebens K. P. (1992) Regional variation in fish predation intensity: a historical perspective in the Gulf of Maine. *Oecologia* **90** 305 - 315.

Wright, J.F. (2000). An introduction to RIVPACS. *In Assessing the Biological Quality of freshwaters: RIVPACS and other techniques*, Wright, J.F., Sutcliffe, D.W., Fuse, M.T. (eds). Freshwater Biological Association: Ambleside: 1-24.

APPENDIX 11.1

1 THE US FRAMEWORK AND GUIDELINES

Most of this text is extracted directly from the US documentation, available on http://www.nero.noaa.gov/ro/doc/webintro.html. Where material of particular relevance to this evaluation is contained in the extracts, it is highlighted with **bold** type. The US document is extensive and very detailed, consistent with the legal context in which the provisions are applied. The extracted material required cuts of some technical material to reduce the length of this section and focus the contents on the issues of interest to WGECO. Where material has been deleted from text extracted from the NOAA website, deletions are marked with '…'. Where WGECO has inserted text into the extracts, the inserted text is in Arial font rather than Times New Roman. For any official interpretations of the US framework and guidelines, the original material should be used.

1.1 What is EFH?

The United States Congress defined EFH as "those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity" (16 U.S.C. 1802(10)). The EFH guidelines under 50 CFR 600.10 further interpret the EFH definition ….:

An EFH provision in an FMP [Fishery Management Plan] **must include all fish species** in the fishery management unit (FMU). An FMP may describe, identify, and protect the habitat of species not in an FMU; however, such habitat may not be considered EFH for the purposes of sections 303(a)(7) and 305(b) of the Magnuson-Stevens Act.

1.2 Definitions (also Annex 3)

Essential Fish Habitat (EFH): those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity (16 U.S.C. 1802(10)).

Waters include 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 (50 CFR 600.10).

Substrate includes sediment, hard bottom, structures underlying the waters, and associated biological communities (50 CFR 600.10).

Necessary means the habitat required to support a sustainable fishery and the managed species' contribution to a healthy ecosystem (50 CFR 600.10).

Habitat Areas of Particular Concern (HAPC) are subsets of EFH identified based on one or more of the following considerations: 1) the importance of the ecological function, 2) extent to which the habitat is sensitive to human-induced degradation, 3) whether and to what extent, development activities are stressing the habitat type, or 4) rarity of habitat type (50 CFR 600.815(a)(8)).

Adverse effect is any impact which reduces quality and/or quantity of EFH. Adverse effects may include direct or indirect physical, chemical, or biological alterations of the waters or substrate and loss of, or injury to benthic organisms, prey species and their habitat, and other ecosystem components. Adverse effects may be site- specific or habitat-wide impacts, including individual, cumulative, or synergistic consequences of actions ((50 CFR 600.910(a)).

Fishery Management Plan (FMP) is a plan to achieve specified management goals for a fishery. It includes data, analyses, and management measures (including guidelines for harvest) for a fishery.

EFH Assessment is a written assessment of the effects of a proposed Federal action on EFH (50 CFR 600.920(e). Federal agencies must provide NMFS with an EFH Assessment for any action that may adversely affect EFH, except for those activities covered by a General Concurrence. An EFH Assessment must contain, 1) A description of the proposed action, 2) An analysis of the effects, including cumulative effects, of the proposed action on EFH and managed species, 3) The Federal agency's conclusions regarding the effects of the action on EFH and, 4) Proposed mitigation, if applicable (50 CFR 600.920(e)(3)). If appropriate, the EFH Assessment should also include the items listed at 50 CFR 600.920(e)(4). The level of detail in an EFH Assessment should be commensurate with the potential impacts to EFH (50 CFR 600.920(e)(2)).

EFH Conservation Recommendations are recommendations provided by NMFS to a Federal or state agency pursuant to section 305(b)(4)(A) of the Magnuson-Stevens Act regarding measures that can be taken by that agency to conserve EFH. EFH conservation recommendations may be provided as part of an EFH consultation with a Federal agency, or may be provided by NMFS to any Federal or state agency whose actions would adversely affect EFH (50 CFR 600.925).

Finding is a determination by NMFS that an existing or modified consultation/coordination process satisfies the Federal agency consultation requirements of section 305 of the Magnuson-Stevens Act (50 CFR 600.920(f)(3)).

Ecosystem means communities of organisms interacting with one another and with the chemical and physical factors making up their environment.

Healthy ecosystem means an ecosystem where ecological productive capacity is maintained, diversity of the flora and fauna is reserved, and the ecosystem retains the ability to regulate itself. Such an ecosystem should be similar to comparable, undisturbed ecosystems with regard to standing crop, productivity, nutrient dynamics,

1.3 US Legal Requirements

1.3.1 Implementing the EFH Mandate and Protection

Section 303(a)(7) of the Magnuson-Stevens Act requires Regional Fishery Management Councils to describe and identify EFH for each federally managed species. Many parties participate in the public process of designating EFH. The eight Councils, which have the responsibility for drafting fishery management plans (FMPs), are charged with proposing EFH descriptions and identifications for each life stage of the managed species in their jurisdiction. These descriptions and identifications must be based on the best available science regarding the habitat requirements of each managed species and are developed through a public process with many opportunities for input. … Fishery Management Council EFH descriptions and identifications can be found at each of the **Council EFH** pages.

1.3.2 Minimizing Adverse Fishing Impacts

In addition to designating EFH, Councils must also minimize adverse impacts from fishing activities on EFH, to the extent practicable, in accordance with Section 303(a)(7) of the Magnuson-Stevens Act. First, Councils are required to assess the impacts of fishing practices on EFH in their regions. Second, if a fishing practice is determined to have an adverse impact on EFH, Councils must adopt measures to minimize that impact, to the extent practicable. To meet this requirement, Councils may develop measures such as fishing equipment restrictions or time/area closures. Councils are encouraged to give special consideration to adverse impacts of fishing on Habitat Areas of Particular Concern (HAPCs).

1.3.3 Consulting on Fishing and Non-fishing Impacts

Section 305(b)(2)-(4) of the Magnuson-Stevens Act outlines a process for NMFS and the Councils to comment on activities proposed by Federal action agencies that may adversely impact areas designated as EFH. ….

1.4 The Mandate

In Section 303(a)(7) of the amended Magnuson-Stevens Act, Congress directs the National Marine Fisheries Service (NMFS) and the eight regional Fishery Management Councils, under the authority of the Secretary of Commerce, to:

- Describe EFH and identify EFH in each fishery management plan,
- Minimize to the extent practicable the adverse effects of fishing on EFH, and
- Identify other actions to encourage the conservation and enhancement of EFH

In Section 305 (b)(2) of the amended Magnuson - Stevens Act, Congress directs each Federal Agency to consult with the Secretary with respect to any action authorized, funded, or undertaken, or proposed to be authorized, funded, or undertaken, by such agency that may adversely affect any essential fish habitat identified under the Magnuson - Stevens Act

1.5 EFH and Fishery Management Plans

Note: The Fishery Management Plans developed by the regional Fisheries Management Councils are the primary tool for bringing essential habitat considerations into the management process. Again, substantial formal guidance is provided on how EFH is to be treated in Fishery Management Plans.

(a) *Mandatory contents*—(1) *Description and identification of EFH*—(i) *Overview* . FMPs must describe and identify EFH in text that clearly states the habitats or habitat types determined to be EFH for each life stage of the managed species. FMPs should explain the physical,biological, and chemical characteristics of EFH and, if known, how these characteristics influence the use of EFH by the species/life stage. FMPs must identify the specific geographic location or extent of habitats described as EFH. FMPs must include maps of the geographic locations of EFH or the geographic boundaries within which EFH for each species and life stage is found.

(ii) *Habitat information by life stage.*

(A) … Pertinent information includes the geographic range and habitat requirements by life stage, the distribution and characteristics of those habitats, and current and historic stock size as it affects occurrence in available habitats. FMPs should summarize the life history information necessary to understand each species' relationship to, or dependence on, its various habitats, using text, tables, and figures, as appropriate. FMPs should document patterns of temporal and spatial variation in the distribution of each major life stage (defined by developmental and functional shifts) to aid in understanding habitat needs. FMPs should summarize (e.g., in tables) all available information on environmental and habitat variables that control or limit distribution, abundance, reproduction, growth, survival, and productivity of the managed species. The information should be supported with citations. ….. FMPs should identify species-specific habitat data gaps and deficits in data quality (including considerations of scale and resolution; relevance; and potential biases in collection and interpretation). FMPs must demonstrate that the best scientific information available was used in the description and identification of EFH, consistent with national standard 2.

(iii) *Analysis of habitat information.*

(A) The following approach should be used to organize the information necessary to describe and identify EFH.

(*1*) *Level 1: Distribution data are available for some or all portions of the geographic range of the species.* At this level, only distribution data are available to describe the geographic range of a species (or life stage). …In the event that distribution data are available only for portions of the geographic area occupied by a particular life stage of a species, habitat use can be inferred on the basis of distributions among habitats where the species has been found and on information about its habitat requirements and behavior. Habitat use may also be inferred, if appropriate, based on information on a similar species or another life stage.

(*2*) *Level 2: Habitat-related densities of the species are available.* At this level, quantitative data (i.e., density or relative abundance) are available for the habitats occupied by a species or life stage. Because the efficiency of sampling methods is often affected by habitat characteristics, strict quality assurance criteria should be used to ensure that density estimates are comparable among methods and habitats. Density data should reflect habitat utilization, and the degree that a habitat is utilized is assumed to be indicative of habitat value. When assessing habitat value on the basis of fish densities in this manner, temporal changes in habitat availability and utilization should be considered.

(*3*) *Level 3: Growth, reproduction, or survival rates within habitats are available.*

At this level, data are available on habitat-related growth, reproduction, and/or survival by life stage. The habitats contributing the most to productivity should be those that support the highest growth, reproduction, and survival of the species (or life stage).

(*4*) *Level 4: Production rates by habitat are available.* At this level, data are available that directly relate the production rates of a species or life stage to habitat type, quantity, quality, and location. Essential habitats are those necessary to maintain fish production consistent with a sustainable fishery and the managed species' contribution to a healthy ecosystem.

(B) Councils should strive to describe habitat based on the highest level of detail (i.e., Level 4). If there is no information on a given species or life stage, and habitat usage cannot be inferred from other means, such as information on a similar species or another life stage, EFH should not be designated.

(iv) *EFH determination.*

(A) Councils should analyze available ecological, environmental, and fisheries information and data relevant to the managed species, the habitat requirements by life stage, and the species' distribution and habitat usage to describe and identify EFH. … Councils should interpret this information in a risk-averse fashion to ensure adequate areas are identified as EFH for managed species. Level 1 information, if available, should be used to identify the geographic range of the species at each life stage. If only Level 1 information is available, distribution data should be evaluated (e.g., using a frequency of occurrence or other appropriate analysis) to identify EFH as those habitat areas most commonly used by the species. Level 2 through 4 information, if available, should be used to identify EFH as the habitats supporting the highest relative abundance; growth, reproduction, or survival rates; and/or production rates within the geographic range of a species. FMPs should explain the analyses conducted to distinguish EFH from all habitats potentially used by a species.

(B) FMPs must describe EFH in text, including reference to the geographic location or extent of EFH using boundaries such as longitude and latitude, isotherms, isobaths, political boundaries, and major landmarks. …. Text and tables should explain pertinent physical, chemical, and biological characteristics of EFH for the managed species and explain any variability in habitat usage patterns, but the boundaries of EFH should be static.

(C) If a species is overfished and habitat loss or degradation may be contributing to the species being identified as overfished, all habitats currently used by the species may be considered essential in addition to certain historic habitats that are necessary to support rebuilding the fishery and for which restoration is technologically and economically feasible. Once the fishery is no longer considered overfished, the EFH identification should be reviewed and amended, if appropriate.

(D) Areas described as EFH will normally be greater than or equal to aquatic areas that have been identified as ''critical habitat'' for any managed species listed as threatened or endangered under the Endangered Species Act.

(E) Ecological relationships among species and between the species and their habitat require, where possible, that an ecosystem approach be used in determining the EFH of a managed species. EFH must be designated for each managed species, but, where appropriate, may be designated for assemblages of species or life stages that have similar habitat needs and requirements. If grouping species or using species assemblages for the purpose of designating EFH, FMPs must include a justification and scientific rationale.

(F) If degraded or inaccessible aquatic habitat has contributed to reduced yields of a species or assemblage and if … the degraded conditions can be reversed through such actions as improved fish passage techniques (for stream or river blockages), improved water quality measures (removal of contaminants or increasing flows), and similar measures that are technologically and economically feasible, EFH should include those habitats that would be necessary to the species to obtain increased yields.

(B) Where the present distribution or stock size of a species or life stage is different from the historical distribution or stock size, then maps of historical habitat boundaries should be included in the FMP, if known.

(2) *Fishing activities that may adversely affect EFH*—

(i) *Evaluation.* Each FMP must contain an evaluation of the potential adverse effects of fishing on EFH designated under the FMP, …. This evaluation should consider the effects of each fishing activity on each type of habitat found within EFH. FMPs must describe each fishing activity, review and discuss all available relevant information (such as information regarding the intensity, extent, and frequency of any adverse effect on EFH; the type of habitat within EFH that may be affected adversely; and the habitat functions that may be disturbed), and provide conclusions regarding whether and how each fishing activity adversely affects EFH. The evaluation should also consider the cumulative effects of multiple fishing activities on EFH. The evaluation should list any past management actions that minimize potential adverse effects on EFH and describe the benefits of those actions to EFH. The evaluation should give special attention to adverse effects on habitat areas of particular concern and should identify for possible designation as habitat areas of particular concern any EFH that is particularly vulnerable to fishing activities. Additionally, the evaluation should consider the establishment of research closure areas or other measures to evaluate the impacts of fishing activities on EFH. In completing this evaluation, Councils should use the best scientific information available, as well as other appropriate information sources. Councils should consider different types of information according to its scientific rigor.

