ICES Cooperative Research Report

Rapport des Recherches Collectives

No. 272

July 2005

Ecosystem Effects of Fishing: Impacts, Metrics, and Management Strategies

Edited by:

Dr Jake C. Rice
Canadian Stock Assessment Secretariat
200 Kent Street, Stn 12036
Ottawa, ONT K1A 0E6
Canada

International Council for the Exploration of the Sea

Conseil International pour l'Exploration de la Mer

CONTENTS

CHAPTER 1

CHAPTER 1

ECOLOGICAL QUALITY OBJECTIVES, REFERENCE POINTS, AND FISHING EFFECTS

In its series of three meetings, WGECO devoted considerable attention to the issue of Ecological Quality Objectives and Precautionary Reference Points for ecosystem properties. As this work progressed, the context in which the deliberations were held also changed. The issue of Ecosystem Objectives evolved from a domain of conceptual thinking, to very practical evaluation of candidate ecosystem objectives, indicators, and reference points. The consequences of the work done by WGECO are reflected clearly in the Bergen Declaration, adopted in March 2002. Both concepts and choices of wording in relevant parts of the Declaration, particularly Annex 3, show the strong influence of the preparations done by WGECO.

In this Chapter we present the logical development of the operational framework for selecting and using Ecosystem Objectives in fisheries management. We start with the framework of single-species reference points that ICES adopted for advice on fisheries management in 1997, and consider what extensions to the approach would be necessary to protect ecosystem properties, as well as single stocks, from serious or irreversible harm from fishing. Once the necessary extensions to the single-species reference points were identified, we considered what ecosystem management objectives would be appropriate in order to structure the selection and use of reference points for ecosystem properties. In undertaking this, it became clear that there was great potential for confusion of terms and concepts, particularly because many groups, with different professional make-ups, were publishing material on this subject. Therefore we undertook a careful exposition of the appropriate language for discussing ecosystem objectives, reference points, and related topics, to ensure that dialogue was consistent with the already established practices in both single-species fisheries management, and protection of habitats and species from pollutants.

Once the conceptual framework of objectives and reference points was developed, we moved to the practical level, and attempted to identify specific candidate objectives, indicators, and reference points for including ecosystem considerations in fisheries management. It rapidly became clear that the criteria for selecting among candidate indicators and reference points were going to be crucial, to keep the whole approach as a scientific process, rather than a popularity contest. Therefore, in the final sections of this Chapter, we develop rigorous and objective screening criteria for selecting indicators and reference points, and evaluating their performance. We test our criteria for selection and performance evaluation of objectives, indicators, and reference points, to provide a factual basis for advice on practice.

1 Reference Points, Including Ecosystem Considerations

The first step was to consider *potential reference points which might be used for including ecosystem considerations in relation to the precautionary approach.* The broader management objectives for which the quantitative reference points are developed and used will be considered in the following section. This material is readily interpretable in the context of current approaches to fisheries. However, WGECO considered a much broader framework than just traditional fisheries management objectives. Many other types of objectives already influence fisheries practices, from very local scales (for example, the protection of specific bivalve beds close to shore-based viewpoints, because they attract concentrations of seaducks) to very large ones (the objective of protecting ecosystem diversity, for example). It is important that the following arguments are viewed as applying in all of these contexts, and not just as serving traditional fisheries management objectives. Likewise, it is important that specific objectives be discussed and set by society in many contexts, in addition to fisheries.

1.1 Statement of the Issue

The precautionary approach (FAO, 1995; Doulman, 1995; Garcia, 1996) has been accepted as a guiding principle in fisheries management. It covers biological, social, and economic aspects of fisheries. In the practical implementation of the precautionary approach ICES has established limit reference points and precautionary reference points for commercial stocks, and has called on managers to set target reference points as well. These reference points are recommended as quantitative management objectives. At the current exploitation pattern of fish stocks, the short-term objective is to have a low probability of fish stocks falling below limit reference points, to ensure a long-term sustainability. This is achieved by advising that stocks be kept above the precautionary reference points, which

accommodate the uncertainty in the stock assessments (ICES, 1997e). Target reference points are viewed as long-term objectives, to be achieved over time through managed rebuilding of stock sizes.

An additional aspect of the precautionary approach is the integration of fisheries management and ecosystem management. An ecosystem approach in the management and assessment of fisheries involves considering all relevant physical, chemical, and biological ecosystem variables (Anon., 1997). It thereby implies a widening of the current implementation of the precautionary approach. The question at stake is whether reference points being developed for commercial species are sufficient to ensure an effective ecosystem management? To explore that question we review the ecosystem considerations of different potential reference points, including ones for target and non-target species of fisheries (single-species reference points), multispecies and ecosystem properties, outputs of mass-balance models, and community metrics.

ICES acknowledges the need to manage fisheries in a manner which ensures ecosystems are sustainable, in the sense that no species becomes extinct. Nonetheless little work has been done thus far on how to define reference points in an ecosystem context. Naturally such definitions would not only be restricted to fish but would need to include other components of marine fauna such as benthos, seabirds, and marine mammals. For many of these groups reference points relating anthropogenic impacts to population status have either been defined elsewhere or are non-existent. In addition, sustainable management in an ecosystem context would need to consider not only how fishing mortality affects individual stocks and their genetic make-up, but also how discarding and physical seabed disturbances affect the system.

One of the largest and most direct effects of fishing is the harvesting of target species. These effects are quantified in single-species population models, form which reference points can be drawn. If it were the case that management complied with reference points as they were intended to be used, fisheries would already be much further on the way to meeting any specified ecosystem objectives. On the other hand, commercially important species are by their nature often highly productive components of the ecosystem. Reducing their abundances through fishing may have great impacts on the dynamics of the food web. Also, because they often are less productive, non-targeted species may be much more vulnerable to mortality caused by fishing than are many commercially important species. It has been proposed that within a single-species approach more sensitive species commonly taken as bycatch could be useful indicators for determining the state of the ecosystem.

Multispecies models contain more ecosystem considerations than their single-species counterparts. The multispecies models used by ICES account for predator/prey relationships. In work completed to date even for target species of fisheries they have led to more conservative estimates of reference points and estimate lower fishing mortality rates for a sustainable fisheries than do single-species models (ICES, 1997c). In that sense they require more conservative fisheries to achieve an equal degree of risk protection.

Fisheries also can affect community structures. Due to the high selectivity of fishing, the values of many community metrics may be altered. The question is, can metrics like shifts in size or productivity at different trophic levels also serve as potential ecosystem reference points? To explore this question the value of multispecies modelling, mass-balance models, MSVPA, and other alternatives are also reviewed with regard to their potential usefulness in providing possible ecosystem reference points.

Thus, to answer the question whether there is a need for extra reference points from an ecosystem perspective we will discuss the relevance of:

- reference levels assessed by various models;
- reference levels for community metrics and indicator species (target and non-target) on the basis of survey data.

1.2 Specific Reference Points Considerations

1.2.1 What ICES already advises

ICES considers a stock to be within safe biological limits if the spawning stock biomass (SSB) is above B_{PA}, and there is a low likelihood of SSB falling below B_{PA} in the medium term, at *status quo* fishing mortalities. B_{PA} plays a key role in ICES advice, as a risk control tool for B_{LIM}. Given the uncertainty in an assessment, advice intended to maintain the *estimate* of SSB above B_{PA} should ensure a high probability of keeping the *true* biomass above B_{LIM}. B_{LIM} is estimated in a variety of ways, but is generally considered to be the SSB below which recruitment is impaired (either the probability of poor recruitment is increased or the probability of good recruitment is decreased markedly). The total allowable catches (TACs) advised by ICES are based on fishing mortalities. ICES advises upper bounds on catches that would be consistent with the Precautionary Approach, but gives short- and medium-term forecasts (if possible) of the stock development at different exploitation levels. Options not consistent with the Precautionary Approach are

designated as such, but the ultimate responsibility of using a precautionary approach in setting the definitive level of a TAC is vested in the fisheries management agencies receiving advice from ICES.

The basis for setting single-species reference points for commercial species is developed in the reports of the Advisory Committee on Fisheries Management (ACFM) Study Group on the Precautionary Approach (ICES CM 1997/Assess:7) and the Comprehensive Fishery Evaluation Working Group (WGCOMP) (ICES CM 1997/Assess:15), and explained in each annual ACFM Report. The Multispecies Assessment Working Group (MAWG) compared single-species and multispecies approaches to estimating the biomass and fishing mortality reference points. They concluded that the theory of a precautionary approach should be elaborated to multispecies fisheries management. Multispecies interactions will affect the biological reference points and responses of populations to rebuilding strategies. The multispecies considerations make the reliability of single-species reference points more uncertain, and suggest even greater caution is necessary to achieve a low risk to the stock (ICES, 1997c).

While ICES made steady progress in developing precautionary reference points in the late 1990s, the implementation of ICES advice on single-species harvesting improved more slowly. Until early in the 2000's TACs for many stocks were set higher than ICES advised, and even to the present many stocks are fished harder than managers intend. Because of the difficulties in reducing the present intensity of fishing in many areas, conservation even of individual targeted stocks is at risk in many fisheries, and ICES has advised increasing numbers of closures (ICES 2003 – ACFM advice). Therefore, discussion of the possible benefits of fisheries management using reference points based on the state of the ecosystem rather than the states of individual harvested stocks is largely speculative. On the other hand, such a discussion might identify compelling reasons at the ecosystem level for fisheries management to practice greater caution.

To begin this speculative discussion the first question to pose is 'If all fisheries were managed so that there was a high probability of achieving conservation objectives for the target fish stocks, would there be a high likelihood of achieving conservation objectives for ecosystems?'. Current knowledge makes the answer to this question clearly 'No' for at least four reasons:

1) the genetic diversity of a target stock might be at risk, even in management regimes that complied with single-species reference points for biomass and fishing mortality (Section 1.2.2.1);

2) the conservation of non-target species could be at risk due to direct bycatch mortality from fishing activities (Section 1.2.2.2);

3) the conservation of dependent predatory species could be at risk due to local depletion of prey aggregations, even if conservation of the prey stock were being achieved on a much larger spatial scale (Section 1.2.2.3);

4) the conservation of some species could be placed at risk through the abundance of scavenging species increasing due to discarding in fisheries (Section 1.2.2.4).

It is not a coincidence that in all four of these situations the reference points which must be added are still single-species reference points. In those cases, the principles and criteria most closely parallel existing approaches to reference points for target stocks. However, WGECO stresses that the issue does not end with single-species reference points. The weight of scientific evidence suggests that there are additional reasons at the ecosystem level why the answer would be 'No.'. Examples of these reasons include documented changes to nutrient cycling and remineralization rates and pathways caused by impacts of fishing gear on substrates (Rowe *et al.*, 1975; Prins and Smaal, 1990) and diverse consequences on food web structure and function, caused by fisheries changing the absolute and relative abundances of target and non-target species (see Chapter 2). These types of risks, and their implications for reference points, are discussed in Section 2.5.

1.2.2 Additional reference points for species, from an ecosystem perspective

1.2.2.1 Genetic reference points for exploited stocks

In some studies it has been demonstrated that even short periods of intensive exploitation can alter the genetic make-up of an exploited population. Longer periods of exploitation, possibly at rates sustainable with regard to target stock size, may induce genetic responses as well (Lande, 1993; Stokes *et al.*, 1994; Waples, 1995). On a case-by-case basis, however, it is often problematic to differentiate phenotypic responses of life history or morphological traits from loss of genetic characteristics in the population (e.g., Rijnsdorp, 1993). Nonetheless, the loss of genetic diversity is a possible consequence of sustained or episodic intensive fishing, and it is not addressed in existing biological reference points based on biomass and fishing mortality. The Convention on Biological Diversity explicitly recognizes the need for management to conserve genetic diversity of stocks, so additional single-species reference points are necessary to fulfill this responsibility.

1.2.2.2   Reference points for non-target species

Despite a reduction in fishing mortality rate of commercial species, which would result from full implementation of the current management advice of ACFM, there may still remain unwanted effects for a number of reasons. Fisheries kill organisms other than the target species. The bycatch mortality can be unsustainable for a non-target species for two different reasons. First, direct exploitation may be too high. Commercial species may, by their nature, be more productive that the "average" marine species, and hence more resilient to exploitation. Some other less productive species, such as elasmobranches and cetaceans and some structure-building benthos, may only be able to withstand much lower mortality rates than the target fishing mortalities for directed fisheries (see Section 1.2.2.3). Even low levels of bycatch mortalities for some may require reference points for specific species such as some seabirds and marine mammals. This is because of their inability to withstand high mortality rates or their potentially high vulnerability to incidental mortality due to at least periodically forming very large aggregations. Hence, specific management targets should be set for the more vulnerable components of the ecosystem. Secondly, because the EU management sets single-species TACs, a fishery targeting a mix of commercial species may continue fishing, and thus generate additional mortality on commercial species, as long as not TACs are taken for some other species. ICES acknowledges this potential problem in the text of the annual advice, and management is moving to fishery-based rather than stock-based approaches (ICES 2003 – ACFM advice). However the estimation of and application of single-species reference points may have to include aspects of multispecies relationships explicitly to provide high likelihood of achieving conservation objectives of stocks taken in mixed fisheries. In the discussion below, these considerations will be developed for potentially relevant species.

Downward or upward trends in populations of many non-target species have been shown for the North Sea and other intensively fished areas (Heessen and Daan, 1996; Anon., 1997). Still not all these species are suitable as a potential reference point in an ecosystem consideration in fisheries management because, to be useful as a reference point, it is desirable to have a very well-defined and clear relation of stock status with fishing activities. Otherwise it will not be possible to formulate effective management measures. The status of top-predators, species which serve as main sources of food, structure-building organisms or representatives of a vulnerable group of species may be particularly useful as reference points. From recent ecosystem and fisheries research, two potential indicator species will be reviewed as an example of potential reference points, the harbour porpoise and the thornback ray.

The most abundant cetacean in the North Sea and the Baltic Sea is the harbour porpoise (*Phocoena phocoena)*. They are distributed throughout the North Sea, but are no longer present in the Southern Bight of the North Sea, the English Channel, or in much of the Baltic Sea. Incidental catches of harbour porpoise have been reported from almost every type of fishery in the North Sea, but bottom-set nets generate the great majority of harbour porpoise bycatch in the ASCOBANS area. Vinther (1994) estimated the annual bycatch in the Danish gillnet fisheries in the central and southern North sea at slightly more than 4500 animals.

A large shipboard and aerial survey (Small Cetacean Abundance in the North Sea, also known as SCANS) was made in 1994. The abundance of harbour porpoises in the North Sea, including the Channel and the Kattegat, was estimated at 304 000 (242 000–384 000) animals in 1994 (Anon., 1997). Of this total, the North Sea population of 170 000 occur in the central and southern North Sea. Genetic studies indicate this unit should be treated as a separate management unit. The harbour porpoise is specially protected under a number of international agreements and directives. The International Whaling Committee (IWC) recommends that a bycatch mortality rate of 1% should lead to research and expression of concern. Mortality exceeding 2% should lead to immediate implementation of management actions in order to reduce bycatch. For the central and southern North Sea, a maximum allowable bycatch of 3400 animals per year would be a sound ecological reference point related to fisheries. If this reference point was already operational, the current estimated annual bycatch of just a part of the fisheries in this region would exceed this biological reference point and effective management measures would be required immediately. Recent bycatch studies in the Celtic Sea estimated the fraction of harbour porpoises caught in fisheries to be 6.2% of the total population size which would also be nonsustainable (Tregenza *et al*., 1997). Equal use of the 2% bycatch of harbour porpoises in this area would lead to a maximum of 725 allowed bycatches per year for the Celtic Sea instead of the current estimated annual bycatch of 2200 animals (Tregenza *et al*., 1997).