(ii) *Minimizing adverse effects.* Each FMP must minimize to the extent practicable adverse effects from fishing on EFHs. Councils must act to prevent, mitigate, or minimize any adverse effects from fishing, to the extent practicable, if there is evidence that a fishing activity adversely affects EFH in a manner that is more than minimal and not temporary in nature, based on the evaluation and/or the cumulative impacts analysis conducted. … In such cases, FMPs should identify a range of potential new actions that could be taken to address adverse effects on EFH, include an analysis of the practicability of potential new actions, and adopt any new measures that are necessary and practicable …

(iii) *Practicability.* In determining whether it is practicable to minimize an adverse effect from fishing, Councils should consider the nature and extent of the adverse effect on EFH and the long and short-term costs and benefits of potential management measures to EFH, associated fisheries, and the nation, consistent with national standard 7. In determining whether management measures are practicable, Councils are not required to perform a formal cost/benefit analysis.

(4) *Non-fishing related activities that may adversely affect EFH.* FMPs must identify activities other than fishing that may adversely affect EFH. Broad categories of such activities include, but are not limited to: dredging, filling, excavation, mining, impoundment, discharge, water diversions, thermal additions, actions that contribute to non-point source pollution and sedimentation, introduction of potentially hazardous materials, introduction of exotic species, and the conversion of aquatic habitat that may eliminate, diminish, or disrupt the functions of EFH. For each activity, the FMP should describe known and potential adverse effects to EFH.

(5) *Cumulative impacts analysis…* To the extent feasible and practicable, FMPs should analyze how the cumulative impacts of fishing and non-fishing activities influence the function of EFH on an ecosystem or watershed scale. An assessment of the cumulative and synergistic effects of multiple threats, including the effects of natural stresses (such as storm damage or climate-based environmental shifts) and an assessment of the ecological risks resulting from the impact of those threats on EFH, also should be included.

(6) *Conservation and enhancement.* FMPs must identify actions to encourage the conservation and enhancement of EFH, including recommended options to avoid, minimize, or compensate for the adverse effects identified [previously]

(7) *Prey species.* Loss of prey may be an adverse effect on EFH and managed species because the presence of prey makes waters and substrate function as feeding habitat, and the definition of EFH includes waters and substrate necessary to fish for feeding. Therefore, actions that reduce the availability of a major prey species, either through direct harm or capture, or through adverse impacts to the prey species' habitat that are known to cause a reduction in the population of the prey species, may be considered adverse effects on EFH if such actions reduce the quality of EFH. FMPs should list the major prey species for the species in the fishery management unit and discuss the location of prey species' habitat.

(8) *Identification of habitat areas of particular concern.* FMPs should identify specific types or areas of habitat within EFH as habitat areas of particular concern based on one or more of the following considerations:

(i) The importance of the ecological function provided by the habitat.

(ii) The extent to which the habitat is sensitive to human-induced environmental degradation.

(iii) Whether, and to what extent, development activities are, or will be, stressing the habitat type.

(iv) The rarity of the habitat type.

(9) *Research and information needs.* Each FMP should contain recommendations, preferably in priority order, for research efforts

(10) *Review and revision of EFH components of FMPs.* Councils and NMFS should periodically review the EFH provisions of FMPs And revise or amend EFH provisions as warranted based on available information.

1.6 HABITAT AREAS OF PARTICULAR CONCERN

According to the language of the Interim Final Rule, EFH that is judged to be **particularly important** to **the long-term productivity of populations of one or more managed species, or to be particularly vulnerable to degradation, s**hould be identified as "habitat areas of particular concern" (HAPC) to help provide additional focus for conservation efforts. The following provisions of the Interim Final Rule provide guidance for habitat areas of particular concern:

(6) (ii) Cumulative impacts from fishing. In addressing the impacts of fishing on EFH, Councils should also consider the cumulative impacts of multiple fishing practices and non-fishing activities on EFH, especially, in habitat areas of particular concern. Habitats that are particularly vulnerable to specific fishing equipment types should be identified for possible designation as habitat areas of particular concern.

(9) Identification of habitat areas of particular concern. FMPs should identify habitat areas of particular concern within EFH. In determining whether a type, or area of EFH is a habitat area of particular concern, one or more of the following criteria must be met:

(i) The importance of the ecological function provided by the habitat.

(ii) The extent to which the habitat is sensitive to human-induced environmental degradation.

(iii) Whether, and to what extent, development activities are, or will be, stressing the habitat type.

(iv) The rarity of the habitat type.

The intent of the habitat areas of particular concern designation is to identify those areas that are known to be important to species which are in need of additional levels of protection from adverse impacts. Management implications do result from their identification. Designation of habitat areas of particular concern is intended to determine what areas within EFH should receive more of the Council's and NMFS' attention when providing comments on federal and state actions, and in establishing higher standards to protect and/or restore such habitat. Certain activities should not be located in areas identified as habitat areas of particular concern due to the risk to the habitat. Habitats that are at greater risk to impacts, either individual or cumulative, including impacts from fishing, may be appropriate for this classification. Habitats that are limited in nature or those that provide critical refugia (such as sanctuaries or preserves) may also be appropriate. General concurrences may be granted for activities within habitat areas of particular concern; however, greater scrutiny is necessary prior to approval of the general concurrence.

Following a review of the scientific literature for information on areas deserving special attention or species with particular habitat associations, the Council has designated an area on Georges Bank as an HAPC for juvenile Atlantic cod (Figure 6). Considering the unique habitat associations and requirements of Atlantic salmon, the Council has designated the habitat of eleven rivers in Maine as HAPCs for Atlantic salmon (Figure 7).

APPENDIX 11.2

THE CANADIAN FRAMEWORK AND GUIDELINES

The Canadian legal requirements to identify and protect critical habitat for specific species is new, and the technical approaches are still under development. Three key documents present the state of thinking about how Canada will provide the scientific support for the habitat requirements of SARA. The central document is the draft Technical Guidance document produced by an interdepartmental (DFO and Department of Environment, plus experts from the National Parks component of Canadian Heritage) working group. It outlines a general approach for all species, both terrestrial and aquatic. Extracts in section 3.2 come directly from the document, with material on population analysis and governance issues removed because they do not address the current Term of Reference. Two reports from DFO Workshops, one national and one regional (Pacific) consider implementation of the national, generic framework for marine and anadromous species. Extracts which provide likely direction for the technical aspects of critical habitat identification are presented from both workshop reports in sections 3.3 and 3.4.

1.7 LEGAL CONTEXT

Two Acts which DFO has to enforce provide definitions of fish habitat

Canada's Fisheries Act (1985) (Sec. 34) contains a definition of habitat for aquatic species that states "'fish habitat' means spawning grounds and nursery, rearing, food supply and migration areas on which fish depend directly or indirectly in order to carry out their life processes"

Section 34 of the Fisheries Act has been in force for three decades. Habitat managers have made tens of thousands of "permitting decisions" under Section 34, without any formal framework for identification and quantification of fish habitat. (A "permitting decision" is a decision whether or not to issue a permit for an activity which will alter aquatic habitat in which fish are known to occur. Under the "no-net-loss" habitat policy, undertakings likely to damage fish habitat must include mitigation measures such that either the damage is repaired fully, or alternative habitat of comparable value is created. Thus the title – a "no-net-loss" policy.)

The Species-at-Risk Act (SARA), implemented in 2003 has another definition of "critical habitat" for all species listed under the Act as:

"that habitat that is necessary for the survival or recovery of a listed wildlife species and that is identified as the species' critical habitat in the recovery strategy or in an action plan for the species" (Sec. 2)

SARA further specifies that:

- "recovery strategy must… include… (Sec. 41)
    - "an identification of the species' critical habitat, to the extent possible, based on the best available information…"
    - "a schedule of studies to identify critical habitat, where available information is inadequate"
- "action plan must include (Sec. 49)
    - "an identification of the species' critical habitat, to the extent possible, based on the best available information…"
    - "a statement of the measures that are proposed to be taken to protect the species' critical habitat"
    - "an identification of any portions of the species' critical habitat that have not been protected"

The recovery plan referred to above must be prepared within one year of legal listing for ENDANGERED, two years for THREATENED or EXTIRPATED. The Recovery Plan must include guidelines and timelines for development of the specific Action Plans.

Section 58 of SARA specifies protection for Critical Habitat which is highly proscriptive, such that designation of are area as critical habitat her significant social and economic consequences.

1.8 PRELIMINARY DRAFT TECHNICAL GUIDANCE FOR THE IDENTIFICATION OF CRITICAL HABITAT UNDER SARA (unpublished)

1.8.1 Preamble

- The federal Species at Risk Act (SARA), as passed by the House of Commons on June 11, 2002, recognizes in its preamble that the availability of habitat for species at risk is a key to their conservation. SARA requires that the 'critical habitat' of endangered, threatened, and extirpated species be identified in recovery strategies and action plans, and includes provisions to protect such habitat. **Critical habitat is defined in SARA as the habitat that is necessary for the survival or recovery of a listed wildlife species and that is identified as the species' critical habitat in the recovery strategy or in an action plan for the species**.

41. (1) If the competent minister determines that the recovery of the listed wildlife species is feasible, the recovery strategy must address the threats to the survival of the species identified by COSEWIC, **including any loss of habitat**, and must include

(*a*) a description of the species and its needs that is consistent with information provided by COSEWIC;

(*b*) an identification of the threats to the survival of the species and threats to its habitat that is consistent with information provided by COSEWIC and a description of the broad strategy to be taken to address those threats;

(*c*) **an identification of the species' critical habitat, to the extent possible, based on the best available information,** including the information provided by COSEWIC, and examples of activities that are likely to result in its destruction;

(*c*.1) a schedule of studies to identify critical habitat, where available information is inadequate;

1.8.2 Introduction

- What follows is a draft technical guide to the identification of critical habitat under SARA. The guide was prepared to describe and encourage a transparent, systematic, consistent, evidence-based, and ecologically sound approach to critical habitat identification. As required by SARA, those identifying critical habitat are expected to use the best available information respecting species status, distribution, abundance, threats, and habitat requirements in defining and locating critical habitat. As a matter of routine, evidence should be cited in support of statements or decisions made pertaining to the identification of critical habitat.

- The guide is designed to help recovery practitioners prepare evidence-based narrative descriptions and maps of critical habitat for species at risk - products expected meet the conditions for identification of critical habitat under SARA.

....

- The guide is written in a generic style with the intent that it be applicable to a wide range of taxa and ecological contexts. Four conceptual stages in the critical habitat identification process are outlined within the guide, these include; (1) Preparatory Stage, (2) Background Research Stage, (3) Analytical Research Stage, and (4) Review Stage. Within each stage a series of issues or tasks are outlined, not all of which may be practical or immediately doable for all species or in all situations. Some parts of the guide include iterative steps - places where step outcomes may encourage a reversal reconsideration of previous steps. A glossary of terms is included at the back of the guide.

- The guide is intended to be used in conjunction with other supporting interpretation and guiding documents –

1.8.3 Preparatory Stage

I. Advise Regional Species at Risk Recovery Co-ordinator(s)

- In situations where the geographic distributions of species overlap, efficiencies may be realized by coordinating efforts to identify critical habitat. Furthermore, knowledge of the coincident occurrence of critical habitat for two or more species may provide for opportunities to identify multi-species critical habitat.

41. (3) The competent minister may adopt a multi-species or an ecosystem approach when preparing the recovery strategy if he or she considers it appropriate to do so.

II. Document Key Decisions and Actions

• The chain of all key decisions and actions (including the underlying rationale) relating to the process and methodology of identifying critical habitat should be documented in writing or illustration (e.g., date-stamped draft maps)…

III. Cooperate and Consult

• SARA requires both cooperation (collaboration) and consultation (discussion) in the preparation of recovery strategies and action plans, components of which may include the identification of critical habitat IV. Create Species Recovery Database

• Many types of data, information, and analyses may be drawn upon throughout the critical habitat identification process. Coordinated and consistent management of these resources, in one or several associated (spatial) databases, will facilitate efficient data integration (across populations, species, regions) and easy comprehension of the data by generations of staff.

1.8.4 Background Research Stage

V. Review and Document Relevant Life History and Ecology

A. Outline primary biological needs

• ... A comprehensive summary of the species essential life history stages and key habitat needs … forces consideration of components of the species' life history that are required for survival or recovery.

B. Identify habitat attributes required to fulfill primary biological needs

• Characterizing the relationship between essential life history stages / key activities and habitat features forces preliminary consideration of the environmental context required for survival or recovery of the species. Habitat attributes that support biological needs of the species may include biotic (e.g., vegetation) and abiotic (e.g., soil) elements, as well as ecological processes (e.g., pollination, disturbance).

• Characterize any known or suspected temporal (e.g., daily, seasonal) differences in habitat use.

• Characterize any known or suspected demographic (e.g., age, sex) differences in habitat use.

• Be aware and account for the possibility that populations may have different types of habitat available to them and may use habitats in different ways.

• Describe any known distinguishing species-habitat associations - refer to application of national vegetation/natural community classification.

• This background knowledge will enable the preparation of a narrative conceptual model of the habitat mosaic (i.e., assembly of habitat features) required by the species for survival or recovery.

C. Summarize species' demography

• Prepare a comprehensive summary of the species' demographic structure (e.g., age, sex composition) and rates (e.g., survivorship, fecundity, recruitment, longevity), including error associated with these values. ….

D. Identify known and/or potential rate-limiting steps for population growth

• Characterizing any rate-limiting steps for population growth will help focus attention on the likely proximate causes of species decline and conservation status. Habitat attributes associated with rate-limiting demographic components of the species biology may be particularly important to account for when defining critical habitat for species survival or recovery. Clearly document all evidence supporting the importance of proposed rate-limiting steps.

E. Characterize species' genetic population structure

...

F. Identify and evaluate the threats facing the species

• Characterize the most important threats faced by the species - those which require mitigation in order to enable species survival or recovery. Highlight habitat-related threats since they are likely to be important determinants in the identification of critical habitat. Clearly document all evidence supporting the importance of proposed rate-limiting steps. Identify both proximate (change in vital rate) and ultimate (change in environment) threats. …

VI. Spatially Locate the Species

A. Map both current and historical distribution and abundance

• Describe and map the estimated number of individuals / populations / conservation units that currently exist and historically existed …Differences between current and historic estimates will help determine potential demographic and distribution objectives for species survival or recovery. Documenting the number of populations / conservation units will also provide a preliminary (minimal) estimate of the number critical habitat parcels that may need to be identified for species survival or recovery.

• Evaluate and describe any potential biases associated with different types of species occurrence data – biases that could skew an understanding of current and historic distribution and status.

• Identify and document areas that have not been adequately surveyed but which could potentially support populations of the species. Knowledge of such areas will contribute to the development of a comprehensive evaluation of the demographic and distribution objectives necessary for survival or recovery of the species.

• Look beyond Canada - describe and document current and historical populations that bridge the national borders …

VII. Spatially Locate the Species Habitat

A. Locate all known occupied habitat patches

• Acquire and overlay informative environmental (e.g., land cover, soils, topography) and socio-political (e.g., land tenure, municipal boundaries) data with species' occurrence data. These data will be required for subsequent habitat modeling. Patterns resolved by combining these data may result in an improved understanding of the species-habitat associations, species distribution limitations, and the location of known occupied habitat patches. Be aware of the fact that occurrence, environmental, and socio-political data are snapshots of information - confidence that such data represents what's on ground diminishes with increasing time before and after the data were captured.

B. Identify areas of potential habitat

• Identify unoccupied areas (and areas not known to be occupied) containing key habitat attributes. Under certain conditions, it may be appropriate to consider unoccupied areas as critical habitat . Some of those conditions might include, presence of appropriate habitat, presence of necessary ecological processes, and a significant potential for future occupation by the species (e.g., should adjacent population undergo range expansion).

• Identify areas where restoration is possible. Whether occupied or unoccupied at present, some areas may be worth considering as restorable - locations that could through management become quality habitat for the species.If necessary, such areas might be deemed critical habitat even before they are restored.

• Consider spatial / temporal shifts in habitat due to natural disturbance regimes. That habitats are dynamic – they change through space and time - suggests that critical habitat identification should be broad enough to encompass such shifts (non-habitat could become habitat and vice versa) and that they be subject to updating and revision over time. The spatial and temporal scales at which some habitats / natural communities operate are known and these could be used to calibrate the size and shape of critical habitat parcels and the frequency with which revisions should be considered.

• Model habitat - whether implicit and conceptual or explicit. ... If explicit models are used it will be necessary to describe the kind of model used - expert opinion, expert rules, habitat suitability index, etc Explicit models are particularly useful because they force one to articulate and codify ones assumptions and understanding of the system and they are repeatable. Where appropriated, use structured format to capture expert opinion – see http://www.srs .fs.fed.us/pubs/viewpub.jso?index=3467

• Model must incorporate uncertainties.

• Guidance for choosing the appropriate habitat model for predicting species' habitat:

Type of Data Available	**Appropriate Habitat Model**
Map(s) + experts	HIS
Locations only	Minimum convex polygon, alpha hulls, kernels
Location + maps	Above + climatic envelopes, multivariate distance methods, CCA
Locations used + random (available) locations	RSF
Presence – absence (used and unused locations)	Logistic regression, GLM, GAM
Abundance	GLM, GAM
Habitat dynamics	Landscape models (new in recovery context)
Not the first choice	Decision trees, neural networks, genetic algorithms

1.8.5 Analytical Research Stage

VIII. Establish Conservation Goal: Survival or Recovery

A. Determine if recovery is technically and biologically feasible

The determination must be based on the best available information…. [Technical detail on population issues, PVA, etc] Operational demographic and distribution objectives underlie the amount and configuration of critical habitat identified. Critical habitat should be necessary and sufficient to support the demographic and distribution objectives required for survival or recovery.

41. (1) If the competent minister determines that the recovery of the listed wildlife species is feasible, the recovery

strategy must address the threats to the survival of the species identified by COSEWIC, including any loss of habitat, and must include

(*d*) a statement of the population and distribution objectives that will assist the recovery and survival of the

species, and a general description of the research and management activities needed to meet those objectives;

IX. Determine Amount & Configuration of Critical Habitat Required to Achieve Goal - Derive Proposed Critical Habitat

A. Determine if adequate information exists to identify critical habitat (with confidence)

• Is there data for determination of: (1) the species' primary biological needs and key habitat attributes; (2) the threats to the species; (3) the general distribution as well as some precise locality information; (4) conceptual or quantitative model of species habitat and species demography; and (5) model validation?

• Require standard filter here. Can you answer - with an acceptable level of confidence - what habitat is thought necessary and sufficient for species survival or recovery? If yes, proceed with identification process. If no, defer identification to later date - but must outline a schedule of studies required to fill the knowledge gaps so that critical habitat eventually can be identified.

• Critical habitat may be identified incrementally. Adequate information may only exist to identify critical habitat parcels / units for certain populations or conservation units of the species. As additional information becomes available additional critical habitat units may be identified. Eventually, over time all critical habitat for species survival or recovery should gradually be identified.

B. Spatially explicit 'rules of thumb' or population modeling

• Employ 'rules of thumb' or models to illuminate the amount and distribution of critical habitat required to meet species survival or recovery. Explicit models are particularly useful because they force articulation and codification of system relationships and assumptions, they're able to accommodate and track potentially a complex series interacting factors, and they're repeatable.

• Where necessary, account for metapopulation and landscape-level issues.

• Where necessary, factor in habitat dynamics and ecological processes (succession, fire, flood).

• Make allowances for flexibility and redundancy.

• See guidance for model selection above. Model must incorporate uncertainties.

C. Validate model

• Enforce quality control on the modeling protocol - models will only be deemed acceptable if validation data have been collected and the model gives sufficiently low false positive and false negative results R:/Critical Habitat/Guidelines/Critical Habitat/Tech Guide for ID of Critical Habitat – V1.doc

X. Quality Control and Peer Review

• Recovery strategies and action plans will be subjected to both administrative (QC) and (external) scientific peer reviews. …

• Critical review of the proposed critical habitat in draft recovery strategies and action plans may demand specialized skill sets (e.g., population and habitat modeling), requiring this material undergo a separate review process.

• Consider and respond to comments and recommendations arising from reviews – document process

• Revise proposed critical habitat as necessary – document process

XI. Post Recovery Strategy / Action Plan (including Proposed Critical Habitat) to SARA Public Registry

• Recovery Strategy or Action Plan (including proposed critical habitat) subject to 60 day public comment period

XII. Finalized Recovery Strategy / Action Plan Posted to SARA Public Registry - Critical Habitat Legally Identified

XIII. Critical Habitat Monitoring

• Design and implement monitoring studies to determine adequacy of identified critical habitat. Is habitat being maintained in appropriate conditions? Is there evidence of habitat utilization? Are population targets are being met?

• Collect appropriate data to improve predictions of PVA and estimates of critical habitat required to meet species survival or recovery.

XIV. Review & Revise Critical Habitat

• As more information becomes available, revise critical habitat accordingly (increase or decrease the amount of critical habitat)

1.8.6 DRAFT GLOSSARY OF TERMS ASSOCIATED WITH THE IDENTIFICATION OF CRITICAL HABITAT (Also Annex 3)

Accuracy: The closeness of a measurement to the actual value of the variable being measured; the closeness of an object's representation to its actual location.

COSEWIC: Committee on the Status of Endangered Wildlife in Canada – A committee of experts that assesses and designates which wild species are in some danger of disappearing from Canada.

Critical habitat: "The habitat that is necessary for the survival or recovery of a listed wildlife species and that is identified as the species' critical habitat in the recovery strategy or in an action plan for the species" [s.2(1)].

Critical habitat identification process: The procedure by which a species' critical habitat is described, quantified, and located geographically. Guidelines are being developed to promote an identification process that is consistent, transparent, and science-based. The recommended steps to identify critical habitat for a species include critical habitat description and delineation.

Critical habitat description: A narrative account of the key habitat attributes that support a species life history requirements and a quantitative estimate of the amount and configuration of habitat needed for the survival or recovery of the species.

Critical habitat delineation: A map and/or narrative account that depicts specific geographic location(s) of critical habitat for a species.

Endangered species: "A wildlife species that is facing imminent extirpation or extinction" [s.2(1)].

Ephemeral habitat: Habitat that exists in any particular location for a brief amount of time (e.g., early successional habitats, vernal pools, areas of shallow snow in the arctic).

Habitat: "…the area or type of site where an individual or wildlife species naturally occurs or depends on directly or indirectly in order to carry out its life processes or formerly occurred and has the potential to be reintroduced" [s.2(1)].

Habitat composition: The presence and amount of each type of a species' habitat within a landscape. Describes the variety and abundance of habitats used by a species.

Habitat configuration: The spatial arrangement of a species' habitat within a landscape. Describes characteristics such as habitat patch size and shape (and therefore amount of edge/core area), and isolation/connectivity between habitat patches.

Individual: "An individual of a wildlife species, whether living or dead, at any developmental stage and includes larvae, embryos, eggs, sperm, seeds, pollen, spores and asexual propagules." [s.2(1)].

Key habitat attributes: The biotic and abiotic habitat features and ecological processes thought necessary to support some or all of a species' life history requirements.

Population: A group of organisms of one species, occupying a defined area and usually isolated to some degree (geographically or functionally) from other similar groups.

Precision: The closeness to each other of repeated measurements of a variable; the smallest difference between adjacent positions that can be recorded.

Recovery: Downlisting in species status to Special Concern or Not at Risk. Downlisting may result from the

establishment of a population(s) with the capacity for long-term viability or self-sustainability.

achievement of recovery objectives and targets outlined in the recovery strategy, or natural changes in the environment that benefit the species.

Residence: “A dwelling-place, such as a den, nest or other similar area or place that is occupied or habitually occupied by one or more individuals during all or part of their life cycles, including breeding, rearing, staging, wintering, feeding or hibernating” [s.2(1)].

Resilience: The capacity of an ecosystem to tolerate disturbance without collapsing into a qualitatively different state that is controlled by a different set of processes. A resilient ecosystem can withstand shocks and rebuild itself when necessary.

Scale: The spatial or temporal dimensions used to measure, study, and represent objects and processes. Scale is defined by two characteristics, extent and grain.

***Extent*:** The area over which observations are made; the duration of those observations.

***Grain*:** The smallest resolvable unit of area or time.

Survival: Maintenance of existing population(s) as indicated by no net loss in abundance and distribution.

Threatened species: “A wildlife species that is likely to become an endangered species if nothing is done to reverse the factors leading to its extirpation or extinction” [s.2(1)].

Viability: e.g., 90% probability of persistence over 100 years or a time horizon based on species generation time

Wildlife species: “A species, subspecies, variety, or biologically distinct population of animal, plant, or other organism, other than a bacterium or virus, that is wild by nature and (a) is native to Canada; or (b) has extended its range into Canada without human intervention and has been present in Canada for at least 50 years” [s.2(1)]. [1]Definitions in quotes and Italics are directly from SARA. All others are proposed definitions, and are subject to change.

1.9 National DFO Workshop on Quantifying Critical Habitat for Aquatic Species at Risk

(CSAS Proceedings 2003/12)

As the protection of critical habitat is likely to be controversial, science-based methods for the identification and quantification of critical habitat are essential. There will be a need to demonstrate cause and effect linkages between specific habitats and species survival before critical habitat is designated.

1.9.1 Methods for measuring critical habitat:

Several quantitative science-based methods for measuring critical habitat [are possible]. The approach and methods depend on the level of information available for the at-risk species (Table 3.3.1). Five Information Levels were identified, ranging from 0 (know nothing) to 4 (knowledge of productivity). Population targets for recovery will be qualitative and critical habitat targets will be broad in scope and geographic area if the information level is low. If the information level is high, population targets will be quantitative and critical habitat will be narrowly defined. In addition to helping determine the appropriate methods for determining CH, the matrix can be used to identify data gaps.

If no information is available on habitat needs for the species at risk the priority would be to conduct a comprehensive search of relevant data and to conduct basic research on biology and habitat requirements. Approaches for operating with limited data included the use of surrogate species, the Traditional Ecological Knowledge (TEK) method, and inference. Guidance will be needed to determine a standard for ‘extent possible’ and the adequacy of or minimum information needed to designate critical habitat.