A second example of a potential species for which an ecological reference point could be described is the thornback ray (*Raja clavata*). Rays and skates have a cartilaginous skeleton and, together with the sharks, belong to the group of elasmobranches. This group of species have life history strategies which fall in the realm of the so-called K-selected species of the classic r/K selection theory (Musick, 1999). This strategy consists of large adult size, late reproduction, and production of few, well-formed young, which makes the species vulnerable to additional mortality such as mortalities caused by fisheries. Rays and skates are a bycatch of demersal fisheries and all species have a commercial value except for the starry ray (*Raja radiata*), which is invariably discarded. Landings of all skate and ray species together decreased from around 18 000 t after both World Wars to the low level of 5000 t around 1975 and has remained at this level since. Taking into account the increase in fishing effort in the North Sea over recent decades, the decrease in biomass is even more severe (Rijnsdorp *et al*., 1996).

Not all ray species are equally affected by commercial fisheries and species can be classified according to their vulnerability to fishing based on information in age at maturity and fecundity (Table 1.2.2.1). Fisheries independent data confirm this. The common skate (*Raja batis*) has virtually disappeared from the North Sea between 1930 and the present, while the starry ray has increased in abundance and seems to stay within safe biological limits (Walker, 1996; Walker and Hislop, in press; ICES, 1997e). The thornback ray is the most common species at the fish market and although this species has virtually disappeared from Dutch and Belgian coastal areas (Walker, 1996), it is still resident along the British coast around the Wash and Thames estuary (Walker and Heessen, 1996; Rogers *et al.*, 1998). Historical tagging data has shown that this coastal area is important for mating and spawning (Walker *et al.*, 1997). The thornback ray may serve as a biological reference point because it is still abundant enough to collect statistically valid information.

Table 1.2.2.1   Life history characteristics of five resident North Sea ray species (table from ACFM, 1997).

	L_{inf}	L_{mat}	A_{mat}	Fec	$Z_{r=0}$	Z_{est}	Rank
Common skate *Raja batis*	237	160	11	40	0.38		1
Thornback ray *Raja clavata*	118	86	10	140	0.52	0.60	2
Spotted ray *Raja montagui*	79	62	8	60	0.54	0.72	3
Cuckoo ray *Raja naevus*	75	56	8	90	0.58	0.69	4
Starry ray *Raja radiata*	71	39	5	38	0.87	0.79	5

(L_{inf}: maximum length; L_{mat} and A_{mat}: length and age at first maturity, respectively; Fec: number of eggs produced per year; $Z_{r=0}$: maximum mortality that species is able to withstand; Z_{est}: estimated level of mortality based on recent survey catches; Rank: ranking in decreasing order of vulnerability).

In the North Sea, the thornback ray is caught as bycatch in demersal fisheries. Fishing mortalities of commercial species are high, ranging from 0.5–0.8 or even higher. Since the catchability of rays is high for these kinds of fisheries, similar fishing mortality rates can be expected. But thornback rays are known to form local subpopulations (Walker and Heessen, 1996). These do not have to coincide with the areas where the demersal fisheries put their highest effort. A reference point for the thornback ray should take into account these spatial aspects.

Based on the life history strategy characteristics, the maximum total mortality the thornback ray population is able to withstand, $Z_{r=0}$, is calculated at 0.52 (Table 1.2.2.1). In order to ensure the continued existence of the thornback ray in the North Sea, the total mortality in areas where sub-populations of thornback ray still occur should be kept below a level of 0.52. Tag experiments show that thornback rays are resident and do not migrate over large distances (Walker *et al.*, 1997). This supports the effectiveness of area-specific measures. ICES already advises to limit the impact of demersal fisheries particularly in those areas where the species still occurs, this may be necessary to protect the stock in the North Sea (ICES, 1997e).

Thus, area-specific maximum mortality seems a suitable and effective reference point for the thornback ray. For accurate estimation of fishing mortality, a major and controllable part of the total mortality, improved data on landings (species specific), discards (juveniles), and disturbance of eggs by demersal gears is necessary, and requested by ICES (ICES, 1997e). With this kind of information it is possible to formulate the most effective fisheries measures in the areas of concern.

1.2.2.3   Reference points for ecologically dependent species

For some years CCAMLR has explored the important role of krill in the Antarctic ecosystem. The breeding success and even survivorship of a number of predators, including several species of seabirds and marine mammals, is affected greatly by the status of krill (Laws, 1984; Croxall and Prince, 1987). Correspondingly, the requirements of these ecologically dependent predators plays a major role in the management of krill fisheries in that region (SC-CAMLR, 1992). Recently the Scientific Working Group of CCAMLR reviewed what would be a precautionary approach to the management of krill fisheries, in light of the expanding ideas about the precautionary approach and progress in the development of reference points. The associated analyses indicate that although a precautionary overall catch limit is necessary for large geographic areas, that limit is not sufficient to safeguard some of the dependent predators. A management approach is proposed which requires geographic subdivison of the overall catch according to varying requirements of predator populations, and uses information on predator populations and their physiological needs in setting harvest levels (Everson and de la Mare, 1996). The proposal does not go as far as proposing specific biological reference points for the ecologically related species and relating those reference points directly to krill management. However, the approach lends itself directly to those developments, and such reference points may be forthcoming in future publications from CCAMLR scientists.

Closer to home, ICES has received requests for advice about possible management measures which might be necessary to protect local aggregations of sandeels near sensitive wildlife concentrations. This issue is discussed in

depth in Chapter 3 and thus will not be reviewed here. The request clearly stems from the same concern; there may be ecologically related species whose conservation is not assured by a management approach that places the stock being targeted at negligible risk overall. Also, the fishery for capelin in the Barents Sea is managed under an approach which gives the feeding requirements of cod (and other predators?) priority over human harvests.

Specific types of biological reference points have not been proposed for such ecologically related species, nor have the links between the reference points and specific management actions been specified. Nonetheless, in at least a few cases, such as colonial seabirds and their prey fish stocks, cod and capelin, and Antarctic top predators and krill, the relationships have been studied extensively, and the management needs are recognized. The knowledge base might be an adequate foundation for development, testing, and implementation of such reference points linked among species.

1.2.2.4   Reference points for species affected by scavengers feeding on discards and offal

Populations of many scavenging seabirds have grown in recent years (e.g., Lloyd *et al.*, 1991). Some of this growth may be due to recovery following a long period of persecution which ended in the early part of the current century, but it is likely that much of the growth of the populations of some species is due to the increased food supply deriving from fishery wastes (e.g., Fisher, 1952; Furness and Barrett, 1985). This growth appears to be continuing in many populations.

Owing to the requirement of seabirds to breed in areas that are free (or virtually free) of mammalian predators that can take eggs or young, there is frequently competition for the limited habitat that meets this requirement. In many cases, this leads to displacement either into nearby suboptimal habitat or away from the area entirely (Howes and Montevecchi, 1993). This displacement in many cases may not be desired by local wildlife managers (and may locally reduce biodiversity). Many of the tern species have been shown to have been displaced by larger gull species (Theissen, 1986; Becker and Erleden, 1986). This has led in many instances to the culling of the large gulls in order to allow terns to return to their original nesting sites (Wanless, 1988; Wanless *et al.* 1996). In Shetland, the great skua population has grown rapidly and was feeding on both sandeels and fishery waste. The availability of sandeels has declined around the Shetland Islands (trends in discard amounts are not known), and the great skua population has now switched to depredating seabirds and their young (Heubeck and Mellor, 1994). Previous regulation of the availability of offal and discards might have limited the growth of the population of great skuas.

Fisheries managers might thus consider reference points addressing discards and offal deriving from fishing operations.

1.2.2.5   Summary of reference points at the species level

Suppose that biologically sound reference points for genetic diversity were added to the existing B and F reference points for target species, and that reference points were also identified for all non-target species and for species ecologically dependent on aggregations being fished. Furthermore, suppose that fisheries complied with these reference points, such that there was a high likelihood of achieving all single-species conservation objectives. Would conservation and sustainability of the ecosystem be achieved with at least an equal likelihood? If the answer to this core question is 'No', there are two ancillary questions. First, what multispecies properties might still be at an unacceptable level of risk? Second, how should these properties be monitored and/or modelled, in order to identify and evaluate the effectiveness of actions taken to reduce the risk?

1.2.3   Biological reference points from an ecosystem perspective

The answer to the first question, raised in Section 1.2.2.5, is that we do not know if conservation and sustainability of the ecosystem as whole would be achieved. There is certainly no empirical demonstration of an ecosystem property that would be at risk, if fisheries management where conducted in ways which placed no constituent species individually at risk, and did not degrade habitat structure. However, the book is not yet closed on this issue.

We do know that without question fishing has changed the size composition of fish in some, possibly many, exploited systems (Pope and Knights, 1982; Pope *et al.*, 1988; Dayton *et al.*, 1995), and in the North Sea in particular (ICES 1996a; Rice and Gislason, 1996). Regardless of the trophic model considered, changing the size composition of predators in the ecosystem has, with high likelihood, changed the way that predation pressure is distributed among lower trophic levels in the ecosystem. The uncertainty is in the magnitude of the change, and its consequences for the ecosystem. We also know that the flux and residency of nutrients within the system must also have changed, as the numbers and biomasses at different trophic levels as well as features of benthos have changed (Rowe *et al.*, 1975; Prins and Smaal, 1990). Again, it is the magnitude and ecosystem consequences which are uncertain. Even if present knowledge is inadequate to answer the first question, it is adequate to highlight that a truly precautionary approach with the possible consequences, as outlined below, should be of serious concern.

A number of multispecies or ecosystem models have been developed which can be used to investigate this question. At this time, though, different models make very different predictions about ecosystem consequences (or lack thereof) of changing the distribution of predation pressure among sizes (and undoubtedly species) of prey. We also know too little about the flux of nutrients at lower trophic levels, and among the benthic, pelagic, and demersal parts of the ecosystem, to know even how the flux of nutrients has changed as a result of reducing the numbers and biomasses of large predators, let alone the consequences of the changes. Therefore, it is premature to draw inferences about impacts of changes in size composition of predatory fish on the sustainability and conservation of the larger ecosystem as a unit, and on the larger question of the need for additional precautionary reference points.

Primary production in marine ecosystems away from the coastal zone are generally controlled by the availability of nutrients and usually nitrogenous forms. In stratified regions, the rate controlling step is the regeneration of nutrients by zooplankton and fish excretion of ammonia. In vertically well-mixed areas, the flux of nutrients from the benthos is also important, decomposers in the benthos being responsible for the ammonification of organic nitrogen, and the reduction of nitrate to ammonia (Sørensen, 1978). High productivity of coastal waters may be dependent on this benthic-pelagic coupling (Rowe *et al.*, 1975). The flux rate of this coupling is dependent on the biological activity in the sediments and, in particular, the nature of the benthic fauna (Prins and Smaal, 1990; Josefsen and Schlüter, 1994). Fishing has the potential to alter these rates by (i) alterations in the benthic fauna, (ii) re-suspension of benthic materials by towed bottom gears, (iii) alterations in the chemical status of bottom sediments, e.g., exposure of anoxic materials, and (iv) alterations in the size of the various food web compartments.

Although we cannot evaluate the likelihood of achieving ecosystem-level objectives using a strategy of achieving all single-species conservation objectives, we do note some important considerations with regard to ecosystem-level reference points. First, it is well established that the dynamics of individual stocks and populations connected trophically contain time lags and buffers (e.g., age structure, density-dependent growth) which can slow down the rate at which the consequences of perturbations of a food web may be manifest. Therefore, we may not yet be observing the full impacts on the ecosystem of past levels of fishing. Moreover, if there were to be changes in major ecosystem properties, most models suggest the changes could be difficult and slow to reverse, and would aggravate the loss in total yield of fish, beyond the yield already foregone due directly to overfishing the target stocks.

Although we are not in a position to recommend that ecosystem reference points are necessary, beyond the reference points which would assure sustainability and conservation of all populations killed directly by fishing, neither are we prepared to confirm that single-species reference points are enough to ensure a precautionary approach. This is a complex problem, with important implications, and much more investigation of model (and ecosystem) dynamics is required. For example, although WGECO has clearly documented that the slope of the biomass spectrum of the North Sea has changed over the past 20 years, we cannot advise what a maximum tolerable slope would be, what a 'good' target slope would be, or even if these are reasonable concepts to consider.

A commitment to a precautionary approach to fisheries management and conservation of biodiversity has to include a commitment to pursue these types of questions much further. Relevant programmes would have to identify:

a)   what ecosystem properties require more than just the conservation of the individual component species?
b)   which of the properties in a) could be placed at risk by fisheries?
c)   what management measures would be necessary to have a high likelihood of achieving conservation of the properties in b)?
d)   how could the properties potentially at risk be measured and monitored?

Some of these questions have fuelled research and debate among community ecologists for decades, and quick resolutions are unlikely. Future meetings of WGECO could address the state of knowledge on these questions more intensively, but would require attendance by diverse specialists, and the opportunity to focus significant time on these questions. However, WGECO stresses that the need for some ecosystem level reference points is real. Even if different theoretical frameworks suggest different properties for ecosystem level reference points (often just because the different frameworks use different biological 'currencies'), in internally consistent ways, every framework indicates that such properties exist (see Section 1.3).

1.3   Models that may give insight

In relation to fisheries impacts, much of the discussion on the implications of using the precautionary approach has focused on how to define target and limit reference points using traditional single-species fisheries models to make predictions of impacts on target species (e.g., ICES CM 1997/Assess:15; ACFM Report, 1997, Part I).

The International Whaling Commission uses single-species models to provide advice on sustainable levels of harvest of cetaceans. The nominal catch limits derived by the revised management procedure (RMP) are based on a comprehensive specification of data requirements in terms of catch history and abundance estimates, the algorithm for

calculating catch limits, including a specification of the population model to be used, how it is fitted to the data, and well defined rules specifying how uncertainty should be taken into account (IWC, 1993). A similar approach has been proposed for small cetaceans in the North Sea (Bravington *et al.*, 1997).

For seabirds and benthos, reference points have not been set and, in particular for benthos, the present knowledge has, with few exceptions, not yet crystallized into models which could readily be used to predict consequences of fishing for individual species or assemblages.

WGECO has previously used the concept of potential jeopardy as a common yardstick to identify particularly vulnerable species in relation to fisheries generated mortality. This approach is closely related to the approach followed in fisheries management where limit reference points in relation to spawning stock biomass such as B_{LIM} have been used. Potential jeopardy is defined as the additional mortality needed to decrease the spawning stock biomass of a certain species to a specific level, say 5% or 10%, of its virgin unfished value. The concept can be applied to calculate the vulnerability of individual species across taxonomic groups. It depends only on life history parameters of the particular species, i.e., on growth, mortality, and age or size at first maturity. However, data to estimate the actual mortality imposed are seldom available and little is known about how life history parameters for particular species would respond to changes in the physical environment, in the amount of food available, and in the abundance of their predators.

Less effort has been spent on investigating how reference points could be defined by models which allow the species to interact. Multispecies fish stock models include species interaction in the form of fish predation and are available for some areas, but have rarely been used for providing management advice. Some of the multispecies models have been extended to include marine mammals and seabirds. Often this has been in terms of the impact mammals and seabirds have on commercially exploited species, only very rarely has the reverse question been asked. At present, the models are therefore of limited use for defining reference points in relation to fisheries generated food limitation for seabirds and marine mammals. However, simpler models have been used to estimate exploitation levels on prey species which take the needs of their predators into account, e.g., the models used to arrive at precautionary catch limits for krill in the Antarctic (Everson and de la Mare, 1996).

Few models describe how community or ecosystem properties would change in response to fishing, and often the existing metrics, such as species diversity indices or slopes of size spectra, are difficult to connect to the perceived state of the affected system. For this reason, such metrics have not yet been used to define limit and target reference points. The models that are available describe either overall metrics such as the slope of the size composition of the fish assemblage, or consider energy flow among trophic compartments. Of the latter type, mass-balance models, such as ECOPATH (Section 1.3.2), offer a range of possible measures that could be used for defining reference points. Another possibility is to utilize more conceptual tools, such as trophic cascade models (see Section 1.3.3). However, in both instances, the challenge is not to derive the metric, but to relate it to changes in the affected system of relevance to society.

1.3.1 Extensions of MSVPA/MSFOR

At its 1996 meeting, the ICES Multispecies Assessment Working Group (MAWG) discussed how to derive reference points in a multispecies context (ICES, 1997c). Several modelling approaches were investigated including classical Lotka-Volterra models, MSVPA/MSFOR approaches, and single-species models with changes in natural mortality due to predation. The investigations demonstrated that reference points derived from single- and multispecies models can be expected to differ and, in particular, that single-species reference points will often tend to be less conservative (and less precautionary) than their multispecies equivalents.

At this meeting, an extended version of the Baltic multispecies spreadsheet MSFOR-type model used at the MAWG meeting was available. The model includes cod, herring, and sprat in the central Baltic and performs a 32-year prediction of the biomass and yield of the three species with an annual time step. The relationship between spawning stock and recruitment is of the Ricker type, and the model includes a description of how growth and maturity of cod changes in response to changes in the amount of available food. The input data are derived from the database used by the Working Group on Multispecies Assessment of Baltic Fish (ICES, 1996b, 1997f, 1997a) (residual natural mortality, fishing mortality, suitabilities, weight-at-age, maturity-at-age, recruitment). The model predictions should therefore be in reasonable accordance with similar predictions made by the MSFOR used by the Working Group on Multispecies Assessment of Baltic Fish even though this model operates with a quarterly time step. However, the model parameters describing changes in growth as a function of available food have not yet been estimated from retrospective runs. At the present stage, the model is therefore intended as a conceptual tool which can be used to demonstrate how competition and predation will affect precautionary reference points and not as a model from which management advice can be directly derived.

The model is able to run in three different modes corresponding to the classical single-species fisheries model (constant natural mortality and growth for all species), the ordinary multispecies model (MSFOR including cod as a predator on herring, sprat, and young cod, with constant weight-at-age for all species), and an extended multispecies model where the amount of herring, sprat, and other food available will influence cod food intake, growth, and maturity-at-age. The extended version was made in order to take the large changes in cod weight-at-age observed over the period 1977–1996 into account, assuming that these changes were due to changes in the food supply of cod. Figure 1.3.1.1 shows how the average weight-at-age for ages 2 to 4 changed from between approximately 30% below the long-term average to 10% to 30% above the long-term average at the end of the period. Figure 1.3.1.2 shows the change in average weight-at-age versus cod biomass.

In the single-species version, recruitment to all of the species are modelled by Ricker curves, with parameters estimated from historic values of stock and recruitment. Natural mortality is constant at values equal to the sum of predation and other natural mortality (M1) in the multispecies *status quo* situation. In the multispecies models, cod recruitment at age 0 is assumed to be directly proportional to spawning stock biomass. Subsequent changes in cannibalism changes the number of cod surviving to age two. Survival is thus lower at high levels of adult cod biomass producing a stock-recruitment relationship similar to the Ricker model used in the single-species case.

In the ordinary multispecies model cod is predating on herring and sprat as well as on their own young. The amount of other food available to cod is assumed to be constant irrespective of a change in cod biomass and intake of other food.

In the extended multispecies model, the annual growth of cod is assumed to be directly proportional to the amount of food available. The biomass of other food is modelled by a surplus production model of the Fox type (Biomass of other food = $1/q * \exp(a + b*$ Biomass of cod)) where the cod's intake of other food in the *status quo* situation and the value of other food assumed in the ordinary multispecies run (30 million t) are used to estimate the q and b parameters, and the constant, a, is fixed at a value producing a biomass of other food which is 10 million t higher in a situation without cod predation. The latter value was adopted because it produced what appears to be sensible values for cod weight-at-age at high biomasses. In the *status quo* situation, the parameters are such that the weight-at-age of cod corresponds to the weight-at-age used in the single-species and ordinary multispecies models. Changes in weight-at-age will influence the proportion mature at age. Based on historic data on maturity and weight-at-age, the relationship between weight and maturity-at-age is modelled by Maturity = $(1 - \exp(-c*W))^{\wedge}d$, where W is weight and c and d are constants.

The fishery is controlled by two variables, 'cod effort' and 'pelagic effort', that are used to multiply the fishing mortalities for cod and for herring and sprat, respectively. In the *status quo* situation where both effort variables are set to 1.0, the average fishing mortality for cod ages 3–7 equals 0.82, while for sprat ages 3–5 and herring ages 3–8 the *status quo* fishing mortalities equal 0.15 and 0.27, respectively.

The initial population numbers in the starting year are set equal to the long-term equilibrium population sizes in the *status quo* situation in order to ease comparisons between this situation and a change in the fisheries. The results from a run where both fisheries were closed (cod effort and pelagic effort both reduced to 0.001) are presented in Figures 1.3.1.3 to 1.3.1.5. A closure is predicted to lead to damped oscillations in spawning stock biomasses resulting in a long-term increase in the biomass of cod and a long-term decrease in the spawning stock biomasses of herring and sprat. Cod weight-at-age will decrease, and so will the proportion mature at age.

The average yield and spawning stock biomass of cod predicted in each of the three models are shown in Figure 1.3.1.6 for various levels of cod effort. Pelagic effort was fixed at 1.0 and the values presented in the figure are averages over the last 10 years of the 32-year prediction period. In the *status quo* situation, the predictions of the three models are identical. When cod effort is decreased from the present level, the biomass and yield of cod increases. This increase is most pronounced in the single-species prediction, less so for the ordinary multispecies mode where recruitment is reduced by cannibalism, and even less for the extended multispecies prediction, where the increase in cod biomass is counteracted both by cannibalism and by reductions in weight-at-age with knock-on effects on maturity and recruitment.

The model was used to examine how biomass reference limits might be derived in a multispecies context. Figure 1.3.1.7.a–1.3.1.7.c show plots of the regions of combinations of 'cod effort' and 'pelagic effort' that produces spawning stock sizes for all three species above or below 10% of their unfished levels (calculated by closing both fisheries) in each of the three modes of the model, e.g., an SSB for cod of 2.0, 1.4, and 0.9 million t in the single, ordinary multispecies and extended multispecies modes, respectively.

The single-species results are shown in Figure 1.3.1.7.a. The area within which the spawning stock biomass of all three species is above 10% of the unexploited level forms a rectangle in the lower right corner of the plot. The present situation (both effort multipliers = 1.0) is right at the upper border of the area. Pelagic effort can be increased to

between two and three times its present level before the herring SSB will fall below the reference limit. Increasing both efforts above the limits will generate an area where only the sprat SSB is above the limit.

In the ordinary multispecies predictions, the limits for the pelagic species become curved, Figure 1.3.1.7.b. The amount of pelagic effort which can be exerted without reducing the pelagic species below the limit now depends on the effort in the cod fishery. If cod is reduced to low levels by an intensive fishery, it is possible to increase the effort in the pelagic fisheries to approximately four times the present level before herring falls below the limit. If the cod fishery is closed and the cod stock increases, the reference limit for herring is close to the present level of effort.

In the extended multispecies case, all of the reference limits are curved, Figure 1.3.1.7.c. In this case, the limits for cod become dependent on the amount of pelagic effort. If pelagic effort is high, cod can sustain less effort before it exceeds the limit. This is because of a reduction in cod growth and proportion mature. If there is plenty of food for cod, i.e., large stocks of herring and sprat, cod will grow faster and mature earlier, and hence tolerate a more intensive fishery.

Species interactions will alter reference points and limits. Reference points for fisheries on forage fish cannot ignore changes in the biomasses of predators feeding on these species. Reference points for fisheries on predators cannot be set without considering how the predators are influenced by the simultaneous exploitation of their prey.

1.3.2 Mass-balance models

A number of metrics based on mass-balance models of trophic interactions in ecosystems are of potential relevance for developing reference points from a multispecies perspective. These include:

> the trophic level of the fishery in an ecosystem;
> the transfer efficiency between trophic levels;
> Finn's cycling index;
> the primary production required to sustain fishery catches;
> mixed trophic impact analysis of the ecosystem, with the fishery as impacting and impacted component.

Details of these metrics are discussed in Chapter 2, Section 2.2.

At present, no recommendations can be made as to how these analytical tools can be used for the definition of reference points. However, their step beyond single-species fisheries management towards explicitly considering the multispecies context in which the fishery operates may contribute to future ecosystem management.

1.3.3 Trophic cascade models

The central role of fish in limnic ecosystems, especially their influence on food web structures, has been known since the early 1960s (Hrbacek, 1962; Brooks and Dodson, 1965; review in Hansson, 1985). In the 1980s, research in this field increased significantly (e.g., see Carpenter, 1988; review by Northcote, 1988) and the concept of cascading trophic interactions (Carpenter *et al*., 1985) has been heavily discussed (e.g., Carpenter and Kitchell, 1988; McQueen and Post, 1988a, b; Leavitt *et al*., 1989; Brönmark *et al*., 1992; Martin *et al*., 1992; Christoffersen *et al*., 1993; Schindler *et al*., 1993). By cascading trophic interactions we mean that, e.g., a top predator like a piscivorous fish does not only influence the ecosystem by reducing the abundance of its prey, but also indirectly influences the food organisms of this prey. For example, if the prey fish is an important zooplanktivore, the predation by the top predator may reduce the predation pressure on zooplankton. The effects of the predation from the top predator cascades down the food web: the decreased predation on zooplankton may allow these to increase in abundance and hence increase the grazing pressure on phytoplankton. These trophic dynamics generally follow the classical food web interaction concept of Hairston *et al*. (1960).

Most of our present knowledge on the role of fish in aquatic ecosystems, in particular their significance as predators and their influence on trophic cascades, derive mainly from studies in lakes. For marine ecosystems, these ecological interactions are much less understood. This is probably because marine systems are more difficult to study than lakes. Compared to the seas, lakes are well defined and geographically delimited ecosystems. Furthermore, there are thousands of lakes with different food web structures that can be compared to evaluate the consequences of these differences. The relative lack in our understanding of the role of fish marine food webs does not, however, imply that the significance of fish predation is less than in freshwater. Nixon (1982) actually showed that at a given primary production, fish yields from marine systems are generally higher than those from freshwater systems. This implies that marine food webs are at least as tightly coupled as those of freshwaters and that fish predation are also central in structuring marine ecosystems.

Several studies have shown that fish predation on zooplankton is intensive in marine systems (e.g., Fulton, 1983; Kimmerer and McKinnon, 1989; Hansson *et al*., 1990; Hassel *et al*., 1991; Hopkins and Gartner, 1992; Rudstam *et al*., 1992; Arrhenius and Hansson, 1993; Luo and Brandt, 1993). There are also a number of articles which describe ecological effects of fish predation on organisms other than their prey and hence supports the presence of cascading trophic interactions or other complex ecological population dynamics processes in marine ecosystems (Skjoldal, 1989; Springer, 1992; Rudstam *et al*., 1994; Parsons, 1991, 1992, 1996; Anon., 1996; Verity and Smetacek, 1996; Shiomoto *et al*., 1997; Hansson *et al*., in press). A direct implication of these results is that the intensive fishery for many common marine fish species is likely to influence marine ecosystem structures, and not only decrease the abundances of the target fish species.

Trophic cascading models have been successful in describing the responses of lower trophic levels of lacustrine systems to perturbations at upper levels. With suitable development of this application in marine systems, this type of model might become useful as a tool for identifying fishing strategies which have a high risk of causing amplified perturbations at lower trophic levels than those being fished. The associated ecosystem reference points might be tolerance limits on perturbations that fishing could impose on any single trophic level or on the suite of levels in the system being modelled. An example of a possible ecosystem reference point is that the relationship between abundances of piscivorous fish and their forage species must be kept within certain limits. Hence, a goal in fisheries management should be to avoid not only growth and recruitment overfishing (Cushing, 1975), but also ecosystem overfishing (i.e., ecosystem changes that drastically change trophic interactions, food web structures, nutrient cycling, etc.).

1.4   Concluding Remarks

This section has been developed by starting from existing practice and asking what must be added. WGECO concluded that one necessary addition to present practice is reference points for non-target species, as developed in Section 1.2. WGECO also concluded that the task does not stop here. WGECO notes that, implicitly, present practice assumes that explicit conservation objectives have been set by management agencies, to justify the development of even the reference points used at present. As recent ICES advice makes clear, even that assumption is not absolutely true. Nonetheless, in endorsing the precautionary approach, governments and management agencies have clearly committed to conservation of all species directly or indirectly affected by fishing (FAO, 1995; Garcia, 1996). Much of the internal debate within WGECO centered on what additional commitments are implicit in this approach, because there are strong theoretical reasons to expect that certain ecosystem properties may be altered by fishing activities.

Will society (and biology) be served by objectives to conserve particular configurations of an ecosystem being fished? Do the diverse international agreements summarized by FAO (1995) <u>require</u> such objectives to be adopted? What does it mean for an ecosystem to be 'at risk', and can an ecosystem be 'at risk' if the species which comprise it are not? Although WGECO looks forward to exploring these fundamental questions at future meetings, it stresses that they must be discussed in many other fora as well, both within and outside ICES.

1.4.1   References

Anon. 1996. The Bering Sea ecosystem. National Academy of Sciences Press, Washington D.C. 307 pp.

Anon. 1997. Assessment report on Fisheries and Fisheries related Species and Habitat Issues.

Arrhenius, F., and Hansson, S. 1993. Food consumption of larval, young and adult herring and sprat in the Baltic Sea. Mar. Ecol. Prog. Ser. 96:125–137.

Becker, P.H., and Erdelen, M. 1986. Die bestandsentwicklung von Brutvogeln der deutschen Nordseekuste seit 1950: Aspekte fur den Artenschutz (Trends of coastal bird populations of the German Wadden Sea since 1950: aspects relevant to species conservation). Berichte der Deutschen Sektion des Internationalen Rats für Vogelschutz 26: 63-73.

Bravington, M.V., Northridge, S.P., Hammond, P.S., and Palka, D.L. 1997. Criteria for assessing the status of harbour porpoises in the *North Sea* and adjacent waters: A suggested way forward. *In* Forty-Eighth Report of the International Whaling Commission (Covering the Forty-Eighth Financial Year 1996–1997), pp. 272–286. Annual Report, International Whaling Commission, Vol. 48.

Brönmark, C., Klosiewski, S.P., and Stein, R.A. 1992. Indirect effects of predation in a freshwater, benthic food chain. Ecology, 73: 1662–1674.

Brooks, J.L., and Dodson, S.I. 1965. Predation, body size, and composition of plankton. Science, 150: 28–35.

Carpenter, S., and Kitchell, J. 1988. Consumer control of lake productivity. Bioscience, 38: 764–769.