Knowledge of presence-absence, distribution and migration corridors, the next information level, can be generated from visual observation, field survey and tagging data (e.g., marine mammals) and by TEK (Table 3.3.2). Presence-absence data can be used to generate cursory maps of species distribution, range and habitat. Designated Marine Protected Areas (MPA, Oceans Act) can be used as a tool for protecting habitat over large areas and can be beneficial for several target species. The assumption is that MPA's include critical habitat for certain target species.

Knowledge of population density, life stage growth and survival rates and productivity will allow the application of increasingly detailed models for identifying critical habitat (information levels 2 to 4 in Tables 3.3.1 and 3.3.2). Density-fish size regression models can be used to determine area-per-individual (API) and hence the habitat area requirements of a population, if density is known for different fish sizes and habitats. In many instances, habitat-dependent functions are poorly known. Further research is needed, using existing or new methods to link fish density and habitat (Probability Density Function [PDF], Ideal Free Distribution [IFD] and other models). Knowledge of life-stage specific rates and functional linkages with habitat will allow the use of sophisticated Meta-population or Population Viability Analysis (PVA) to determine critical habitat (Table 3.3.2). PVA models can only be applied to 'data-rich' species (including knowledge of density-dependent survival), and they need to be validated for each species. At-risk species with level 4 information (productivity) are rare. Long-term data on fish productivity and demonstrated linkages with habitat can be used to quantify critical habitat with reasonable confidence.

Additional generic methods common to both freshwater and marine environments were: 1) life history approaches (e.g., models specified previously in Table xxx with information level 3 or higher); 2) mapping techniques, ranging from simple occurrence data, intermediate GIS approaches using local knowledge (TEK) data, to more sophisticated habitat mapping (e.g., hydroacoustic multibeam) with associated ground-verification of habitat types and species use; 3) modelling techniques, from simple to sophisticated depending on the amount of data, with sensitivity analysis to identify potential life stage or habitat bottlenecks; 4) experimental approaches; 5) micro-habitat approaches, but with the reservation that it is often difficult to link the results to a population level scale; and 6) behavioural approaches (e.g., using acoustic tags).

Similarities between freshwater and marine methods were: both require hierarchical approaches depending on available information: as knowledge increases, uncertainty decreases and CH can be defined more precisely; both involve assessment across scales and ecotones (landscape, estuaries, transition zones); the need to couple life history and habitat; and a common focus on modelling. Dissimilarities are also apparent: marine environments are larger scale that requiring remote sensing tools and dynamic habitat (i.e., habitat associated with phenomena rather than specific places) may be a more frequent feature of marine systems (upwelling, ice edge) than freshwater systems. Tracking dynamic habitat is a challenge.

1.9.2 Process for identifying critical habitat -

A listing of operational guidelines for assessing critical habitat was presented in a flow chart (Figure 3). The four steps in the chart, assessment, decision to designate regardless of cost, decision based on cost, and defer, emphasize that the process is iterative. These guidelines were preliminary as workshop discussion did not advance far enough to provide details on several key issues.

Uncertainty and precaution should be explicit at all stages of the process. Recovery management activities need to be monitored to evaluate effectiveness; performance indicators and adaptive management provisions should be part of the plan. Science peer review of all aspects of the recovery plan would be consistent with a precautionary approach and would lead to advice for refinements as more information became available.

Key implementation issues were apparent. A broad operational definition of critical habitat is needed for both Science and managers, that must include physical, biological and water quality aspects, as all affect survival and population viability. Habitat can be either spatially static (specific geographic area) or dynamic (gyres, upwelling), emphasizing that recovery plans must be flexible and encompassing

At the outset, survival and recovery goals in recovery plans should be quantitative. Therefore both population (target size; survival parameters) and habitat goals (geographic range in area; habitat quality requirements) need to be defined. Basic life stage information for each species and habitat use by each stage is a prerequisite for identifying critical habitat. Knowledge gaps will be a challenge for many species, and therefore dealing with uncertainty and adopting a precautionary approach will be paramount, particularly for data-poor species.

1.9.3 Research Needs for identifying critical habitats

Five basic research priorities were identified that parallel the information needs. Increased knowledge of functional linkages between habitat and population dynamics (survival, growth, recruitment) is a research priority for all at-risk species, even for well-studied species. Survival and recovery goals need to be determined by Science, including the determination of a minimum viable population size. Knowledge of carrying capacity and production is needed in addition to the basic life stage information mentioned above. Adopting a life-stage approach for identifying CH was advocated by all participants. The initial identification of critical habitat by Science will be interim, acknowledging that it will be an iterative process that requires refinement as the knowledge level increases. The needs for monitoring, habitat mapping, more case history studies, data based development, and for gaining a better understanding and predictive ability for the abundance – habitat relationships of marine species were all highlighted as needs for consolidating a framework for identification of critical habitat.

Table 3.3.1. Hierarchy of Information Level and the corresponding gradient in detail for population targets and critical habitat targets for at-risk species.

Information Level	**Life History Stage**			**Habitat or ecosystem features**			**Model(s)**	**Population Target**	**Critical Habitat Target**
	a	b	c	i	ii	iii			
0 - Know nothing							TEK, surrogate species, inference	Qualitative	Broad in scope & area; precautionary
1 - Presence/ absence data							Hanski model[1]; cursory mapping		
2 - Population density data							API, stock assessment techniques		
3 – Life stage process rates (survival, growth, fecundity)							PVA, Meta-population; others (as applicable to species, available information)		
4 -Productivity							Population – habitat capacity models	Quantitative	Narrow, well-defined

[1]Hanski, I. 1982. Dynamics of regional distribution: the core and satellite species hypothesis. Oikos 38:210-221.

Table 3.3.2. Summary of proposed methods for identifying critical habitat as discussed at the workshop. Methods are grouped by Information Level (see Table 2).

Information level	Species	Methods	Data used	Importance
0 Nothing				
1 P/A	Marine mammals	Field observation	Tagging and visual observation	Unpredictable spatially dynamic habitat leads to large scale and high uncertainty when mapping CH
1	Multispecies, multitrophic	Traditional Ecological Knowledge (TEK) ; GIS	Questionnaire	Areas of high diversity and productivity were tractable and could be mapped.
1	Beluga whale	Field observation; tissue analysis of carcasses and biopsies	Systematic surveys; photo ID; VHF radio; genetic; telemetry; fatty acids and stable isotopes for diet.	Habitats are sex and age dependent; boundaries are not fixed but vary with availability of resources.
1	Multispecies, marine	MPA's, habitat mapping.	Multibeam acoustic bathymetry, video	MPA (Oceans Act) as a tool for protecting habitat.
2 Density	Surf clams; haddock	PDF models	Maps of seabed characteristics from acoustic surveys. Population distribution relative to habitat type.	Novel PDF and habitat-dependence functions are needed; on-going research.
2	Cod	IFD theoretical framework; density-fish size relationship	Time series of trawl catches including years of high and low abundance (density, condition, length-at-age).	Knowledge of important habitat when population is at a low level. Habitat use was dependent on abundance. Habitat use at low abundance indicates critical habitat.

Information level	Species	Methods	Data used	Importance
2	Atlantic sturgeon	BACI	Occurrence by life stage (spawning or pre-spawning, feeding areas). Acoustic and trawl determination of density.	Habitat categorization and mapping is a key component.
2	Multispecies; marine littoral	Various (statistical); collaborative field research	Field research, includes survival and growth rates of young fish	Both biotic (predators) and physical structure is important for survival; multispecies dependency on critical habitat areas.
2	Capelin	Field research	Knowledge of site-specific spawning substrate and egg viability	Life stage bottleneck at spawning, as year-class strength is determined during egg incubation.
2	Yellow perch and pumpkinseed	Density-fish size regression	Electrofishing survey in littoral Great Lakes	Area per fish is habitat dependent but quantifiable; broadly applicable model.
3 Process rates (survival, growth, fecundity)	Generic	Habitat-based Meta-population model	Functional links (d-d survival) between life history stages and habitat	Both amount and configuration of habitat is important.
3	Chinook salmon; canary rockfish	PVA model	Life stage specific population vital rates and habitat needs.	Sensitivity analysis to determine the response of population growth to changes in vital rates, which in turn are linked to habitat characteristics.
3	Lake charr; deepwater sculpin; generic	Area per individual (API) model	Three life stages; literature population rates, density-size relationships by life stage.	Indicates viability is related to rearing and adult habitat moreso than spawning. API model is broadly applicable.

Information level	Species	Methods	Data used	Importance
3	Atlantic salmon	Meso habitat models (gradient) in FW; thermal and migration mapping in marine.	Historical data; freshwater, juvenile abundance; marine, mark-recapture data	When marine survival is low, all FW habitat is critical for population survival; PVA needed.
3	White sturgeon, abalone	Habitat mapping; age-structured population model	Literature, includes process rates for white sturgeon	Appropriate scale for mapping and other criteria for determining CH are needed.
4 Productivity	Coho salmon	Population dynamic model;metapopulation dynamics; regression.	Existing life stage specific data. Temporal trends in smolt production and knowledge of habitat types; distribution and habitat use at low population abundance.	Simple population dynamic model; knowledge of important habitat when population is at a low level.

Figure 3.3.1. Flow chart for listing operational guidelines for assessing critical habitat .

1. **Step 1 Assessment:**
   - Recovery team required to assess habitat necessary for recovery.
   - Consider threats (as identified by COSEWIC at time of listing).
   - Undertake quantitative assessment (modelling):
     - comprehensive search for relevant data;
     - consider surrogate species, allometric relationships;
     - identify sensitive parameters (assumptions) and thus, information gaps.

2. **Step 2 Decision:**
   - Is a "critical" designation necessary, regardless of cost?
     - Standard for type 1 vs type 2 critical values must be identified by policy or precedent;
     - Accepting this burden of proof, can some habitat be identified as necessary for recovery?

   If yes, then designate as critical habitat;

   If no, then go to Step 3.

3. **Step 3 Decision continued:**
   - Is a "critical" designation worthwhile?
     - Could habitat protection reasonably be expected to improve viability?
     - Consider probable benefits of habitat protection;
     - Consider costs of designation.
   - Use "decision analysis" (including uncertainty and stakeholder input), to determine whether expected value of designation outweighs costs.

   If yes, then designate;

   If no, then go to Step 4.

4. **Step 4 Defer designation of critical habitat:**
   - Pursue recovery by other means;
   - Conduct additional research on habitat requirements;
   - Re-assess ***(go back to step 1... when ready).***

1.10 Proceedings of the DFO Pacific Region Critical Habitat Workshop 26-28 March 2003, Nanaimo, BC. CSAS Proceedings 2003/10

Many of the issues already addressed were treated at this workshop too. A few new themes or new perspectives on perspectives already addressed in other framework documents arose at this meeting. They included:

There may often be a number of scenario options that are combinations of the above that have "equal science merit", and the scenario ultimately chosen for the identification of critical habitat may to a large part be determined by the social and economic costs associated with these "equal science" options. However, often options may not really be equal if different perspectives, or optics, are considered. For example, we must also consider other species, and the potential synergies of effort and "umbrella-type" habitat protection. When there are species-at-risk that utilize similar habitats (e.g., sea otters and abalone, white sturgeon and salmon), a joint review of proposed critical habitat designations in relevant recovery plans may be desirable. This is often likely to be an issue, as many listed species are at risk because their ecosystems or habitats are severely threatened.

In comparison of freshwater and marine species' habitat threats, and interpreting habitat to also include water quality parameters, the concern was repeatedly brought up as to the importance of upland activities and threats, such as runoff, sedimentation, changes in vegetative cover, etc., in determining the suitability of freshwater habitats. Ephemeral habitats, such as seasonally flooded habitat, are also increasingly being recognised as important habitat for some species-at-risk, and so there are likely to be temporal components to some critical habitat designations. It was also felt that species in freshwater habitats were generally more vulnerable to extinction than most of those in the marine environment because of the linear nature of freshwater systems, and thus have greater susceptibility to fragmentation; an often smaller scale (less buffering); and possibly greater species isolation (analogous to island biogeography). The extensive ranges, pelagic distribution and high mobility in either the larval or adult stages of many marine and anadromous species, such as cetaceans, leatherback turtles and salmon, pose other unique challenges to critical habitat definition.

In separating critical habitat from essential habitat [there was]… recognition that for most species, there is a lot of mediocre habitat and a lesser amount of higher quality habitat. Conceptually, assuming that habitat quality can be linked to the viability of a species, there is likely a quality/quantity habitat trade-off (more poorer habitat may be equal to a lesser amount of higher quality habitat), thus indicating that there may be multiple spatial configurations that a recovery team can consider that achieve the same population viability. This approach may provide the opportunity to identify a number of options in a recovery strategy.

Conceptual model For many species there are likely different configurations of habitat that are equally critical, that is, provide equal viability. Two very different options to improve viability by designating critical habitat might be:

- to designate critical habitat in those areas that are most likely to be impacted by a threat, thereby eliminating the most recognized threats , or

- to designate critical habitat more extensively in other areas that are not yet threatened, thereby preserving these areas. This configuration may be more affordable socially and economically than option #1.

Assuming n the HSI can be linked to the life history parameters that determine viability, there is likely a trade-off between the quality and quantity of habitat such that the viability conferred by protecting more poorer habitat may be equal to that from protecting a lesser amount of higher quality habitat. Thus, there may be multiple spatial configurations of habitat that achieve population viability.