Carpenter, S.R. 1988. Complex interactions in lake communities. Springer-Verlag, New York.

Carpenter, S.R., Kitchell, J.F., and Hodgson, J.R. 1985. Cascading trophic interactions and lake productivity. Bioscience, 35: 634–639.

Christoffersen, K., Riemann, B., Klysner, A., and Søndergaard, M. 1993. Potential role of fish predation and natural populations of zooplankton in structuring a plankton community in eutrophic lake water. Limnology and Oceanography, 38: 561–573.

Croxall, J.P., and Prince, P.A. 1987. Seabirds as predators on marine resources, especially krill, at South Georgia. Pp. 347–368. *In*: Croxall, J.P. (Ed.) Seabirds Feeding Ecology annd Role in Marine Systems. Cambridge University Press.

Cushing, D. H. 1975. Marine ecology and fisheries. Cambridge University Press, Cambridge. 278 pp.

Dayton, P. K., S. F. Thrush, *et al*. 1995. 'Environmental-Effects of Marine Fishing.' Aquatic Conservation-Marine and Freshwater Ecosystems, 5(3): 205–232.

Doulman, D.J. 1995. Structure and Process of the 1993–1995 United Nations Conference on Straddling Fish Stocks and Highly Migratory Fish Stocks. FAO Fisheries Circular. No. 898, Rome, FAO. 1995. 81 pp.

Everson, I., and. de la Mare, W.K. 1996. Some thoughts on precautionary measures for the krill fishery. CCAMLR Science, Vol. 3(1996): 1–11.

FAO. 1995. Precautionary approach to fisheries. Part 1: Guidelines on precautionary approach to capture fisheries and species introductions. Elaborated by the Technical Consultation on the Precautionary Approach to Capture Fisheries (Including Species Introductions). Lysekil, Sweden, 6-13 June 1995 (A scientific meeting organized by the Government of Sweden in cooperation with FAO). FAO Fisheries Technical Paper No. 350. Part 2. Rome, FAO, 1995. 52 pp.

Fisher, J. 1952. The fulmar. Collins, London. pp.

Fulton, R.I. 1983. Interactive effects of temperature and predation on an estuarine zooplankton community. Journal of Experimental Marine Biology and Ecology, 72: 67–81.

Furness, R.W., and Barrett, R.T. 1985. The food requirements and ecological relationships of a seabird community in North Norway. Ornis Scandinavica, 16: 305–313.

Garcia, S.M. 1996. The Precautionary Approach and its Implications for Fisheries Research, Technology, and Management: An Updated Review. Pages 1-65 *in* FAO. 1995. Precautionary approach to fisheries. Part 2: Scientific Papers Prepared for the Technical Consultation on the Precautionary Approach to Capture Fisheries (Including Species Introductions) Lysekil, Sweden, 6-13 June 1995 (A scientific meeting organized by the Government of Sweden in cooperation with FAO). FAO Fisheries Technical Paper No. 350. Part 2. Rome, FAO, 1995. 210 pp.

Hairston, N., Smith, F., and Slobodkin, L. 1960. Community structure, population control, and competition. American Naturalist, 94: 421–425.

Hansson, S. 1985. Effects of eutrophication on fish communities, with special reference to the Baltic Sea - a literature review. Report, Institute of Freshwater Research Drottningholm, 62: 36–56.

Hansson, S., Arrhenius, F., and Nellbring, S. In press. Food web interactions in a Baltic Sea coastal area. Forage fishes in marine ecosystems, Proceedings from the Alaska Sea Grant College Program.

Hansson, S., Larsson, U., and Johansson, S. 1990. Selective predation by herring and mysids, and zooplankton community structure in a Baltic Sea coastal area. Journal of Plankton Research, 12: 1099–1116.

Hassel, A., Skjoldal, H.R., Gjøs'ter, H., Loeng, H., and Omli, L. 1991. Impact of grazing from capelin (*Mallotus villosus*) on zooplankton: a case study in the northern Barents Sea in August 1985. Polar Research, 10: 371–388.

Heessen, H. J. L., and Daan, N. 1996. Long term trends in ten non-target North Sea fish species. *ICES Journal of Marine Science*, 53: 1063–1078.

Heubeck, M., and Mellor, R.M. 1994. Changes in breeding numbers of Kittiwakes in Shetland 1981-1994. Scottish Birds, 17: 192–204.

Hopkins, T.L., and Gartner, J.V. 1992. Resource-Partitioning and Predation Impact of a Low-Latitude Myctophid Community. Marine Biology, 114: 185–197.

Howes, L.A., and Montevecchi, W.A. 1993. Population trends and interactions among terns and gulls in Gros Morne National Park, Newfoundland. Canadian Journal of Zoology, 71: 1516–1520.

Hrbacek, J. 1962. Species composition and the amount of zooplankton in relation to the fish stock. Rozpr CSAU, Ser mat nat sci, 72: 1–117.

ICES. 1996a. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 1996/Assess/Env:1.

ICES. 1996b. Report of the Working Group on Multispecies Assessment of Baltic Fish. ICES CM 1996/Assess:2.

ICES. 1997a. Report of the Baltic Fisheries Assessment Working Group. ICES CM 1997/Assess:12.

ICES. 1997b. Report of the Comprehensive Fishery Evaluation Working Group. ICES CM 1997/Assess:15.

ICES. 1997c. Report of the Multispecies Assessment Working Group. ICES CM 1997/Assess:16.

ICES. 1997d. Report of the Study Group on the Precautionary Approach to Fisheries Management. ICES CM 1997/Assess:7.

ICES. 1997e. Report of the ICES Advisory Committee on Fisheries Management. *ICES Cooperative Research Report No. 223*.

ICES. 1997f. Study Group on Multispecies Model Implementation in the Baltic. ICES CM 1997/J:2.

IWC. 1993. Forty-Third Report of the International Whaling Commission. Rep. Int. Whal. Commn. 43.

Josefsen, S.B., and Schlüter, L. 1994. The influence of an intertidal mussel bed (*Mytilus edulis* L.) on nutrient fluxes in the Kerteminde Fjord, Denmark; a flume study. *In*: Changes in fluxes in estuarieds: Implications from science to management, K.R. Dyer and R.J. Orth (eds.). Olsen and Olsen, Fredensborg, pp. 249–256.

Kimmerer, W., and McKinnon, A. 1989. Zooplankton in a marine bay. III. Evidence for influence of vertebrate predation on distribution of two common copepods. Marine Ecology Progress Series, 53: 21–35.

Lande, R. 1993. Risks of population extinction from demographic and environmental stochasticity, and random catastrophes. American Naturalist, 142: 911–927.

Laws, R.M. (Ed.). 1984. *Antarctic Ecology* (2 volumes). 717 pp.

Leavitt, P.R., Carpenter, S.R., and Kitchell, J.F. 1989. Whole-lake experiments: The annual record of fossil pigments and zooplankton. Limnology and Oceanography, 34: 700–717.

Lloyd, C.S., Tasker, M.L., and Partridge, K. 1991. The status of seabirds in Britain and Ireland. Poyser, London.

Luo, J.G., and Brandt, S.B. 1993. Bay Anchovy *Anchoa mitchilli* Production and Consumption in Mid-Chesapeake Bay Based on a Bioenergetics Model and Acoustic Measures of Fish Abundance. Marine Ecology Progress Series, 98: 223–236.

Martin, T.H., Crowder, L.B., Dumas, C.F., and Burkholder, J.M. 1992. Indirect effects of fish on macrophytes in Bays Mountain Lake - evidence for a littoral trophic cascade. Oecologia, 89: 476–481.

McQueen, D., and Post, J. 1988a. Limnocorral studies of cascading trophic interaction. Verhandlungen - Internationale Vereinigung für Theoretische und Angewandte Limnologie, 23: 739–747.

McQueen, D. and Post, J. 1988b. Cascading trophic interactions: uncoupling at the zooplankton- phytoplankton link. Hydrobiologia, 159: 277–296.

Musick, J.A. 1999. Ecology and conservation of long-lived marine animals. *In:* Life in the slow lane: ecology and conservation of long-lived marine animals. Ed. by J.A. Musick. Am. Fish. Soc. Symp. 23:1–10.

Nixon, S.W. 1982. Nutrient dynamics, primary production and fisheries yields of lagoons. Oceanologica Acta, 357–371.

Northcote, T. 1988. Fish in the structure and function of freshwater ecosystems: a 'top-down' view. Canadian Journal of Fisheries and Aquatic Sciences, 45: 361–379.

Parsons, T.R. 1991. Impact of fish harvesting on ocean ecology. Marine Pollution Bulletin, 22: 217.

Parsons, T.R. 1992. The removal of marine predators by fisheries and the impact of trophic structure. Marine Pollution Bulletin, 25: 51–53.

Parsons, T.R. 1996. The impact of industrial fisheries on the trophic structure of marine ecosystems. *In* Food webs: integration of patterns and dynamics, pp. 352–357. Ed. by G. A. Polis and K. O. Winemiller. Chapman and Hall, New York.

Pope J.G., Stokes, T.K., Murawski, S.A., and Idoine, S.I. 1988. A comparison of fish size composition in the North Sea and on Georges Bank. *Ecodynamics: Contributions to theoretical ecology*, W. Wolff, C.-J. Soeder, and F.R. Drepper (eds.). Springer-Verlag, Berlin, pp. 146–152.

Pope, J.G., and Knights, B.J. 1982. Comparison of length distributions of combined catches of all demersal fishes in surveys in the North Sea and at Faroe Bank. *In:* Multispecies approaches to fisheries management advice. Ed. by M.C. Mercer. Canadian Special Publication in Fisheries and Aquatic Science, 59: 116–118.

Prins, T.C., and Smaal, A.C. 1990. Benthic-pelagic coupling: The release of inorganic nutrients by an intertidal bed of *Mytilus edulis*. *In*: Trophic interactions in the marine environment. M. Barnes and R.N. Gibson (eds.). Aberdeen University Press, Aberdeen. pp. 89–103.

Rice, J.C., and Gislason, H. 1996. Patterns of change in the size spectra of numbers and diversity of the North Sea fish assemblage, as reflected in surveys and models. *ICES Journal of Marine Science*, 53: 1214–1225.

Rijnsdorp, A. D., Van Leeuwen, P. I., Daan, N., and Heessen, H.J.L. 1996. Changes in abundance of demersal fish species in the North Sea between 1906-1909 and 1990-1995. ICES Journal of Marine Science, 53: 1054–1062.

Rijnsdorp, A.D. 1993. Fisheries as a large scale experiment on life history evolution: disentangling phenotypic and genetic effects in changes in maturation and reproduction of North Sea plaice, *Pleuronectes platessa* l. Oecologia, 96: 391–401.

Rogers, S.I., Rijnsdorp, A.D., Damm, U., and VanHee, W. 1998. Demersal fish populations in the coastal waters of the UK and continental NW Europe from beam trawl survey data collected from 1990-1995. Netherlands J. Sea Res.

Rowe, G.T., Clifford, C.H., Smith jr., K.L., and Hamilton, P.L. 1975. Benthic nutrient regeneration and its coupling to primary productivity in coastal waters. Nature, 255: 215–217.

Rudstam, L.G., Aneer, G., and Hildén, M. 1994. Top-down control in the pelagic Baltic ecosystem. Dana, 10: 105–129.

Rudstam, L.G., Hansson, S., Johansson, S., and Larsson, U. 1992. Dynamics of planktivory in a coastal area of the northern Baltic Sea. Marine Ecology Progress Series, 80: 159–173.

SC-CAMLR 1992. CCAMLR Ecosystem Monitoring Program: Standard Methods. CCAMLR, Hobart, Australia. 237 pp.

Schindler, D.E., Kitchell, J.F., He, X., Carpenter, S.R., Hodgson, J.R., and Cottingham, K.L. 1993. Food web structure and phosphorus cycling in lakes. Transactions of the American Fisheries Society, 122: 756–772.

Shiomoto, A., Tadokoro, K., Nagasawa, K., and Ishida, Y. 1997. Trophic relations in the subarctic North Pacific ecosystem: Possible feeding effect from pink salmon. Marine Ecology Progress Series 150:75–85.

Skjoldal, H.R. 1989. Does capelin influence the growth of phytoplankton in the Barents Sea ecosystem. Abstracts for the 1989 annual meeting of the American Society of Limnology and Oceanography.

Sørensen, J., 1978. Capacity for denitrification and reduction of nitrate to ammonia in a coastal marine environment. *Applied Environmental Microbiology* 35: 301–305.

Springer, A.M. 1992. A review: Walleye pollock in the North Pacific - how much difference do they really make? Fisheries Oceanography 1: 80–96.

Stokes, T.K., Law, R., and McGlade, J. (eds.). 1994. *The Exploitation of Evolving Resources.* Springer Verlag, Berlin, Germany.

Theissen, H. 1986. Zur bestandsentwicklung und situation von Mowen (Laridae) und Seeschwalben (Sternidae) in Schleswig-Holstein - sowie gedanken zum 'Mowenproblem' (Population dynamics of gulls (Laridae) and terns (Sternidae) in Schleswig-Holstein – and thoughts on the so-called gull-problem). Seevogel 7: 1–12.

Tregenza, N. J. C., Berrow, S. D., Hammond, P. S., and Leaper, R. 1997. Harbour porpoise (*Phonoeca phonoeca* L.) bycatch in set gillnets in the Celtic Sea. *ICES Journal of Marine Science*, 54: 896–904.

Verity, P.G., and Smetacek, V. 1996. Organism life cycles, predation, and the structure of marine pelagic ecosystems. Marine Ecology Progress Series, 130: 277–293.

Vinther, M. 1994. Incidental catches of harbour porpoise (*Phonoeca phonoeca*) in the Danish North Sea gill-net fisheries: preliminary results. Proceedings of the Scientific Symposium on the North Sea Quality Status Report 1993. 18–24 April 1994, Ebeltoft, Denmark: 210–213.

Walker, P. A. 1996. Ecoprofile Rays and Skates on the Dutch Continental Shelf and North Sea. NIOZ/RWS Report No. 3053. 69 pp.

Walker, P. A., and Heessen, H. J. L. 1996. Long-term changes in ray populations in the North Sea. *ICES Journal of Marine Science*, 53: 1085–1093.

Walker, P. A., Howlett, G., and Millner, R. 1997. Distribution, movement and stock structure of three ray species in the North Sea and the eastern English Channel. *ICES Journal of Marine Science*, 54: 797–808.

Walker, P.A., and Hislop, J.R.G. In press. Sensitive skates or resilient rays? Spatial and temporal shifts in ray species composition in the central and north-western North Sea between 1930 and the present day. *ICES Journal of Marine Science*.

Wanless, S. 1988. The recolonisation of the Isle of May by common and arctic terns. Scottish Birds 15: 1–8.

Wanless, S., Harris, M.P., Calladine, J., and Rothery, P. 1996. Modelling responses of herring gull and lesser black-backed gull populations to reduction of reproductive output: implications for control measures. Journal of Applied Ecology 33: 1420–1432.

Waples, R.S. 1995. Evolutionary significant units and the conservation of biological diversity under the Endangered Species Act. American Fisheries Society Symposium, 17:8–27.
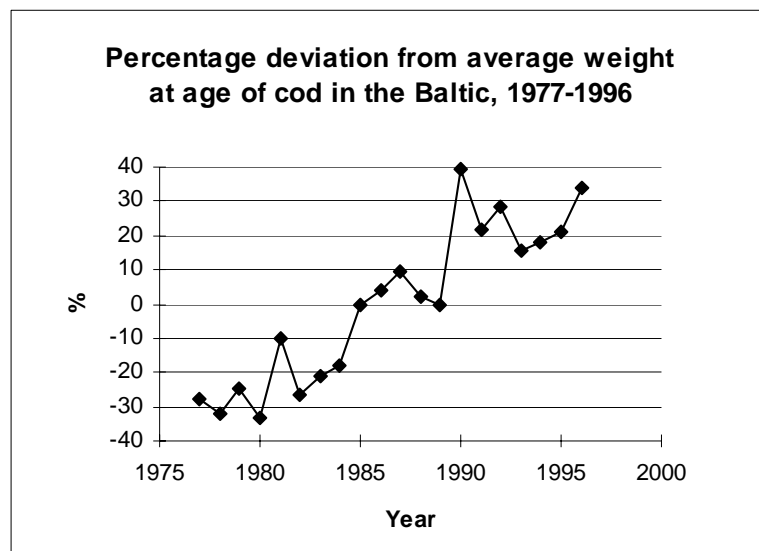
Figure 1.3.1.1   Percentage deviation from average weight-at-age of cod in the Baltic Sea, 1977–1996. Data from ICES CM 1997/J:2.
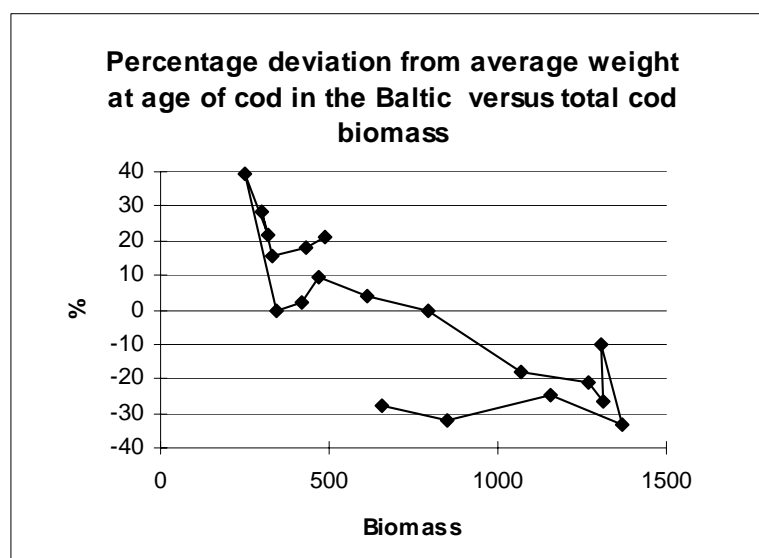
Figure 1.3.1.2   Percentage deviation from average weight-at-age of cod in the Baltic versus total biomass, 1977–1996. Data from ICES CM 1997/J:2.
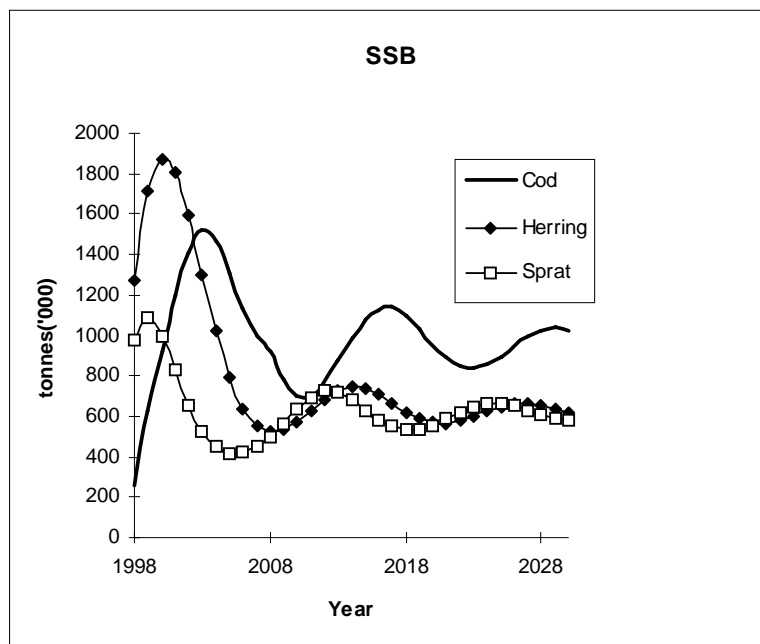
Figure 1.3.1.3   Predicted SSB of cod, herring and sprat after a closure of all fishing. Output from multispecies model with dynamic cod growth.
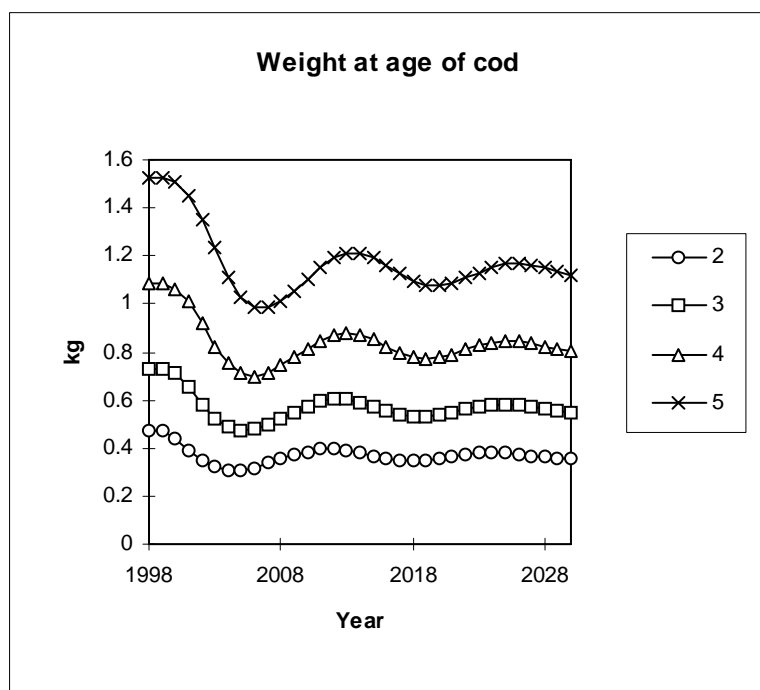
Figure 1.3.1.4   Predicted change in weight-at-age of cod after a closure of all fishing. Output from multispecies model with dynamic cod growth.

Cod. Proportion mature at age in 1998 and 2030
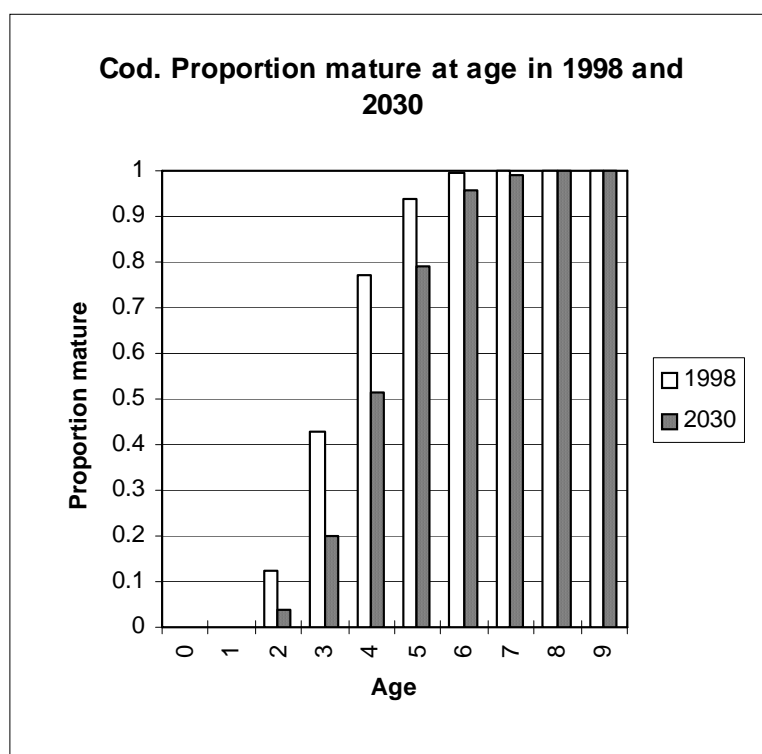
Figure 1.3.1.5 Predicted change in percent mature at age of cod after a closure of all fishing. Values for 1998 correspond to *status quo* fishing, values for 2030 to final year of prediction. Output from multispecies model with dynamic cod growth.
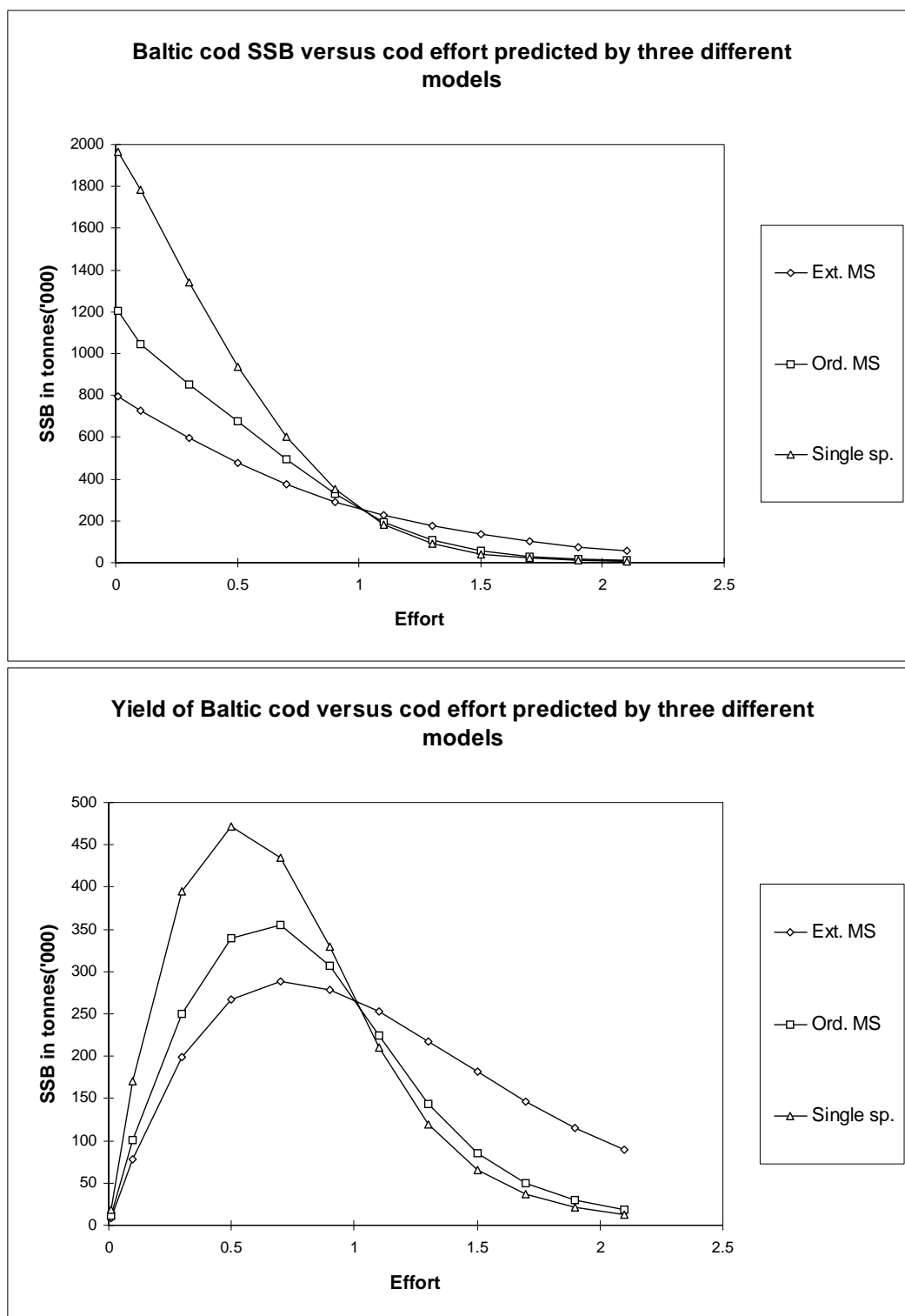
Figure 1.3.1.6 Average SSB and yield of Baltic cod predicted by the single-species, ordinary multispecies, and extended multispecies versions of the spreadsheet model.
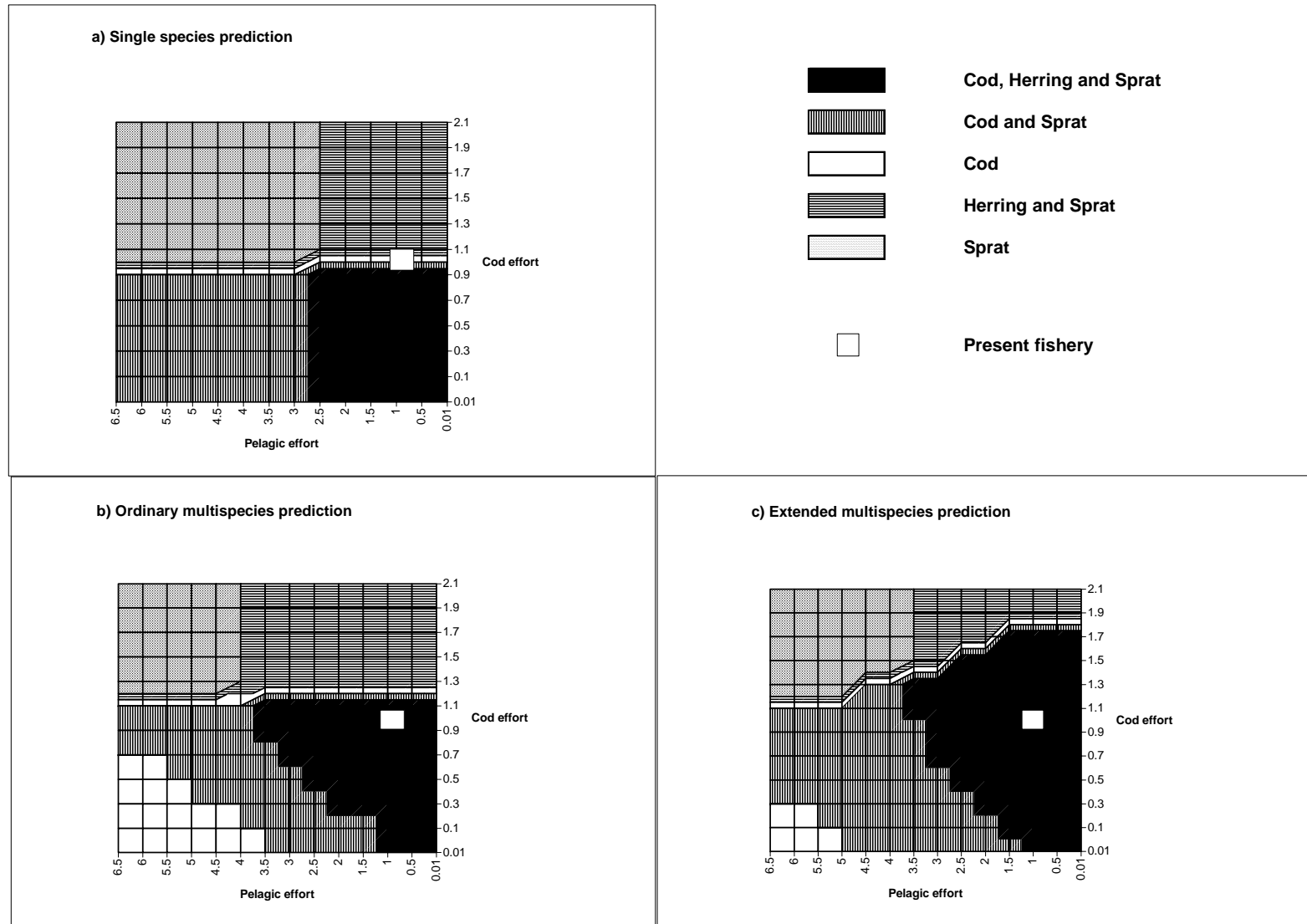
Figure 1.3.1.7   Combinations of effort levels in the cod and pelagic fisheries in the Baltic resulting in equilibrium SSBs above 10% of the unexploited SSBs (a) for cod, (b) for herring, and (c) for sprat. Predictions assuming single-species, ordinary multispecies, and extended multispecies model structures.