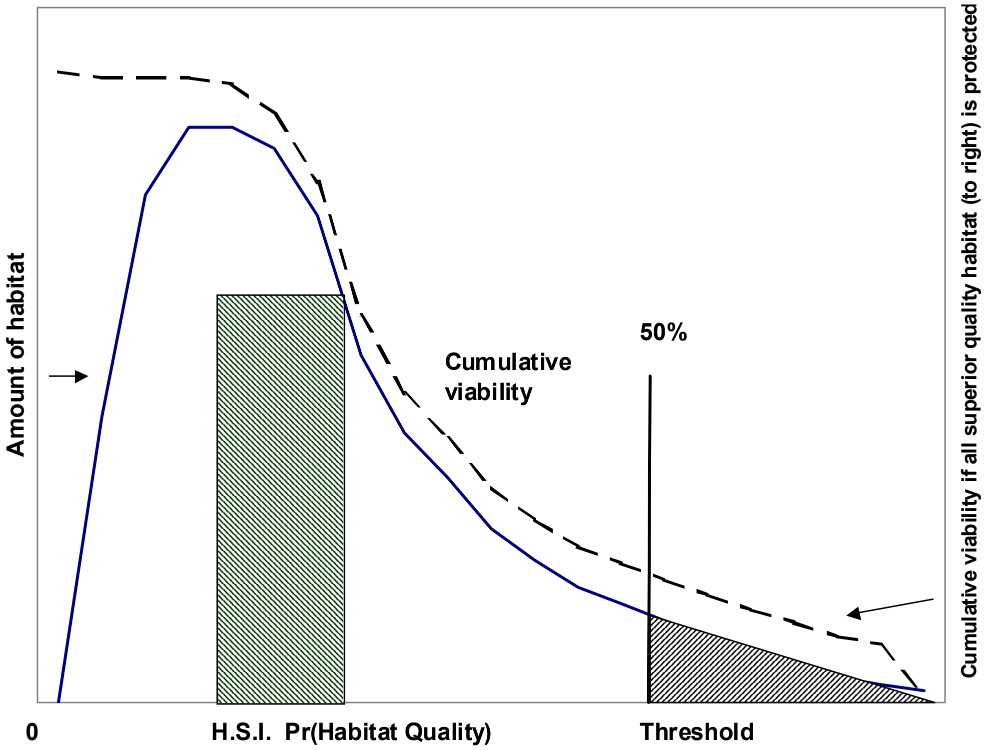


Figure 3.4.1*:* This figure is a conceptual model of the description above. The probability of stream habitat quality ranges from 0 (poor quality) to 1 (high quality). The figure depicts that there is a lot of poor quality habitat and little high quality habitat. This scenario illustrates a quality/quantity habitat trade-off where protecting a large amount of poor quality habitat (thatched rectangle) is likely equivalent to a small amount of high quality habitat (thatched triangle).

APPENDIX 11.3

A COMPARISON OF US AND CANADIAN GLOSSARY TERMS

Comparison of definitions of terms which are in common (or nearly so) between the US and Canadian frameworks.

In the core definition of what comprises "essential fish habitat' (US) and "critical habitat" (Canada), the definitions use different words. The US specifies "necessary for *spawning, feeding, breeding, or growth to maturity"* [italics added, whereas the Canadian specifies "necessary for survival or recovery *and* [italics added] is identified as the species' critical habitat in the recovery strategy or in the action plan for the species." Biologically the definitions are mutually compatible. However there is major difference in practice, in that US definition merely gives the biological conditions, such that by implication all habitat meeting those conditions should be treated as "essential". In the Canadian definition, the additional language after the "and" clearly guides that whatever biological conditions are met by a tract of habitat, it is not "critical" unless a species recovery plan or action plan explicitly states that it is "critical". This difference has profound implications for practice.

The difference in core definitions is accommodated in the choice of other terms to define explicitly in the remaining lists by the two countries. In the US list of definitions, five related to defining what habitat and ecological considerations should be taken into account in particular ways; waters, substrate, ecosystem, and healthy ecosystem. Four terms define components of the *governance process* by which EFH is designated: Fishery Management Plan, EFH Assessment, EFH Conservation Recommendation; and Finding. Two terms explain how the the governance processes are to relate to the ecological processes; Necessary and Adverse Effect. In the Canadian list of definitions there are four terms which relate to the *governance process*; COSEWIC, Critical Habitat Identification Process, Critical Habitat Description, and Critical Habitat Delineation. Seventeen terms relating to scientific aspects of identifying critical habitat are included in the list of definitions, including statistical terms (accuracy, precision) as well as ecological ones (endangered species, ephemeral habitat, population, recovery, survival, etc.).

Overall the definitions are not incompatible, but they are clearly intended to serve different functions. In the US, it is important to be clear about what steps must be followed in defining critical habitat. What happened at each step is not specified in detail, beyond the overall reference to "best available science" being used in those processes. In Canada, it is enough to state that critical habitat is that habitat which is called critical in a recovery or action plan. However the Canadian guidelines give importance to guiding the nature of the science actions and choices made in the unspecified ways that the critical habitat designations end up in the recovery and action plans.

In terms of applying the US and Canadian experience to guidance for implementing the habitat-related provisions of the Species and Habitats Directive and the Water Framework Directive, definitions from either country's list might be borrowed and details adapted as needed. However, the differences in these lists of definitions, as with the provisions in the overall frameworks of the two countries, make very clear that although the identification of critical or essential or important or even just rare habitats is completely a science-based exercise, the guidelines need to be matched to the strengths and weaknesses of the governance system in which the habitats will be identified and plans for their protection will be developed and implemented.

12 PREPARATIONS TO SUMMARISE THE EFFECTS OF FISHING ON NORTH SEA BIOTA FOR THE PERIOD 2000-2004, AND ANY TRENDS IN THESE EFFECTS OVER THE RECENT DECADES

j) start preparations to summarise the effects of fishing on North Sea biota for the period 2000-2004, and any trends in these effects over the recent decades.

12.1 Introduction

As requested, we have restricted ourselves to a consideration of the datasets and information that would be required to undertake an assessment of the effects of fishing on North Sea biota, and have suggested ways in which these could be assembled. This includes an assessment of both the requirements related to the effects of fishing and to the effort statistics needed to determine the impact. We have used four broad ecosystem categories to help identify appropriate datasets for the effects, and follow these with an assessment of the types of data required for different ecosystem components. In addition to the direct effects of fishing on target and non-target populations and on habitats, we have made it clear where we feel that work is required on the indirect effects, and which will therefore require input from groups with the relevant expertise.

The interpretation of previous work of this group (ICES, 2003), suggests four important areas that must be covered by management in addressing the effects of fishing across the ecosystem. These are:

1. Direct mortality of commercially exploited populations,

2. The protection of species that are ecologically dependent on other species affected by fisheries (i.e., a need of a species for a particular aspect of the habitat (physical, chemical or biological) or through ecological linkages within the food web. This includes both vertical links – species and their predators and species and their resources, and horizontal interactions such as competition for food or space),

3. Impacts of fisheries on non-target species and ecologically important and sensitive habitats, and protection of habitats that are 'essential' or at risk,

4. Preservation of genetic diversity to maintain adaptability of populations in the face of environmental change, future utility of genetic resources for medical and other purposes, changes in life history traits (e.g., age and size at maturation, growth) and changes in behaviour (e.g., timing of spawning).

12.2 Direct and indirect effects

We feel that it is important to distinguish between the direct effects of fishing (mortality and alteration of habitat) and the consequences of these effects on the ecosystem components (indirect effects), which ultimately depend on the interaction of the direct effects with other factors important in structuring the component (e.g. other biotic or abiotic drivers). For example, of the four broad ecosystem categories detailed above, 1 explicitly includes direct effects, 2 indirect effects and 3 and 4 a mixture of direct and indirect effects. The significance of making this distinction is that there may be potential to incorporate the direct effects in increasingly realistic indices of fishing disturbance based on effort statistics. Within WGECO we feel that there is the relevant experience to develop ecological indices of fishing disturbance (the direct effects) based on effort statistics and in Section 12.4 an example of the work currently being undertaken, with relevance to the data needed to take this forward, is given. The indirect effects are described primarily with the purpose of aiding the interpretation of trends and status of individual ecosystem components within the REGNS framework. The potential for WGECO to contribute to the determination of trend and status in these indirect effects is discussed in the concluding statement.

12.3 Ecosystem components

The following ecosystem components have been used in section 9 as the basis for a coherent and integrated management scheme. They can be used equally well here to identify the major components of the ecosystem that will be vulnerable to the effects of fishing, and that will require investigation. For the purpose of this exercise a number of the components have been treated together to reflect the overlap in effects of fishing and in the requirement for data.

12.4 Mammals/birds

Based on previous reviews by WGECO of the effects of fishing to seabird and marine mammal populations (ICES, 2000; 2003) the following summary is given. The main direct effects are the mortality sustained by individuals as bycatch. Bycatch of marine mammals is most prevalent in fixed gears such as set nets, drift nets and gillnets, although the implications of this mortality at the population level are uncertain at the North Sea scale. Seabirds also suffer considerable levels of mortality as bycatch in set net fisheries in addition to that sustained in long-line fisheries. We acknowledge that at present none of these fisheries constitute a significant proportion of the overall North Sea effort, however in some more localised areas the implications of bycatch on seabird and marine mammal populations may be greater. To be able to assess the significance of the level of bycatch mortality to seabirds and marine mammals it is important to have access to reliable information on local population size, distributions and fishing mortality of each species.

Much of the other work undertaken in reviewing fisheries interactions with seabirds and marine mammals is in reference to the effect of food subsidies from the discarding process on scavenging populations (ICES 2000, 2002, 2003). Although there is clear evidence that significant proportions of discards may be taken by seabirds, the implications at the population level have to be interpreted in reference to other factors important in driving variability in these species (ICES, 2003). The consequences of potential changes in scavenging populations at the population and community level should be considered.

There is little evidence of widespread direct competition between fisheries and marine mammals/seabirds in the North Sea. Indirect effects are also difficult to elucidate (see Section 4 for an example). Due to the long-lived / low breeding productivity characteristics of seabirds and marine mammals, responses to changes in fish populations e.g. size spectra, caused by fishing may be delayed and prolonged. Evaluating or predicting these effects over short time-frames will be very difficult.

To address the impacts of fishing on the seabird and marine mammal populations of the North Sea, the following data will be required:

- Species abundance and (offshore) distribution, and variation in this by season
- Knowledge of diet by species
- Age structure of the population

Due to the migratory nature of marine mammal and seabird species it will be important to assess the range and distribution of individual populations in order to be able to consider overlap of fishing effort in those fisheries that have direct and indirect effects on these components.

In an ideal world, this data should cover the years 2000-2004, but such data as exist are patchy and incomplete. Better data exist for the past 25 years, if aggregated. This of course loses potentially important variance between years. Resolution of the aggregated data can be at any scale (all records of seabirds and marine mammals are fully geo-referenced). Trend data is available for breeding population sizes of seabirds and seals, but not for cetaceans. Diet information is heavily biased towards breeding seasons/haulout periods, or from dead individuals found on beaches. Offshore foods are poorly known for all species. Direct mortality in fishing gear is generally poorly described – ICES has already recommended independent observer schemes as the only way of reliably quantifying such mortality. Very few observer programmes have ever been established

12.4.1 Plankton

To the best of our knowledge there are no significant effects of fishing on plankton (phytoplankton or zooplankton). While we acknowledge that change in the population size and distribution of plankton feeding members of the other components may itself be a consequence of fishing effects, there is no known evidence that this is a significant driver in the structuring of North Sea plankton. As such, there is no explicit requirement for data on plankton in order to establish the direct effects of fishing on North Sea biota. However, as plankton are a key driver of some of the other ecosystem components there will be a data requirement in interpreting overall trends in these.

12.4.2 Habitats/Nutrients

In line with the definition given in Section 9, we refer to habitats here as the physical and chemical environment. The effects of fishing on habitat are related to the physical disturbance by bottom gears in contact with the seafloor. In summary these include removal of large physical features, reduction in structural biota and a reduction in complexity of habitat structure (leading to increased homogeneity) (ICES 2002, 2003). The extent of these changes is related to the types of fishing gear being used and the initial level of complexity in both physical and biogenic structure (See Auster & Langton, 1999 and Johnson, 2002 for review). Much of the work that has already been undertaken on alteration of habitat in the towpath has taken place in areas other than the North Sea (See review in ICES, 2002,2003). Given that many of the habitats studied previously are of high structural complexity, we suggest that the comparability with effects in the North Sea is likely to be low.

At the same time, the resuspension of sediments that occur during the trawling process may be associated with the release of contaminants and heavy metals that have previously been stabilised in the sediments. The effects of resuspension events on nutrient fluxes have also been studied, but again, most of the available literature is not from the North Sea. We are aware that work is currently being undertaken in the Southern North Sea and consider that the significance of the effects of trawling on nutrient cycling and localised fluxes must be addressed in North Sea studies.

As has been stressed before (ICES, 2002, 2003), the most important requirement in order to address the impacts of fishing on habitats, is for widescale mapping at a scale that is comparable with the resolution of fishing effort data available.

12.4.3 Benthos

Many of the direct and indirect effects of fishing to benthos are comparable with those of fish communities (Section 12.3.5). Benthic invertebrates suffer mortality both in the gears and in the towpath of the gear. Large size, fragile morphology and low mobility have all been associated with increased vulnerability (ICES 2000, 2002, 2003). Thus within communities, selective mortality is likely to lead to reduced abundance of large species with low intrinsic rates of increase, and dominance of smaller species with higher intrinsic rates of increase. Changes in size distribution have also recently been described for a number of areas in the North Sea (Jennings *et al*., 2001; Duplisea *et al*., 2002) and the implications of this on secondary productivity discussed. The generality of these findings should be examined across a greater area. The interaction between scavenging populations and the increases in moribund material in the towpath of the gear has been described in a number of studies in the Southern North Sea and Irish Sea but the implications of this at the population level and the scale of the North Sea are unknown.

The importance of the physical features of habitats in determining the community structure of benthos is well documented (Duineveld *et al*., 1991; Hall, 1994). We therefore stress the importance of the overlap between effects of fishing on physical habitat and the effects on the resident benthic communities. The availability of well-defined habitat maps as requested in Section 12.3.3, will significantly improve our ability to assess the effects of fishing on benthos.

To address the impacts of fishing on the benthos of the North Sea, the following data will be required:

- Species identity
- Species abundance
- Species individual biomass
- Total biomass of the community
- Sediment characteristics
- Species abundance at length

Ideally, the spatial extent of the samples should be from the entire North Sea, preferably from few extensive surveys rather than many regional sampling programmes to ensure consistency of sampling efficiency. Ideally for benthic invertebrates, samples should be at a higher spatial resolution than the ICES rectangle but we acknowledge the difficulties in accessing this data even at the scale of the rectangle.

Data should be disaggregated to the greatest extent possible, but we acknowledge that there are no data at the scale of the North Sea covering the entire period 2000-04. The assessment of the state may therefore have to be based on a more restricted number of years. Where possible selected time-series from at least the early 1980s should be made available to derive information on trends in these effects 'over the recent decades'.

12.4.4 Fish

We consider commercially targeted fish and shellfish within this section as data required to summarise status and trend in these are the same as those required for fish communities. This summary of properties of fish populations and fish communities builds on previous work undertaken by WGECO (ICES 2001, 2002), examining the sensitivity of demersal populations to fishing activity. Within populations, selective fishing mortality is expected to lead to changes in growth rate and reductions in age and size at maturity. Within communities, selective mortality leads to reduced abundance of large species with low intrinsic rates of increase, and dominance of smaller species with higher intrinsic rates of increase. Variation in life history characteristics within populations is much lower than among species in a community, and thus selective effects of fishing are most readily observed at the community level.

Changes in size distributions in response to exploitation have also been described. As fishing mortality increases, mean size of individuals in the community drops, and small individuals form a larger proportion of the biomass. Consequently, the (negative) slope of size spectra generally became steeper while the intercept increased. Size-based approaches such as these provide an effective way of describing gross community responses to fishing, but the structure of the size spectrum and the observed response is based on a combination of factors including: (1) differential vulnerability of larger species; (2) within-population changes in mean size; (3) genetic changes in life history; and (4) predator-prey relationships within the community.

Although many of these relationships have been studied and are the subject of ongoing research, weaknesses have included limited reference to quantitative differences in fishing effort when making temporal and spatial comparisons among fish communities, and an inadequate understanding of the effects of sampling gears on the properties of size-spectra and life-history based metrics. In addition, there has been limited consideration of differential responses of species groups to patterns of change in the community, and the sensitivity of the community trophic structure to fishing impacts.

To address the impacts of fishing on the fish populations in the North Sea, the following data will be required:

- Species identity
- Species abundance at length
- Species individual weight at length
- Species individual age at length
- Species individual maturity at length
- Representative stomach sampling for trophic analysis

The spatial extent of the samples should be from the entire North Sea, preferably from few extensive surveys rather than many regional sampling programmes to ensure consistency of sampling efficiency. Resolution should be at least at the level of ICES rectangles, with at least one sample from each rectangle in the North Sea.

Data should be disaggregated to the extent possible, and cover the period 2000-04 to provide a description of state for this period. Selected time-series from at least the early 1980s should be made available to derive information on trends in these effects 'over the recent decades'.

In addition, it will be necessary to have information from the literature describing the life history characteristics of selected species, and the current state of knowledge on stock discrimination. This will be particularly relevant to an assessment of the effects of fishing on metapopulations, rather than regionally discriminated stocks.

12.5 Fishing pressure and impact

12.5.1 Introduction

Greenstreet & Rogers (2000) stated that fishing effort has never been evenly distributed across the North Sea. Different gears, directed at different target species, with differing levels of impact on the components of the ecosystem, have been used at varying intensities across the North Sea. In order to develop spatially and temporally resolved indices of the ecological disturbance of fishing on marine biota, at the very least there is a clear need to obtain data for the amount of fishing effort in a given area at a given time per fleet and gear. Most of the countries that fish within the North Sea record routine measures of fishing effort at the scale of the ICES rectangle. These data are however variable in the procedures and measures used to record the data and the length of time for which they are available (Greenstreet *et al*., 1999).

There are various sources of data on the impact of fishing on the various ecosystem components but these are far from comprehensive. Catch data exist on a fleet-by-fleet basis for most of the commercial species but these data do not include discard mortality. Data that estimate the discards of commercial fish species or non-target fish species are much scarcer and for benthos or marine mammals almost non-existent. For benthos there is the additional mortality in the path of the gear that does not show up in the catches and hence usually remains unreported.

This leaves two other ways to determine the impact of fishing: (1) derive from observed changes in the state of these components (see section 12.3) or (2) estimate from fishing effort data.

12.5.2 Estimating fishing impact

When estimating impact from fishing effort additional data are necessary. Here, we will present a hierarchical overview of the type of data that are needed for indices of fishing impact that perform increasingly better at describing the impact as the information content of them is increased.

The impact of fishing can be described at four levels of increasing information content (Figure 12.1). As the lowest spatial resolution for management in the North Sea is the ICES rectangles (approximately 30x30 Nm), this is also the scale at which the impact indices are determined. The bottom level is that of fishing effort; as each country fishing in the North Sea routinely collect these data (e.g. for assessment purposes), they should be available for each fishing method. However, neither the quality of the data nor access to them for research purposes, is guaranteed. Information on fishing practices and gear characteristics does however allow the calculation of second level indices: the intensity at which the seabed was swept or a volume of water trawled. The impact of fishing on a habitat, fish- or benthic community is not only determined by the measure of effort (e.g. days-at-sea or intensity) but also how this effort is distributed within that area; an even distribution of effort will have a bigger impact than that same amount of effort concentrated on a relatively small area, leaving the remainder unaffected. Using information on the micro-scale distribution of effort results in the third level indices: micro-scale frequency distribution of swept area or volume trawled. Finally, by combining the 3rd level indices with information on the effects of the gear and the abundance of the ecosystem components in a spatial unit we reach the highest (4th) level indices which actually give the proportion of a habitat that is destroyed or the mortality of an ecosystem component induced by fishing. Below an example will be provided of the extent to which the various factors may affect the impact estimates. For a proper assessment of the effects of fishing in the North Sea, data that allow quantification of these factors need to become available for each of the fishing methods operating in the North Sea.

As an example of the effect of including fishing parameters and gear characteristics we estimated the impact using two fishing practice parameters for two segments of the Dutch beam trawl fleet: trawling speed and proportion of the day spent trawling. These parameters are necessary to transform the effort in days-at-sea into a better (Level 2) estimate of impact: e.g. area trawled. The composition of the Dutch beam trawl fleet varies considerably in terms of the relative importance of the two segments both between years and ICES rectangles. In the inshore rectangles the fleet consists entirely of eurocutters while the other extreme is often found in the offshore rectangles. On average the eurocutters make up about 18% (period 1990-2002) of the fleet. Therefore the estimated level 2 impact may differ almost by a factor of 5 depending on the relative importance of the two segments in a particular area.

For one ICES rectangle, the effects of including the spatial micro-scale distribution of effort and information of the gear effect on an ecosystem component (expressed as gear encounter mortality) are combined into a Level 4 impact estimate (i.e. % population mortality) (Figure 12.2). Clearly the fishing impact differs markedly depending on the combination of fishing effort and the gear effect. This supports the findings of a number of other studies that have used higher resolution effort distribution data to evaluate the disturbance of fishing in benthic invertebrate populations of the southern North Sea (Bergman & van Santbrink, 2000; Piet *et al*., 2000).

Table 12.1. Fishing parameters of two segments of the fleet based on logbook data.

Hp segment	Width of the beam trawl (m)	Trawling speed (knots)	Proportion of the day trawling (%)	Time per day trawling (hr)	Area trawled per day (km^2)
Eurocutters (<300Hp)	2 x 4	4.6	60.2	14.4	0.99
Large vessels (>300Hp)	2 x 12	6.1	68.1	16.3	4.40

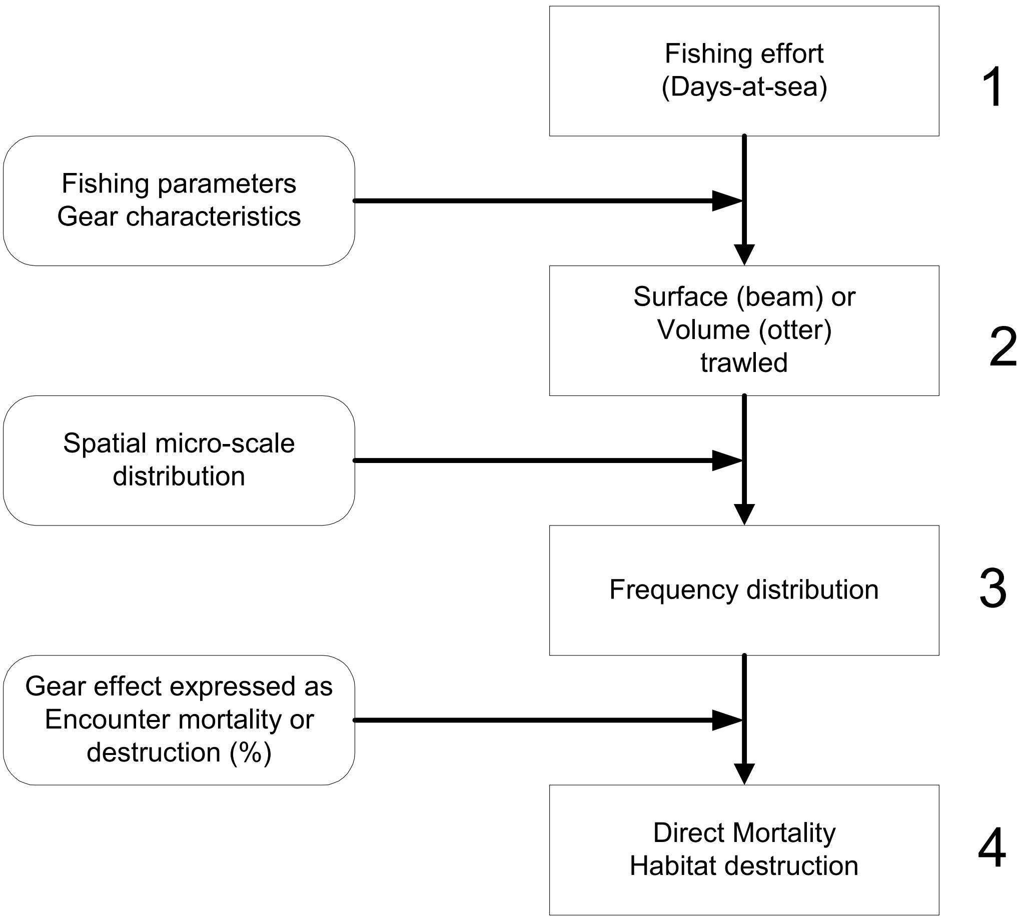


Figure 12.1. The overview that describes how indices of fishing impact can be derived from fishing effort data at different levels of information content. Level 1 is fishing effort, Level 4 the best estimate of fishing impact. The boxes on the left describe the type of information required, the level is indicated to the right. Encounter mortality or destruction is the % mortality or % habitat destroyed caused by a singular passagte of a specific type of gear. Direct mortality is the % mortality of an ecosystem component (or % of a habitat destroyed) in an area caused by a known amount of effort of a fishery that operates that gear type.

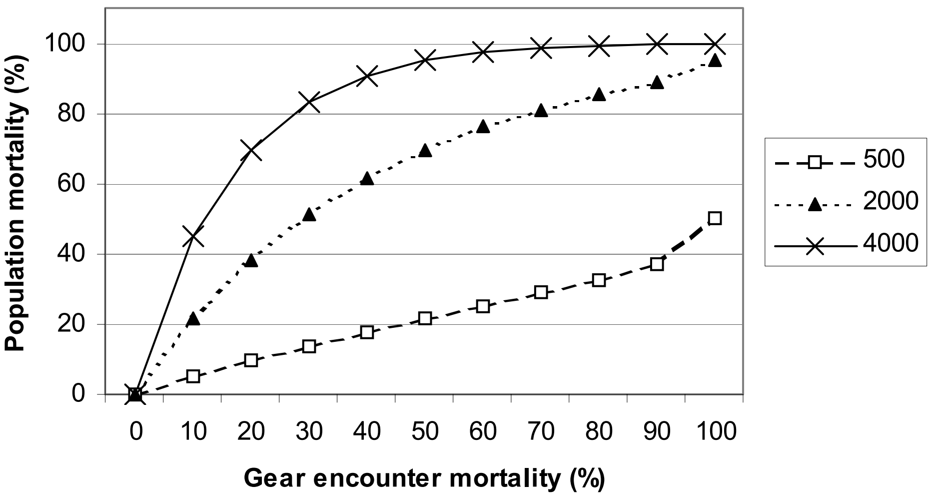


Figure 12.2. Calculated population mortality for four different effort (in days-at-sea per ICES rectangle) scenarios depending on the effect of the gear (expressed as proportion mortality per encounter).