2 Ecosystem Management Objectives

2.1 Introduction

WGECO begins its consideration of ecosystem management objectives with an acknowledgement that these are longer-term considerations. Under present conditions management of fishing effort at levels which deliver a high probability that conservation objectives are achieved for the target stocks (i.e., $SSB > B_{pa}$) is likely to be the single biggest change that would ensure conservation of the ecosystem. This is especially so if combined with targeted protection of key habitats/features. Looking ahead, however, OSPAR and the North Sea Conference of Ministers consider the implementation of an ecosystem approach in fisheries management as an important step for the integration of fisheries and environmental issues. Since there is not yet a clear and agreed upon definition of an ecosystem approach (NRC, 1999), the approach taken in this chapter will be along several lines. There are a growing number of documents which describe the features of ecosystem management approaches (e.g., Anon. 1995; Christensen *et al.*, 1996; Lanters, 1999) reference points for ecologically dependent species (species that are so tightly linked ecologically to the target species.

In this section we place the initial work on reference points for ecosystem conservation into the context of other areas of the ecosystem where management may be required to achieve sustainability and where reference points may be defined. Such considerations implicitly recognise the need for integrated management of the marine environment and that managers will have to operate with multiple management criteria. Such multiple management criteria are now an accepted part of fisheries management in multispecies fisheries such as in the North Sea, even if methods for simultaneously meeting them all are not perfected.

In developing an ecosystem scale management perspective it must be recognised that the objectives set will include much wider considerations than those traditionally addressed for fisheries management. The overall ecosystem objective should involve sustainability. Sustainability means different things to different people. We take it to mean that current activities do not compromise the ability of the environment to provide resources and services in the future, nor reduce the choices available to future generations. Further, we should recognize that with regard to fisheries there are three aspects to sustainability:

- Sustainable fisheries. The level, and composition, of landings are sustainable.
- Sustainable fishing industry. This is the socio-economic sustainability of fishing and includes considerations of the viability of communities dependent on fisheries, the size and nature of the fishing industry and all linked economic and social activities—including merchants and fish processing sectors, chandlers, vessel building and repair, etc.).
- Sustainable ecosystems. The nature, species composition and functioning of the environment are not placed at risk of changes that seem long lasting and difficult to reverse.

It is not for scientists to advise on the balance among these three, but such a consideration must form an explicit part of any ecosystem management scheme. It should however be recognised that a number of existing international agreements (Table 2.1.1) already place a priority on sustaining the ecosystem, arguing that pursuit of social and economic sustainability cannot be allowed to result in an unacceptable risk to conservation of the ecosystem.

Any ecosystem approach to management must also have mechanisms for dealing with the inherent uncertainty in predictions of marine system dynamics. The application of a precautionary approach to fisheries management has seen advances in recent years but these will need to be developed and extended if any management scheme based on an ecosystem approach is to be effective. In particular, admonitions that uncertainty about the status of single-species cannot be used as a reason to defer cost-effective measures to reduce risk, must be expanded to acknowledge the greater uncertainty about ecosystem status and trajectory.

WGECO stresses that science has to deal with the complexity of the marine system that includes thousands of species and many different types of habitats. The degree of mutual coherence is poorly known and predictive scientific models are not, and may never be, available. In addition, human use may already have changed the most sensitive components of the marine system, hampering identification of reference levels. If any changes are observed in ecosystems it is important to differentiate between changes that form part of natural variability and those that represent the effect of one or more human activities. It is in this context that operational reference points are considered for species, habitats, genetics, and emergent properties of ecosystems. For this analysis each class of properties is

approached in the same manner; we first ask how are the properties placed at risk by fishing, and then what objectives would protect those properties.

Table 2.1.1   An overview of the main global conventions, laws and treaties applying to the conservation and management of marine living resources. These are often given regional specificity in 'local' conventions such as Annex V of the OSPAR Convention which covers protection of species and habitats.

Convention or treaty	Year	Main objective
UN Law of the Sea	1982	Regulation of the management and authority of all living marine resources. Establishment of an Exclusive Economic Zone
Bonn Convention	1983	Protection of migratory stocks of wild species (species moving across national borders)
CITES and GATT		General Treaties governing prevention of trade in endangered species (CITES) on reduction of environmental impact (GATT)
Convention on Biological Diversity (CBD)	1992	Result of UNCED Conference. Protection of biodiversity at level of genetics, species and ecosystems
Agenda 21 - Chapter 17	1992	Result of UNCED Conference. Protection of all marine and coastal areas by rational use and development of living resources
FAO Code of Conduct	1995	Code of Conduct for Responsible Fisheries by considering ecosystem and socio-economic aspects of fisheries and the precautionary approach
Jakarta Mandate	1997	Elaboration of CBD for marine systems in which Marine Protected Areas form a major issue
UN Convention on Migratory and Straddling Fish Stocks	not in force	Conservation and protection of border crossing and high seas fish stocks

2.2   Population and Species Reference Points/Objectives

What is at risk and how do fisheries place them at risk?

2.2.1   Populations of target and non-target species

If improperly managed, fisheries can place **populations of both target and non-target species** at risk, through inflicting unsustainable mortality over periods of time long enough to impact abundance. The mortality can be severe enough to cause a population decline directly, to spawning biomasses at which either the probability of good recruitment is reduced or the probability of poor recruitment is increased. These are the criteria presently used by ICES to decide if a stock is inside or outside safe biological limits.

Where fisheries inflict less severe mortality, the fishery will change **the age composition of the stock** relative to the unexploited condition. The changes may be great enough that spawning biomass comprises disproportionately first-time spawners or total biomass may depend excessively on new recruits. Neither of these changes is desirable, as there is evidence that for at least some species first-time spawners have lower reproductive value on a per kilogram basis (Trippel, 1998), and dependence of biomass in incoming recruitment makes the stock more vulnerable to short-term periods of poor recruitment or environmental stress. Hence, reference points even for target species should ensure a suitable age composition as well as adequate total spawning biomass and sustainable fishing mortality.

Without being killed, target or non-target species may suffer **injury or exposure which results in increased vulnerability to predation**. This can result from physical damage as gear passes over individuals or as individuals pass through gear, or from rough handling and release. Once injured or exposed, if predators are present, the biological effect is much the same as for direct mortality. Hence, the seriousness of this effect would be evaluated in the same way as for direct fishing mortality: is the total death rate sustainable and is biomass being conserved?

NOTE: For all the direct effects above, from a biological perspective, and according to the international agreements reviewed in Table 2.1.1, ICES is concerned about the conservation of all species. Hence we ask the same questions about the sustainability of all populations in the face of total mortality and the contribution of fishing

mortality to total mortality. **There is no justification to apply different standards to species of commercial and non-commercial importance.**

The direct mortality due to being killed by fishing gear can become excessive if effort is too high, either overall, or in the area where the species suffering unsustainable mortality is concentrated. Injury or exposure by gear that results in increased vulnerability to predators can jeopardise conservation of a species if a biologically important fraction of a population encounters the gear and is not retained.

2.2.2 Spatial properties

Fishing can successively deplete meta-populations so that even if local subpopulations are not demonstrated to be genetically distinct, the species or stock ceases to be present in progressively larger parts of its historic range. Although special circumstances would be required, it is theoretically possible that a population as a whole could be above its biomass reference point and experiencing total mortality still below the mortality reference point, yet the fishery could be causing a reduction in range. The key circumstances include intense localized exploitation and low mobility of the species being killed. There are several reasons that managers should take safeguards that fisheries do not cause major reductions in range. It has been theorized that a species (or stock) becomes less resilient to environmental challenges as distribution contracts, if only by becoming more vulnerable to catastrophes (Tuljapurkar, 1990). Some studies have conjectured that a reduction in spawning area reduces reproductive potential by not allowing full seeding of larval/juvenile habitats (Burgman *et al.*, 1993; Groom and Pascual, 1998). Also as a population becomes spatially concentrated, q (catchability to fishing gear) goes up and the stock becomes more vulnerable to further overfishing, even when fleet behaviour has not changed.

Reduction in range or in meta-population structure can occur if a fishery is not distributed representatively across the full range of the species of concern, and redistribution of the species is slow relative to its population dynamics responses to fishing mortality.

2.2.3 Dependent species

Fishing can deplete a population locally so dependent predators cannot find sufficient food to survive or reproduce at sustainable rates, even though the stock as a whole is within safe biological limits, and the population genetic diversity may not be compromised. Evidence for this effect, and reasons to be concerned about it, are reviewed in ICES (1999b).

Conservation of ecologically dependent species can be jeopardized if the fishing fleet is more mobile than the dependent species and the prey is widely distributed but slowly mobile. Given those two factors, a fishery may cause local depletions of prey for periods of time that are long relative to the needs of the dependent species, if the fishery concentrates harvests disproportionately in areas important to the dependent predator.

2.2.4 Scavenger-caused effects

Fishing can produce so much waste that species which feed on offal and discards can increase greatly in abundance. The incidental mortality that the scavenging species inflict on alternate prey may become unsustainable, or through competition for limited space the scavenging species may cause reproduction below replacement rates for the species displaced from breeding (or other) sites. Evidence for this effect is reviewed in Section 2.6.1.4.

Fishing produces wastes (discards and offal) which can be concentrated and readily available as food for scavengers who can exploit this food source. If the scavengers also prey on species that cannot use this food supply, or compete with them for breeding space, then a fishery that increases food to scavengers may cause mortality or poor recruitment of species who are eaten or out-competed by scavengers.

2.3 Habitat Features

What is at risk and how do fisheries place them at risk?

Marine habitats are generally distinguished by the physical nature of the environment; e.g., silty-mud is distinct from muddy-sand, frontal regions separating mixed and stratified waters. These can include biologically produced features such as reefs and turf. Changes in the nature, extent and spatial distribution (degree of patchiness) of habitat features can compromise the ability of the ecosystem to support a natural species assemblage and hence normal ecosystem function (Dayton *et al.*, 1995; NRC, 1999).

There are limited data on the impact of fishing on habitats within EU waters (see Section 2.6.2.1). In addition to the impacts recognised from bottom trawls (Section 2.6.2), there are data which suggest that deep-water fisheries to the west of Scotland, around the Faroe Islands, and in northern Norway have caused substantial damage to beds of the cold-water coral *Lophelia*, and data also indicate damage to *Sabellaria* reefs in coastal waters of the North Sea and Irish Sea

(STECF, 1999). In the northwest Mediterranean, changes in the size and species composition of fish populations caused by fisheries may have led to large changes in benthic communities as a result of increased abundance of sea urchins (Sala *et al.*, 1998). This is an example of a habitat modification mediated through changes in the food web.

Section 2.6.2.2 presents the conclusions about aspects of marine habitats which may be put at risk by fishing. They are repeated briefly here.

Bottom-towed gears can remove some physical features

Bottom-towed gears may cause the loss of physical features in the environment such as peat banks, boulder reefs, or gravel banks. These changes are <u>always</u> permanent, and lead to an overall reduction in habitat diversity. This in turn can lead to the local loss of species and species assemblages dependent upon such features. Examples might include attached bryozoan/hydroid turf and essential fish habitat such as herring spawning grounds. Even when substantial quantities of the habitat feature remain, if the habitat has become highly fragmented, this may compromise the viability of populations dependent upon it.

Bottom-towing of gears can cause a reduction in structural biota (biogenic features)

Loss of structure-forming organisms such as colonial bryozoans, *Sabellaria*, hydroids, sea-pens, sponges, mussel beds, and oyster beds can result from the impact of bottom-towed gears. These changes <u>maybe</u> permanent, and lead to an overall loss of habitat diversity. This in turn can lead to the local loss of species and species assemblages dependent upon such biogenic structures. Essential fish habitat such as juvenile gadoid nursery habitat would be an example. Even when substantial quantities of the biogenic feature remain, if the feature has become highly fragmented, this may compromise the viability of populations or species dependent upon it.

Bottom-towed gears can cause a reduction in complexity

Towing of bottom fished gears can cause the redistribution and mixing of surface sediments as well as degradation of habitat and biogenic features. This can lead to a decrease in the physical patchiness of the sea floor (i.e, decreased heterogeneity) within fishing grounds. These changes are not likely to be permanent.

Bottom-towed gears alter the physical structure of the sea floor

Towing of gears on the sea floor can cause a reshaping of seabed features such as sand ripples and damage to burrows and associated structures (e.g., mounds and casts, microhabitats). These features provide important habitats for smaller animals such as meiofauna.

2.4   Genetic Properties of Populations

What is at risk and how do fisheries place them at risk?

Total genetic variation within a species can be partitioned into variation within and among populations. Fisheries may have consequences for both types. Within populations, phenotypic changes associated with fisheries are well documented for a number of species and include changes in morphological and life history traits such as weight- and length-at-age, and age- and length-at-maturity, spawning time, etc., (e.g., Rijnsdorp, 1993; Rowell, 1993; Millner and Whiting, 1996; Trippel *et al.*, 1997), many of which may be correlated (ICES, 1997). Such changes may arise through relaxation of intra-specific competition, response to shifts in environmental conditions (phenotypic plasticity) and to change in genetic composition; it is often difficult to establish which of these effects is responsible for the observed response. To the extent that the changes are genetically based, intensive selective fishing will result in changes in gene frequencies, and possibly in loss of alleles within the exploited populations.

Populations that are reproductively isolated, with little or no gene flow between them, will tend to diverge genetically either through different selective forces or through genetic drift. Salmonids have high among-population variance resulting from their homing behaviour at spawning time (e.g., Gharrett and Smoker, 1993). However, even in species that have free-drifting larvae, gametes or spores (approximately 70% of marine invertebrate species have pelagic larvae; Mileikovsky, 1971) and are ultimately distributed over a wide area, local populations can often be discerned (e.g., cod: Ruzzante *et al.*, 1997; squid: Shaw *et al.*, 1999; marine algae: Van Oppen *et al.*, 1996). In such species, loss of sub-populations results in loss of the unique characteristics of the genome of the sub-population.

Natural selection acts within populations, while the genetic potential of the species to adapt to environmental changes depends on the total genetic diversity represented among populations. It is necessary to maximize both types of variation to maintain full potential for evolutionary change within a species. In general, modelling studies have shown

that size-selective fishing favours slow-growing and late-maturing fish, although there are exceptions to this (ICES, 1997).

Fishing mortality is a highly selective process, both with respect to the size of the organism captured and location (ICES, 1997). The fishery may also directly or indirectly favour capture of one sex over another (e.g., American lobster, shrimp), altering the sex ratio and/or sex-specific size frequency of the breeding population. In addition, migratory stocks may be under different selection pressures in different parts of their range due to different fishing methods. Fishing therefore has the potential to affect the genetic diversity and genetic structure of a species.