12.5.3 Fishing effort data requirements and availability

In order to describe the impact of fishing on the North Sea biota starting from the effort data, we distinguish sources of additional data that can be used to improve these estimates. The lowest level effort data are available for most international fleets, however as mentioned earlier, access to these data is not always guaranteed eventhough it is routinely collected. Between international datasets there may be inconsistencies in the units used (days-at-sea, Hpday-at-sea, hours trawled etc.) or the way the fishing methods are distinguished within the national fleets. The availability of fishing parameters can help to combine the different types of effort data and provide better indices of the actual impact this fishing effort has. Fishing impact can also be better estimated if the distribution of frequencies with which an area is trawled or a volume of water fished is known as well as the effect that a singular passage of the gear has on a particular ecosystem component.

Different sources of data are now becoming available to track the microscale distribution of individual fleets. A proportion of the Dutch fleet has been tracked for over 10 years. Initially 'black boxes' (automated position recording systems) were installed on 10% of the fleet and these gave positions every 6 minutes to an accuracy of approximately 100m (data from 1993-2000). Since the 1st January 2000 it has been compulsory for EC registered fishing vessels over 24m to report their location at least every 2 hours, using the VMS (Vessel Monitoring through Satellite) system. Sometimes the measured speed is also given, but if it is not measured and needs to be calculated from the distance covered between position registration it should be realised that this will give an underestimate of the data because (1) the vessel does not follow a straight line between the two registrations and (2) if the vessels starts hauling between two registrations the calculated mean speed will decrease. These points will become increasingly important as the time interval between registrations is increased.

Due to problems instigating the VMS system on an international scale, reliable data may only be available for some countries from later dates (Dinmore *et al*., 2003). If VMS data are only available for a sample of the fleet this sample needs to be representative in order for the frequency distribution to be used for the whole fleet. However, unless the entire fleet is sampled with a small enough time interval between registrations it will always remain uncertain whether there really has not been any fishing in the squares that were deemed un-fished. Taking the remarks above into consideration the VMS data are ideal to provide a level 3 impact for the various metiers and are routinely collected by each EC country with a fishing fleet operating in the North Sea for enforcement purposes. However, often they are not made available for scientific purposes. The Dutch data are available for 30% of the fleet and the German data are available for the whole fleet, but access to the data from other countries is more difficult (S. Ehrich, S. Jennings, P. Kunzlik & G. Piet, *pers comm*.). It is known that VMS data from both Scottish and English fleets are restricted but it is not known whether there is any access to data from Belgium, Denmark, France or Norway. Quantifying fishing impact and hence the ecosystem effects of fishing is **severely hampered without such data**.

Another source of information on effort distribution is the overflight data, which is based on the positions of vessels taken by aeroplane observers twice a week (Jennings *et al.*, 2000). This is potentially available for all boats fishing in UK waters and may help to resolve effort distribution where VMS data are not accessible. A number of smaller scale studies of microscale effort distribution also exist for *Nephrops* targeted fleets in the Clyde Sea and the Fladden Ground of the North Sea (Marrs *et al*., 2000 & 2002; J. Atkinson & I. Tuck, *pers. comm*.).

12.5.4 Gear effect data requirements and availability

In fishery science catchability is a known concept (e.g. Dickson 1993) but for fish there are no estimates of the encounter mortality caused by the passing of the gear. However, several studies exist that have at least identified some of the factors involved i.e. herding (Engås and Godø 1989, Engås 1994), swimming speed (Wardle 1975, 1977, He 1993) or other behaviour aimed at escaping the net (Bublitz 1996, Albert et al. 2003) and finally selectivity (Myers and Hoenig 1997, Reeves *et al*. 1992) for those fish that end up in the net. To further complicate the matter, encounter mortality consists not only of mortality of fish caught in the net (i.e. catchability) but also mortality of other fish (e.g. those that die after passage through the net).

For the benthic community encounter mortality has been determined for some species (for review and meta-analysis see Collie 2000 and Kaiser *et al.* submitted) as there are sampling techniques that allow the determination of absolute pre- and post-haul abundances. In Piet *et al*. (2000) the disturbance (expressed as population mortality) caused by the fishery was determined for a limited number of benthic species for which encounter mortalities were known.

Determining the "gear effect" expressed as encounter mortality may be difficult as almost every vessel will have a gear with slightly different characteristics and rigging (e.g. for a beam trawl the number of ticklers, use of chain mat or flip-up rope etc.) which affects the catchability and hence encounter mortality of that gear. Considering this variation it would seem appropriate to distinguish each of these gear types but because of the amount of work necessary to estimate encounter mortality it is simply not feasible to determine this for each of the gear type – ecosystem component combinations. As a way forward we suggest distinguishing a limited number of "standard" gear types for which encounter mortality can be determined and classifying the various gear types as belonging to one of those "standard" gear types (See Section 12.5).

12.5.5 Synthesis

The best estimate of impact of fishing in the North Sea can be determined for any ecosystem component if the following data are available:

1) Data that describe the total mortality (catch, discard and other) induced by all metiers operating in the North Sea at a spatial resolution of ICES rectangles and a minimum temporal resolution of years.

2) If such data do not exist direct impacts may be estimated with:

- Effort data of all metiers at a resolution of at least ICES rectangles but preferably higher (i.e. 1x1 Nm squares) and a temporal resolution of at least years but preferably higher (quarters).
- Fishing parameters and gear characteristics that allow effort to be transformed to a unit of frequency trawling
- Abundance/distribution data of the ecosystem component. For this the same requirements apply as for effort
- An estimate of encounter mortality/damage for each of the gears in the metiers

The level of availability of the above data determines the accuracy of the estimated impact.

12.6 Conclusions and recommendations for the assessment of status and trends in fishing effects

In order to assess the effects of fishing on marine biota of the North Sea we consider that the status and trends of Habitats, Nutrients, Benthos, Fish, Seabirds and Marine Mammals should be assessed. We have suggested the data required to assess this in relation to each component but acknowledge that the resolution of data available will vary depending on the component considered (e.g. much greater resolution for fish than habitats). We are however aware that work is currently being undertaken to collect data for some of the components that are under-represented (e.g.

habitats, benthos) and stress the importance of this data being made available to WGECO, where possible for the 2005 meeting, if we are to consider the effects across all North Sea biota.

We have also addressed the requirements for fishing effort data to be able to assess status and trends in the actual impacting activity (pressure). We are aware that the resolution of data available to do this varies between countries and fleets but suggest that a number of procedures may be undertaken to map effort at differing levels that are of relevance to the ecological disturbance it is associated with. Considering the availability of data on a North Sea scale and the likely development in both accessibility of data and furthering of current understanding of the indirect effects of fishing in the near future (i.e. over the next year), WGECO recommend the following procedures for the assessment of fishing effects in the North Sea. We emphasise, however, that the availability of data will determine whether a procedure is carried out across the North Sea or at a smaller spatial scale as a case study.

1. Describe the distribution of fishing effort by fleet and gear at the scale of the ICES rectangle.

This will give a quantification of the pressure and will be possible at a coarse level (e.g. otter trawl, beam trawl, pelagic trawl) (Level 1 in section 12.4.2). Jennings *et al* (1999) compiled international effort data for demersal gears for the period 1990-1995 and the EC project MAFCONS will be updating this dataset to include recent years in 2004. This will allow for an examination of recent trends and status for the years 2000-2003/4. However, this compilation will not include any set net gears or pelagic gears. WGECO request that effort data for the years 1984-2004 (where possible) are supplied by the relevant working groups for all gears, and in particular, set net and pelagic fisheries. Without these data it will be impossible to assess the effects of fishing on marine biota of the North Sea. We also stress that with the continued restriction on access to effort (and catch, see below) data from a number of member states fishing in the North Sea we will always underestimate the effects. There are no clear relationships, for example based on the relationship between TACs and effort, on which we can estimate the amount and distribution of effort missing from these fleets. Further to this, we recommend that all microscale distribution data based on VMS recordings also be compiled where possible. The exact data required are outlined in Table 12.2 and in order to assess the distribution of fishing effort, we suggest the following term of reference for the stock assessment groups meeting in 2005 to be available to WGECO before they meet in April 2005:

ToR: *Extract and compile effort data for all gear types and all fleets based on logbook data at the scale of the ICES rectangle across the North Sea area. This data should cover the period 2000-2004 and be provided, where possible, for the period 1984 -2004 to assess trends.*

2. Describe the distribution of catch mortality by fleet and gear at the scale of the ICES rectangle.

This will give a quantification of the direct effects of fishing on biota caught in the fishing gears. This will not however account for mortality in the path of the gear or alteration of habitat. Building on the compilation of effort data as requested in procedure 1, catch data (including landed and discarded components) will be compiled. We are aware that landed catch data are available for commercial fish and shellfish species, but consider that there is still much progress to be made in compiling data on discards, particularly in relation to ecosystem components other than the fish. Observer programmes are being undertaken to assess discarding practices and we stress the importance of making these data available to WGECO. Catch data should be available by gear and fishing method at the scale of the ICES rectangle for the years 2000-2004 where possible and over a longer time period (last 10-20 years) where possible (See Table 12.2). Landed catch data will be compiled from the published assessments but we suggest the following term of reference for WGFE to improve the assessment of discards. This should be available to WGECO before we meet in April 2005.

ToR: *Extract and compile discards data for all gear types and all fleets at the scale of the ICES rectangle across the North Sea area. This data should cover the period 2000-2004 and be provided, where possible, for the period 1984 -2004 to assess trends.*

3. Model the direct effects of fishing resulting from towpath mortality, by gear and method at the scale of the ICES rectangle.

Using the methodology described in Section 12.4.2 we propose that the approach of determining the impact of fishing in the towpath be furthered using the data compiled in procedure 1 for demersal gears at the scale of the ICES rectangle. In order to undertake this, the data outlined in point 2 of Section 12.4.5 will be required. As suggested earlier, estimating encounter mortalities may only be possible at this time for benthic communities (based on a virtual community). In order to do this at the scale of the North Sea, we will need disaggregated effort data (by gear and vessel type) at the smallest spatial resolution possible and data for distribution of benthic communities at the North Sea scale. The data required to do this are covered by the requests made in reference to procedures 1 and 4.

4. Analyse the status and trends in effects of fishing on ecosystem components

In section 12.3 we summarise the data required to assess the effects of fishing on each marine ecosystem component. These include both direct and indirect effects and we have suggested in the relevant sections where we think this data may be difficult to access. In Table 12.2 the data required to address the effects of fishing on each ecosystem component are listed and the suggested source of data given. In reference to each of the ecosystem components we suggest the following terms of reference and indicate the group(s) to which the request is given. As stated above the work undertaken in reference to these ToRs should be available to WGECO before we meet in April 2005.

In reference to Marine mammals and Seabirds WGECO suggest the following ToR to be addressed by WGSE and WGMME where appropriate:

ToR: *For each species effected by fishing, extract and compile data on species abundance, age structure of the population and offshore distribution with variation in this by season, at the scale of the ICES rectangle across the North Sea area. This data should cover the period 2000-2004 and be provided, where possible, for the period 1984 -2004 to assess trends. Also where possible, provide information on diet and variation of this for all species described.*

In reference to Habitats, WGECO suggest the following ToR to be addressed by WGMHM:

ToR: *Extract and compile habitat mapping data at EUNIS level 4 or above at the scale of the ICES rectangle across the North Sea area. Also provide maps of sediment characteristics at the scale of the ICES rectangle across the North Sea area.*

In reference to Benthos, WGECO suggest the following ToR to be addressed by WGBE:

ToR: *Extract and compile data on benthic abundance and biomass per species at the scale of the ICES rectangle across the North Sea area for the years that it is available, preferably for the years 2000-2004. Where individual species biomass or abundance data are not available provide total abundance and biomass at the level of the community. This is in reference to both infaunal and epifaunal communities and meiofaunal communities if the data are available. Where possible compile data over longer time periods to reflect trends. Also provide data on sediment characteristics of the relevant sampled stations where available.*

In reference to Fish, WGECO suggest the following TOR to be addressed by WGFE:

ToR: *If the following data are not already available from the ICES database, extract and compile fish individual abundance at length, weight at length, age at length and maturity at length, for all species (both commercial and non-commercial) available at the scale of the ICES rectangle across the North Sea area. This data should cover the period 2000-2004 and be provided, where possible, for the period 1984 -2004 to assess trends. Further to this, extract and compile all available stomach content data for diet analysis.*

Table 12.2 Data required to assess the status and trends in effects of fishing on North Sea marine biota. All data should be disaggregated to the scale of the ICES rectangle where possible and available for the period 2000-2004 and for recent decades to assess trends.

	Data required	Source
Ecosystem component		
Marine mammals/ Seabirds	Species abundance and (offshore) distribution, and variation in this by season	WGSE, WGMME
	Knowledge of diet by species	WGSE, WGMME
	Age structure of the population	WGSE, WGMME
Habitats	Habitat mapping at EUNIS level 4 or above	WGMHM
Benthos	Species identity and abundance	WGBE
	Species individual biomass	WGBE
	Total biomass of the community	WGBE
	Sediment characteristics	WGBE, WGMHM
	Species abundance at length	WGBE
Fish	Species identity	WGFE
	Species abundance at length	WGFE
	Species individual weight at length	WGFE
	Species individual age at length	WGFE
	Species individual maturity at length	WGFE
	Representative stomach sampling for trophic analysis	WGFE
Fishing activity data		
Fishing effort	Effort data for all gear types and all fleets (logbook data)	Assessment groups
Catch	Discards data	WGFE

REFERENCES

Albert, O. T., A. Harbitz, et al. (2003). "Greenland halibut observed by video in front of survey trawl: behaviour, escapement, and spatial patterns." Journal of Sea Research **50**: 117-127.

Auster, P.J. & Langton, R.W. (1999). The effects of fishing on fish habitat. *American Fisheries Society Symposium **22***.

Bergman, M.J.N. & van Santbrink, J.W. 2000. Fishing mortality of populations of megafauna in sandy sediments. *In*, Kaiser, M.J. & de Groot S.J. (Eds.). *Effects of Fishing on Non-Target Species and Habitats: Biological, Conservation and Socio-economic Issues.* Pp. 49-68. Blackwell Science, Oxford.

Bublitz, C. (1996). "Quantitative evaluation of flatfish behaviour during capture by trawl gear." Fisheries Research **25**: 293-304.

Collie, J.S., Hall, S.J., Kaiser, M.J. & Poiner, I.R. 2000. A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology* 69, 785-798.

Dickson, W. (1993). "Estimation of the capture efficiency of trawl gear. I Development of a theoretical model." Fisheries Research **16**: 239-253.

Dickson, W. (1993). "Estimation of the capture efficiency of trawl gear. II: Testing a theoretical model." Fisheries Research **16**: 255-272.

Dinmore, T.A., Duplisea, D.E., Rackham, B.D., Maxwell, D.L. & Jennings, S. 2003. Impact of a large-scale area closure on patterns of fishing disturbance and the consequences for benthic communities. *ICES Journal of Marine Science 60(2),* 371-381.

Duineveld, G.C.A., Künitzer, A., Niermann, U., de Wilde, P.A.W.J. and Gray, J.S. (1991). The macrobenthos of the North Sea. *Netherlands Journal of Sea Research* **28**: 53-65.

Duplisea, D.E., Jennings, S., Warr, K.J. & Dinmore, T. 2002. A size-based model of the impacts of bottom trawling on benthic community structure. *Canadian Journal of Fisheries and Aquatic Science **59***, 1785-1795.

Engås, A. (1994). The effects of trawl performance and fish behaviour on the catching efficiency of demersal sampling trawls. Marine fish behaviour in capture and abundance estimation. A. Fernö and S. Olsen. Oxford, Fishing News Books**:** 45-68.

Engås, A. and O. R. Godø (1989). "The effect of different sweep lengths on the length composition of bottom-sampling trawl catches." J. Cons. Int. Explor. Mer **45**: 263-268.

Godø, O. R. and S. J. Walsh (1992). "Escapement of fish during bottom trawl sampling - implications for resource assessment." Fisheries Research **13**: 281-292.

Greenstreet, S.P.R. & Rogers, S.I. 2000. Effects of fishing on non-target fish species. *In*, Kaiser, M.J. & de Groot, B. Effects of Fishing on Non-Target Species and Habitats: Biological, Conservation and Socio-economic Issues. Oxford, U.K, Blackwell Science. Pp. 217-234.

Greenstreet, S.P.R., Spence, F.B., Shanks, A.M. & McMillan, J.A. 1999. Fishing effects in northeast Atlantic shelf seas: patterns in fishing effort, diversity and community structure. II. Trends in fishing effort in the North Sea by UK registered vessels landing in Scotland. *Fisheries Research **40***, 107-124.

Hall, S.J., Raffaelli, D. & Thrush, S.F. 1994. Patchiness and disturbance in shallow water benthic assemblages. *In*, Giller, P.S., Hildrew, A.G. & Rafaelli, D.G. (Eds.). *Aquatic Ecology: Scale, Pattern and Process*. Blackwell Science, Oxford. Pp. 333-375

He, P. (1993). "Swimming speeds of marine fish in relation to fishing gears." ICES Journal of Marine Science **196**: 183-189.

ICES. 2000. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 2000/ACME: 02.

ICES. 2001. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 2001/ACME: 02.

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

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

Jennings, S., Alsvåg, J., Cotter, A.J.R., Ehrich, S. Greenstreet, S.P.R., Jarre-Teichmann, A., Mergardt, N., Rijnsdorp, A.D. & Smedstad, O. 1999b. Fishing effects in northeast Atlantic shelf seas: patterns in fishing effort, diversity and community structure. III. International trawling effort in the North Sea: an analysis of spatial and temporal trends. *Fisheries Research 40*, 125-134.

Jennings, S., Warr, K.J., Greenstreet, S.P.R. & Cotter, A.J.R. 2000. Spatial and temporal patterns in North Sea fishing effort. *In*, Kaiser, M.J. & de Groot, B. *Effects of Fishing on Non-Target Species and Habitats: Biological, Conservation and Socio-economic Issues.* Oxford, U.K., Blackwell Science. Pp. 3-14.

Jennings, S., Dinmore, T.A., Duplisea, D.E., Warr, K.J. & Lancaster, J.E. 2001. Trawling disturbance can modify benthic production processes. *Journal of Animal Ecology* ***70*****,** 459-475.

Johnson, K.A. 2002. *A Review of National and International Literature on the Effects of Fishing on Benthic Habitats*. NOAA Technical Memorandum NMFS-F/SPO-57. 72pp.

Marrs, S.J., Tuck, I.D., Arneri, E., Atkinson, R.J.A., Santojanni, A. & Stevenson, T.D.I. 2000. Improvement of *Nephrops* stock assessment by use of micro-scale mapping of effort and landings. Final Report of the European Commission Study Contract 97/0100.

Marrs, S.J., Tuck, I.D., Arneri, E., La Mesa, M., Atkinson, R.J.A., Ward, B. & Santojanni, A. 2002. Technical improvements in the assessment of Scottish *Nephrops* and Adriatic clam fisheries. Final Report of the European Commission Study Contract 99/077.

Myers, R. A. and J. M. Hoenig (1997). "Direct estimates of gear selectivity from multiple tagging experiments." Canadian Journal of Fisheries and Aquatic Sciences **54**: 1-9.

Piet, G.J., Rijnsdorp, A.D., Bergman, M.J.N., van Santbrink, J.W., Craeymeersch, J. & Buijs, J. 2000. A quantitative evaluation of the impact of beam trawling on benthic fauna n the southern North Sea. *ICES Journal of Marine Science 57*, 1332-1339.

Reeves, S. A., D. A. Armstrong, et al. (1992). "The effects of mesh size, cod-end extension length and cod-end diameter on the selectivity of Scottish trawls and seines." ICES Journal of Marine Science **49**: 279-288.

Wardle, C. (1975). "Limit of fish swimming speed." Nature **255**: 725-727.

Wardle, C. S. (1977). Effects of size on the swimming speeds of fish. Scale effects in animal locomotion. T. Pedley. New York, Academic Press**:** 295-313.

13 FOOD FOR THOUGHT

13.1 A possible analytical approach to increasing the use of the information in the metrics associated with multiple EcoQOs

The field of Psychometrics has over a century of experience with testing data which have many of the characteristics of the evaluation of multiple EcoQOs and metrics: multiple criteria which overlap somewhat in information content but vary in importance for different uses; at best ordinal scores of cases on the criteria; and varying strength of evidence. Psychometrics has had to address these analytical problems, particularly in the field of personality and aptitude testing (Dorfman and Hersen 2001, Murphy and Davidshofer 2001). In that field, unless the test is focused on a very specific property, all credible tests have scores reported on a variety of dimensions (the "profile"). For many types of tests substantial instruction and training is provided to those who have to interpret the information contained in the profile. Test results are interpreted relative to normative samples – scores of hundreds to thousands of subjects whose performance traits are known accurately on exactly the properties that the test is intended to measure. For many tests different norms must be provided for applications in different contexts.

Two important messages come from this work. First, the information in inherently multi-dimensional traits (like the state of an ecosystem as reflected in the "values" of metrics associated with multiple EcoQOs) should not be collapsed into misleadingly simple scores to compare. Second, it will only be possible to begin to interpret the value of metrics and EcoQOs from a variety of ecosystem properties once there is a great deal of experience with using the EcoQOs actively in management, so we acquire the necessary normative information about what types of EcoQOs really do support decision-making in various governance settings. It is hard to consider it a virtue that most ecosystems have been badly overfished. However, at least the present situation provides lots of replicates of the diverse ways that excessive or otherwise irresponsible fishing can damage ecosystems. Hence, it may be possible to develop the necessary "normative diagnostics" for ecosystems which have been damaged in various ways (top predators overfished, productive capacity of habitats reduced, vulnerability of recruitment dynamics to environmental forcing amplified, etc). Using such diagnostics and tracking performance of well chosen metrics associated with well chosen EcoQOs, it will be possible to identify when and how fishing is beginning to move the ecosystem into damaged or undesirable states. This would be by the full profile of status on the EcoQOs, though, and not on some aggregate of their individual performances.

14 RECOMMENDATIONS FOR FUTURE ACTIVITIES

14.1 Summary of recommendations

During the course of our work this year, WGECO have reached a number of conclusions and recommendations for further work by ICES, its' customers or for the wider science community.

We have identified specific recommendations for ICES customers in different parts of the report, and these can be found in the appropriate sections. In summary, we have made recommendations for revised wording of the EcoQO relating to spawning stock biomass of commercial fish (section 3.1), and black-legged kittiwake breeding productivity (section 4.2). We recommend that the proposed EcoQO for average weight and average maximum length of the fish community is a poor performance metric and should only be used for surveillance (section 5.5.1), and have recommended that OSPAR consider dropping the element (p) concerned with the density of opportunistic species as these are ubiquitous and provide no link to human impacting activities (section 5.5.2). We also recommend that EcoQO element (o) concerned with the density of fragile (sensitive) species be advanced by the use of a selection of a very limited suite of 'sentinel' species (section 5.5.2). Criteria for selecting these sentinel species will need to be developed and must take into consideration data availability. Our work on threatened and declining species metrics (EcoQO element b) suggested that observed trends in the abundance of vulnerable species provide little information to help with short-term management decisions, and that an alternative objective would be a 'response' indicator of listed species for which a recovery plan had been prepared and implemented (section 5.5.3).

WGECO was interested to learn about the proposed new ICES Advisory Framework, and could see several ways in which the Group could contribute to the science and advisory process. In section 6.3 we suggest a suite of five roles that we feel WGECO could fulfil, and request further information on how REGNS and BSRP will carry out their assessments in time for WGECO 2005. In considering our ToR related to the preparations for an integrated assessment of the North Sea, we have drawn up a framework for assessing the effects of fishing on the status and trends of marine biota (section 12.5), and have recommended six specific ToRs for other ICES groups who will be able to provide the data that we will need.

Sprat and herring are large, important fisheries for the Baltic States. After examining the impacts of industrial fishing in the Baltic we concluded that complete evaluation of the ecosystem effects of these fisheries was not possible, and so recommended that sampling and assessment of the species compositions of the caught pelagic fish in the Baltic should be re-evaluated and revised at national level (section 8.4.1).

In addition to these recommendations, we have identified two other areas of work that we feel would be productive to pursue.

14.2 Cross-calibration of fishery independent surveys

In recent years several metrics have been developed and applied to fish survey data that vary in their time series longevity. In TORs c) and g) it became apparent that few surveys cover an entire management area or go far back in time. It was also noted that although surveys may show similar trends, the absolute values and hence potential reference levels are survey-dependent. Therefore it is recommended that, where possible, cross-calibration of surveys should take place. This may allow a greater coverage of an area, a continuation of the time-series after one survey has stopped, and potentially allow the reference points derived from defunct surveys to be applied to ongoing data sets.

WGECO therefore encourages WGFE to explore the possibility of cross-calibrating those surveys that can deliver EcoQ metrics and when possible provide conversion coefficients for the survey-combinations.

14.3 Consideration of Strategic Environmental Assessment

It is apparent from the planned new Advisory Structure that further activities in the ICES community will involve a greater degree of integration of advice in order to better support an ecosystem approach to management of marine activities. In this regard, it is likely that the requirements for Environmental Impact Assessment and Strategic Environmental Assessment (Annex II of the EIA Directive 97/11/EC) of human activities in the marine environment will in due course be applied to parts of the commercial fishing industry. Under these circumstances it would be appropriate for WGECO to be involved as expert contributors to such a process and / or providing Quality Assurance for the activity. We recommend that ICES takes note of this impending issue, and invites WGECO to be more closely involved through a ToR on developing a framework.

14.3.1 Suggested ToR

Commence development of a framework for the carrying out of Strategic Environmental Assessment for a fishery.

14.4 Election of new chair

In line with tradition this was Chris Frid's third and hence last meeting as Chair. During the meeting the members of WGECO elected Dr Stuart Rogers (UK) as their nominee for the chair of future meetings.

14.5 Future meetings

The membership of WGECO confirmed their readiness to meet, if necessary, for 8 days from 13 – 20 April 2005 at ICES Headquarters, Copenhagen.

Annex 1

WORKING GROUP WGECO

14–21 April 2004

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