Selective breeding programmes for cultured fish (e.g., salmon) and invertebrates (e.g., abalone) have shown that significant amounts of genetic heritability (the proportion of phenotypic variation that is inherited from one generation to the next) exist for yield-related traits. Life-history traits, being closely linked to fitness, have relatively lower heritabilities; however, even these are capable of showing a substantial selection response in only a few generations (ICES, 1997). Although extrapolation of heritability estimates determined from breeding programmes to those in wild fish stocks should not be made, this research has demonstrated clearly that there is genetic variation in those traits selected for by fishing.

The stronger the selectivity (in the fishery sense 'selective') of the fishery for certain traits, and the greater the proportion of total mortality made up of fishing mortality, the greater will be the effect of fishing on the genetics of the exploited population. The persistence of fishing-induced genetic changes will depend upon the other selective forces operating on the species, the proportion of genetic diversity affected and the reproductive biology of the species. In some cases, genetic change may not be readily reversed by altering fishing practices (Law and Grey, 1989). Consequently, fishing can cause evolution of phenotypic traits of the exploited species (Law and Rowell, 1993), although the time scale over which it operates is unknown. Fishing can also selectively harvest some sub-populations intensively, while harvesting other sub-populations lightly. In these cases, a rate of fishing mortality which is sustainable at the scale of the whole species may successively eliminate isolated sub-populations, and reduce the total genetic variability of the stock or species.

2.5   Emergent Properties of Ecosystems

2.5.1   Emergent properties: What are they?

In the previous section we considered ecosystem level reference points. Discussions within WGECO highlighted issues such as:

- o   food web dynamics;
- o   species richness and evenness (diversity);
- o   distribution of life histories;
- o   production:biomass ratios.

These are not direct biological properties but are functions of the entire ecosystem and are referred to as emergent properties. They are important not only because they may tell us something about the functioning/status of the ecosystem, but also as they have been widely perceived as indicators of environmental status.

2.5.1.1   Does fishing put emergent properties at risk?

There has been considerable speculation as to the extent to which fishing may alter these emergent ecosystem properties (see ICES (1998a) and the previous section of this report). It is also true that many press and popular articles have been highly emotive in their commentary on this issue. We have reviewed the **evidence** that has emerged since our last consideration and can find none which would cause us to revise our conclusions.

WGECO stresses that the need for some ecosystem objectives and corresponding reference points is real. **At this time WGECO believes that we are not in a position to recommend that objectives and reference points for ecosystem emergent property are necessary, beyond those which would assure sustainability and conservation of all species and habitats impacted by fishing. Neither are we prepared to confirm that single-species, habitat and genetic objectives and reference points alone are enough to ensure a precautionary approach to ecosystem management. Some study may yet provide compelling evidence that objectives for emergent properties of ecosystems are also required to ensure conservation of the ecosystem, but to this time none have.**

2.6   Objectives and Reference Points for Management

Ecosystem approaches to marine management will require many objectives and reference points. Exceeding any reference point, whether for target species, non-target species, habitat change or genetic health, should invoke mitigation measures intended to increase the likelihood of achieving the relevant objective.

2.6.1   Populations and species

2.6.1.1   Direct mortality

For target species of fisheries conservation can be achieved by following the precautionary approach. Special importance should be given to two activities. One is setting B_{pa} and F_{pa} sufficiently far from the biological limits to allow for uncertainty in estimates of present biomasses and fishing mortalities, and uncertainty about the future states of nature (especially, but not exclusively, future recruitment) for the time scale of management and the degree of risk aversion managers (and society) demand. The other is implementing harvest control rules, to ensure that necessary conservation measures are implemented in a timely way when a reference point is violated. Together, these measures should keep target species inside safe biological limits with high probability (ICES, 1998b). Occasionally the biology of a species makes an escapement goal or a total mortality a more appropriate reference point than an exploitation rate, but those circumstances are well understood (ICES, 1999b).

For non-target species there is no reason to take a different approach to assuring conservation. The implementation problem is the practical impossibility of setting biomass and fishing mortality reference points for every non-target species in the ecosystem, and then assessing compliance. As a practical solution we propose setting objectives with biomass and fishing (or total) mortality reference points for non-target species of high vulnerability, and monitoring their compliance. This proposal assumes that the documented conservation of a set of non-target species of high vulnerability gives high probability of also ensuring conservation of other non-target species of lower vulnerability. We suggest that vulnerability should be evaluated with regard to:

o   the ability of the species to tolerate an increase in mortality (see Section 2.2.1): long-lived species of low fecundity are likely to be more vulnerable than short-lived species of high fecundity, controlling for factors such as likelihood of exposure to specific gears);

o   the likelihood that the gear will encounter the species (there should be a relatively high probability of exposure to the gear);

o   the likelihood that an encounter with the gear will kill or injure the species (species which are soft or brittle may be more vulnerable than species with hard shells or leathery epidermis);

o   the proportion of the population which is in the area where the fishery operates (a large part of the species' range should lie within the area of activity of the fishery on macro [geographic] and micro [habitat] scales);

o   it must be possible to quantify at least the sign of the trend of the population, and ideally more;

o   moreover, because most population trends are likely to be affected by several factors as well as fishing (Daan *et al.*, 1996), it will often be important to monitor several areas with substantial contrast in fishing intensity.

2.6.1.2   Range

For objectives addressing reduction of range and loss of population structure, the same reasoning applies with regard to the impossibility of assessing all species and the need to select species whose conservation is likely to ensure conservation of less vulnerable species. Within the field of ecology there is significant debate and conflicting data about the relationship between population size and range occupied (see MacCall, 1990; Fretwell, 1972). The current weight of evidence suggests that it is not appropriate to generalize that a reduction in range necessarily corresponds to a decline in abundance. Nonetheless, it is a symptom which warrants investigation when observed (e.g., Baltic cod, Section 4.2). The assumption that a reduction in range corresponds to a reduction in abundance may be safer for moderately sedentary species than for highly mobile ones, particularly if the mobile species routinely migrate extensively and opportunistically. Therefore, the assumption may be appropriate for many benthic species.

Even for the sedentary species information is usually lacking regarding the dependence of local recruitment on local spawning. Hence, there is likely to be controversy about the scale at which a documented effect should trigger a management action, that is, about the value of the precautionary reference point for range reduction, given that an objective related to maintaining the range distribution of a species has been adopted. The properties characteristic of a good candidate species for setting objectives and reference points regarding range reduction vary with the mobility of the species.

For species which are moderately sedentary, appropriate properties include:

o   presence and abundance can be quantified well with properly designed monitoring programmes, including the use of proper statistical approaches to analysing change in infrequent observations, if the species is uncommon;

o information linking fishing to the loss or depletion of local populations is sound—this often requires evidence of direct mortality, physical injury from gear combined with increased risk of predation, or loss of essential habitat features for the species caused by fishing gear (use of the latter type of evidence also presupposes knowledge of essential habitat for the species);

o it possesses at least some of the characteristics of vulnerability discussed under direct mortality;

o it is desirable, but not essential, that there be some knowledge of the degree to which local recruitment depends on local population status;

o Even for species with these characteristics, it usually will NOT be clear what decline in range should be used as a reference point to trigger management action. Baillie and Groomsbridge (1996) and CITES (1994) have adopted range criteria, but these were developed for species with population dynamics of birds and mammals. Many sedentary benthos may be viewed more like plants, and there is substantial debate about the shape of the functional relationship linking change in range to change in abundance and threat to conservation.

If a species is quite mobile, appropriate properties include:

o factors affecting changes in distribution are known. Ideally, this includes not just knowledge of typical migration patterns, but also some understanding of how migration routes and timings, and areas occupied during a season, change with environmental conditions such as temperature, salinity, oxygen, etc.;

o change in range can be documented with appropriate quantitative methods. These must reflect the uncertainty in spatial distribution appropriately, if the reference points are to have a sound relationship to degree of risk aversion;

o there should be plausible links (with some documentation) between fishing depleting local populations (the proximate mechanism could be either direct mortality or loss of essential habitat) or fishing reducing population numbers and the decline in abundance resulting in contraction of range.

Even with the above information available for a candidate species, it often will not be clear what decline in range should be used as a reference point to trigger management action. Because of at least differences in dispersal properties of reproductive propagules, criteria developed for birds and mammals may not be appropriate for mobile marine species. Present knowledge of the spatial dynamics of most mobile marine species is inadequate to state how large a decrease in range corresponds to a marked increase in likelihood that the population is suffering unsustainable mortality. Moreover, the functional relationship of abundance to range is likely to be non-linear and have species-specific parameters which could vary with migration habits, diets, and life history parameters, and be difficult to parameterize. Therefore the step from adopting a conceptual objective protecting the ranges of species in an ecosystem to operational reference points on measurable indicators of range or area occupied, may be very difficult.

2.6.1.3   Ecologically dependent species

For **ecologically dependent species,** the same reasoning applies with regard to the need to select species whose conservation is likely to also ensure that less dependent species are not at risk from the fishery depleting the common food supply. Some ecologically dependent species (particularly seabirds and marine mammals) show parental care, so food depletion may be detected with reproductive failure rather than waiting for population-scale response to be quantified.

Characteristics for good species for which to set objectives and reference points include:

o diet is reasonably well known, including information on inter-annual variability;

o evidence is available that the species of prey being harvested by the fishery is well represented in the diet;

o evidence is available that prey-switching from the species being harvested is rare, or at least does not result in complete compensation when the prey has become rare;

o evidence is available that the foraging range of the species of interest does not extend well beyond the region of operation of fishery on a time scale relevant to the rate of renewal of the prey;

o there is a population parameter (such as breeding success, growth rate) related to feeding whose trend can be quantified. The population parameter is best if it is not strongly influenced by non-feeding conditions.

Because many population parameters are influenced by diverse environmental factors, the reference point suitable to trigger management action may have to be a sustained change in the population parameter, corresponding to activity of the fishery over a comparable period.

As an example, the ICES Study Group on Effects of Sandeel Fishing (ICES, 1999b) presents a rationale for using a three-year depression in breeding success of kittiwakes (*Rissa tridactyla*) as a reference point corresponding to local depletion of sandeels.

2.6.1.4   Scavengers

For **scavengers** one is considering management action to address a higher order relationship, in that the increase in scavenging species is only a concern because they may reach abundances where they are detrimental to other populations. Correspondingly, an appropriate objective must be for a species whose populations are likely to be negatively impacted by abundant scavengers. Moreover, one must be confident that the scavengers presenting the threat to the species of concern are those whose populations are benefiting from fish remnants produced by the fishery. If both the scavengers and the populations that the scavengers are affecting are birds, it might be appropriate to use declining access to breeding sites, increased disturbance of breeding activities (from courtship to fledlging success), or direct mortality as indicators of impact.

Characteristics of good species for which to set reference points for impacts of scavengers have many similarities with criteria for choosing ecologically dependent species (Section 2.6.1.3), and include:

- o   The link between the scavenger population and the population of concern is tight, and well documented;
- o   The feature(s) of the population of concern which are being monitored can be quantified well;
- o   The trend in the feature being monitored can be shown to be causally linked to the impact of scavengers, and is not often likely to experience large perturbations due to the other factors;
- o   The increase in the scavenger population can be shown to be causally linked to the provision of fish remnants.

2.6.2   Habitats

Protection of habitats is a prerequisite for protecting the species dependent upon the habitats. Given the recognised loss of habitat features in some areas, development and implementation of ecosystem management objectives ensuring the protection of the remaining areas must be seen as a priority, particularly if habitat features which are vulnerable to disturbance (see Section 2.3) are uncommon.

The most straightforward approach to habitat protection is the complete exclusion of damaging activities from all habitats at risk. It may, however, be that a certain level of habitat degradation may be acceptable, for example because the effects are reversible. Ultimately management for habitat considerations may extend to all habitats, but at least initially such considerations are likely to be restricted to a sub-set of habitats. We set out below factors, which may influence the choice of such a sub-set for setting habitat objectives, and the reference points that might be appropriate.

2.6.2.1   Criteria for selection

Criteria that might be used to select habitats for conservation include:

- o   High degree of 'endemic' biota, for example, sea lochs and coastal lagoons;
- o   Restricted distribution, inherently rare habitats such as *Lophelia* reefs;
- o   High biological diversity. The Jakarta Mandate requires protection of habitats with high biological diversity—candidate areas might include sub-littoral reefs and boulder beds;
- o   EC Habitats and Species Directive Annex 1 list—the EC Directive provides a list of habitats within Europe which it believes should be protected;
- o   Identified in Biodiversity Action Plans—in the UK this includes *Sabellaria* reefs, *Modiolus* beds, *Lophelia* reefs, deep mud;
- o   Essential fish habitat—such as gravel banks for herring spawning.

2.6.2.2   Possible objectives and reference points

The stage of development of objectives and reference points for populations is well in advance of that for habitats. Current knowledge therefore does not allow a full discussion (cf. Section 2.5), rather we point to features which warrant further investigation, and for which general objectives could be set. In each case, given the general objective, then corresponding reference points would have to be selected as well:

- o   Specified proportion of initial area maintained in un-impacted condition;
- o   Some property of the spatial distribution—e.g., minimum of n% in un-impacted condition in any ICES rectangle. This, at least partially, addresses the issue of patchiness;
- o   Some measure of habitat quality (e.g., epibiota: biomass per unit area) across the whole habitat unit. Reasonable reference points would allow some use of, and hence effect on, an area. This requires knowledge of the form of the relationship between the degree of change in range and the risk that the change is irreversible;
- o   As current management of target species is done within a precautionary framework, including multiple objectives and reference points (biomass and fishing mortality), management of habitats may also require

combinations of criteria. Such multi-criterion objective might be 'no more than x% change in a metric in the entire habitat unit and a minimum of y% in un-impacted condition in an ICES rectangle'.

2.6.3 Genetic properties

A number of management measures are available to conserve genetic diversity of exploited species (ICES, 1999a) and some of these could require objectives and reference points specific to genetic properties of the stock or species. Genetic diversity is directly related to N_e, the effective number of spawning individuals in a population, and the most appropriate variable for assessing population viability (Barton and Whitlock, 1997). Complex social systems, skewed sex ratios, and other complicating factors of breeding systems may result in N_e being smaller than the number of mature individuals in a population (Burgman *et al*., 1993). Maintaining large N_e increases the likelihood that favourable mutations will become widespread and deleterious ones will be unduly expressed.

Population size is the single most important factor in sustaining a high level of genetic variation within a population of a species, and for essentially all fished species maintaining a population above B_{pa} has a high likelihood of ensuring that the number of potential breeding individuals also exceeds N_e.

Given a mean population size, N_e is negatively influenced by extreme fluctuation in population size, variation in the number of offspring per family and unbalanced sex ratios. An objective to keep a population above B_{pa} will prevent fluctuations serious enough to result in unacceptable risk to N_e. Variation in offspring per family is not amenable to measurement or control in the wild, so objectives addressing that factor usually are not appropriate. (This could be a concern for harvesting of moderately sedentary intertidal species, such as abalone (*Haliotis*) and sea urchins (*Strongylocentrotus*), where 'mating' opportunities are restricted by the linear nature of the habitat. However, setting B_{pa} can accommodate the need for a reasonable density as well as abundance of mature individuals.

The sex ratio of a population is rarely considered as a management objective, although if the sex ratio of breeders departs from 1:1, N_e and genetic variation will be reduced. An effective population of 50 males and 50 females is nearly 2.8 times larger, genetically, than one of 10 males and 90 females. Some jurisdictions manage species such as snow crabs (*Chionocetes*) and shrimp (*Pandalus*) with size limits which allow only males to be harvested. In such cases target exploitation rates are set to ensure that enough males survive to mate with all females. Under such approaches, it is unlikely that the skew in sex ratio will be so bad that N_e reaches values which reflect significant risk to the population.

For species where there is a high degree of population sub-division, that is high among-population genetic variation, reference points may be needed for the individual populations. Tools for population risk assessment, such as population viability analysis, may be appropriate for developing objectives for subpopulations (Burgman *et al*., 1993; Beissinger and Westphal, 1998; Dunham *et al*., 1999). The reference points themselves, however, are still likely to be numbers or biomasses, and function like B_{pa}. When the extinction risk of many local populations must be considered, the same problems of practicality are encountered as with reference points for all possible species of by-catch. Suggestions in Section 2.4 are relevant here. In addition, Allendorf *et al*. (1997) have provided a set of qualitative criteria for ranking conservation value of salmonid stocks, and these warrant review for wider application.

Objectives and reference points for selection differentials may be important, but further work within that field is required before it will be possible to identify reference points which can be applied within existing precautionary frameworks. More must be known about the relationship between selection differential and conservation risk, and how to measure selection differentials in operational settings, before even objectives can be proposed for this important property.

2.6.4 Emergent properties

While not ruling out the need to continue to monitor developments in this area, WGECO finds no evidence that such ecosystem properties need, or even can be, subject to direct management objectives. However, WGECO acknowledges that even if objectives and reference points for emergent properties are not warranted by present knowledge, many metrics of ecosystem properties, such as measures of diversity, can serve a valuable role in communicating with many clients of marine science, for example as part of the approach proposed in Lanters (1999) and illustrated in Figure 2.6.4.1.
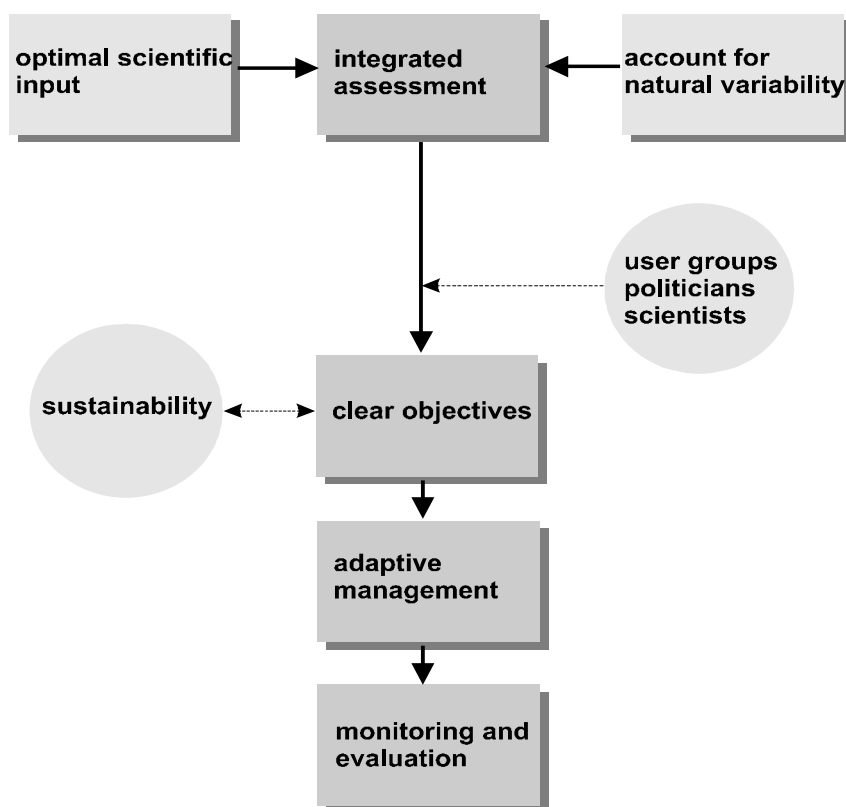
Figure 2.6.4.1   A framework for ecosystem management. The central line shows how to make it operational. Other elements are preconditions for ecosystem management (from Lanters, 1999).

2.7   Conclusions and Way Forward

A number of international agreements require protection of the marine ecosystem. WGECO believes that in some areas there is now urgent need for key habitats to be afforded protection. A difficulty at present is that our knowledge about benthic habitats is limited. We have some knowledge about soft bottom habitats and communities, but the diversity of habitats associated with hard bottoms and their special topographical features, including habitat-forming species such as deep-sea corals and *Sabellaria* reefs, are not well known. There is therefore a need for classifying and mapping the distribution of benthic habitats in the North Sea, and WGECO supports strongly ICES work in this area.

Development and implementation of population objectives and reference points for non-target species is hampered by our lack of knowledge of the biology and ecology of many species and the often rather subjective allocation of taxa to groupings such as 'sensitive to fishing'. There is a need to increase our knowledge of the ecology of the benthos and the development of robust and objective criteria, and scales/metrics, for the independent assessment of vulnerability/fragility of habitats and species.

It is generally accepted that discards have a negative effect on the ecosystem. They provide no economic return and the extra time spent sorting the catch places an economic burden on the industry. Minimising unwanted catch must therefore remain an important management objective. This must be achieved by better selectivity of the gear and the release back into the water of the unwanted catch alive and in good condition.

At this time WGECO believes that we are not in a position to recommend that objectives and reference points for emergent properties of ecosystems are necessary, beyond the ones which would assure sustainability and conservation of all species and habitats impacted by fishing. Neither are we prepared to confirm that single-species, habitat and genetic objectives and reference points alone are enough to ensure a precautionary approach to ecosystem management, only that no properties have been shown to be placed at risk if the constituent components are conserved.

Failure to address socio-economic issues limits our ability to make progress with implementing existing biologically based management. Further development of integrated management objectives as a basis for an ecosystem approach to management requires development of socio-economic models that allow integration of ecological and social issues.

WGECO feels that the way ahead involves:

- o   rapid implementation of habitat objectives and reference points for key habitats;
- o   rapid movement to fishing efforts that provide a high likelihood of achieving target species reference points;
- o   further development of genetic objectives and objectives for non-target species;
- o   reduction in unwanted catch without increasing the quantity of damaged material left on the sea floor;
- o   development of mechanisms linking ecosystem management tools to appropriate objectives and reference points;
- o   development of strategies and tools for addressing the social costs of reduction in harvest required to meet ecosystem (and single-species) objectives.

2.8   References

Allendorf, F.W., Bayles, D., Bottom, D.L., Currens, K.P., Frissell, C.A., Hankin, D., Lichatowich, J.A., Nehlsen, W., Trotter, P.C., and Williams, T.H. 1997. Prioritizing Pacific salmon stocks for conservation. Conservation Biology, 11: 140–152.

Anon, 1995. The Ecosystem Approach: Healthy Ecosystems and Sustainable Economies. Volume I. Overview. Report of the Interagency Ecosystem Management Task Force. Virginia, United States.

Baillie, J., and Groomsbridge B., 1996. IUCN Redlist of Threatened Animals. IUCN Publ. Serv. Gland, Switzerland. 368 pp.

Barton, N.H., and Whitlock, M.C. 1997. The evolution of metapopulations. *In* Metapopulation biology: ecology, genetics, and evolution. Ed. by I.A. Hanski and M.E. Gilmin. Academic Press, San Diego, California.

Beissinger, S.R., and Westphal, M.I. 1998. On the use of demographic models of population viability in endangered species management. Journal of Wildlife Management, 62: 821–841.

Burgman, M. A., Ferson, A.S., and Akçakaya, R. 1993. Risk assessment in conservation biology. Chapman and Hall, New York, NY, USA.

Christensen, N.L., Bartuska, A.M., Brown, J.H., Carpenter, S., D'Antonio, C., Francis, R., MacMahon, J.F., Noss, R.F., Parsons, D.J., Peterson, C.H., Turner, M.G., and Woomansee, R.G. 1996. The Report of the Ecological Society of America Committee on the Scientific Basis for Ecosystem Management. Ecological Applications, 6: 665–691.

CITES. 1994. Convention on International Trade in Endangered Species of Wild Fauna and Flora, Ninth Meeting of the Conference of the Parties, Fort Lauderdale (United States of America), 7 to 18 November 1994. Conference of the Parties Resolution Conf. 9.24.

Daan, N., Richardson, K., and Pope, J.G. 1996. Changes to the North Sea Ecosystem and Their Causes: Århus 1975 Revisited. *ICES Journal of Marine Science*, 53: 879–883.

Dayton, P.K., Thrush, S.F., Agardy, M.T., and Hofman, R.J. 1995. Environmental effects of marine fishing. Aquatic Conservation: Marine and Freshwater Ecosystems, 5: 205–232.

Dunham, J., Peacock, M., Tracy, C.R., Nielsen, J., and Vinyard, G. 1999. Assessing extinction risk: integrating genetic information. Conservation Ecology, 3(1): 2.

Fretwell, S.D. 1972. Populations in a seasonal environment. Monographs in Population Biology, 5 Princeton University Press: 217 pp.

Gharrett, A.J., and Smoker, W.W. 1993. Genetic components in life history traits contribute to population structure. *In:* Genetic conservation of salmonid fishers, pp. 197–202. Ed. by J.G. Cloud and G.H. Thorogaard. Plenum Press, London.

Groom and Pascual. 1998. The analysis of population persistence: an outlook on the practice of viability analysis. *In:* Conservation biology for the coming decade. Ed. by P.L. Fiedler and P.M. Kareiva. Chapman and Hall, New York, USA, pp. 4–27.

ICES. 1996. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 1998/Assess/Env:1.

ICES. 1997. Report of the Working Group on the Application of Genetics in Fisheries and Mariculture. ICES CM 1997/F:4.

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

ICES. 1998b. Report of the Study Group on the Precautionary Approach to Fisheries Management. ICES CM 1998/ACFM: 10.

ICES. 1999a. Report of the Working Group on the Application of Genetics in Fisheries and Mariculture. ICES CM 1999/F:1.

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

Lanters, R.L.P. 1999. Basic elements for the implementation of an ecosystem approach to marine management. ICES CM 1999/Z:10. 7 pp.

Law, R., and Grey, D.R. 1989. Evolution of yields from populations with age-specific cropping. Evolutionary Ecology, 3: 343–359.

Law, R., and Rowell, C.A. 1993. Cohort-structured populations, selection responses, and exploitation of the North Sea cod. *In:* The exploitation of living resources, pp. 155–173. Ed. by T.K. Stokes, J.M. McGlade, and R. Law. Lecture Notes in Biomathematics 99. Springer-Verlag, Berlin, Germany. 264 pp.

MacCall, A.D. 1990. Dynamic geography of marine fish populations. University of Washington Press, Seattle, WA. 153 pp.

Mileikovsky, S.A. 1971. Types of larval development in marine bottom invertebrates, their distribution and ecological significance: a re-evaluation. Marine Biology, 10: 193–213.

Millner, R.S., and Whiting, C.L. 1996. Long-term changes in growth and population abundance of sole in the North Sea from 1940 to the present. *ICES Journal of Marine Science*, 53: 1185–1195.

NRC (USA). 1999. Sustaining marine fisheries. National Research Council, National Academy of Sciences, Washington, D.C. 164 pp.

Rijnsdorp, A.D. 1993. Fisheries as a large-scale experiment on life-history evolution: disentangling phenotypic and genetic effects in changes in maturation and reproduction of North Sea plaice, *Pleuronectes platessa* L. Oecologia, 96: 391–401.

Rowell, C.A. 1993. The effects of fishing on the timing of maturity in North Sea cod (*Gadus morhua* L.). *In:* The exploitation of living resources, pp. 44–61 Ed. by T.K. Stokes, J.M. McGlade, and R. Law. Lecture Notes in Biomathematics 99. Springer-Verlag, Berlin, Germany. 264 pp.

Ruzzante, D.E., Taggart, C.T., Cook, D., and Goddard, S.V. 1997 Genetic differentiation between inshore and offshore Atlantic cod (*Gadus morhua*) off Newfoundland: a test, and evidence of temporal stability. Canadian Journal of Fisheries and Aquatic Sciences, 54: 2700–2708.

Sala, E., Boudouresque, C.F, and Harmelin-Vivien, M. 1998. Fishing, trophic cascades, and the structure of algal assemblages: evaluation of an old but untested paradigm. Oikos, 82: 425–439.

Shaw, P.W., Pierce, G.J., and Boyle, P.R. 1999. Subtle population structuring within a highly vagile marine invertebrate, the veined squid *Loligo forbesi*, demonstrated with microsatellite DNA markers. Molecular Ecology, 8: 407–417.

STECF. 1999. Report on the STECF Workshop on Research Priorities on Environmental Impact of Fishing. Fuengirola, Spain, 18–20 October 1999.

Trippel, E.A. 1998. Egg size and viability and seasonal offspring production of young Atlantic cod. Transactions of the American Fisheries Society, 127: 339–359.

Trippel, E.A., Morgan, M.J., Frechet, A., Rollet, C., Sinclair, A., Annand, C., Beanlands, D., and Brown, L. 1997. Changes in age and length at sexual maturity of northwest Atlantic cod, haddock and pollock stocks, 1972–1995. Canadian Technical Report on Fisheries and Aquatic Sciences, 2157. 120 pp.

Tuljapurkar. 1990. Population dynamics in variable environments, lecture notes in biomathematics. Springer Verlag, Germany.

van Oppen, M.J.H., Klerk, H., Olsen, J.L., and Stam, W.T. 1996. Hidden diversity in marine algae: some examples of genetic variation below the species level. Journal of the Marine Biological Association of the United Kingdom, 76: 2.

3 COMMUNITY-SCALE ECOQOS

3.1 An Introduction to Ecological Quality Objectives

This section builds from the general themes of ecosystem objectives and reference points, to the specific context of ecosystem management initiatives in the Northeast Atlantic. The inputs to the documents prepared in advance of the Bergen Convention and Bergen Declaration are prominent in this work.

3.1.1 History of EcoQOs

OSPAR and the North Sea Task Force (NSTF) have a relatively long history in the development of Ecological Quality Objectives (EcoQOs), as an approach to implementing in the Northeast Atlantic the provisions of Annex V (on the protection and conservation of the ecosystems and biological diversity of the maritime area) of the 1992 Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR Convention) (see Annex 2). Skjoldal (1999) gives a comprehensive overview of their evolution. Interestingly, the first call for a definition of terms of EcoQOs was in a draft of the European Commission Ecological Quality of Water Directive (Skjoldal, 1999). This is the ancestor of the EU Water Framework Directive, of growing importance for the management of coastal waters. However, the major starting point of EcoQOs has been the mutual demand of OSPAR and the North Sea Conferences for some method that allows assessment of the ecological status of the marine environment and defines objectives for the preferred ecological status. The basis for the concept was developed, beginning in 1992, during a sequence of three international workshops. Ecological Qualities (EcoQs) and the objectives derived from them have since been a permanent item on the OSPAR agenda, receiving regular attention during workshops and meetings. The result of all these efforts is that the scientific and political community connected to OSPAR began to develop and adapt a conceptual framework for EcoQs and EcoQOs.

In some countries, additional scientific effort has been directed towards their further development. In 1997, the basis was laid for the further advancement of the concept of EcoQOs through the Intermediate Ministerial Meeting on the Integration of Fisheries and Environmental Issues in the North Sea (IMM). During this meeting, both the Environmental and Fisheries Ministers composed a list of conclusions and recommendation on the integration issue. They are brought together in the Statement of Conclusions (IMM, 1997). Conclusion 2.6[1] calls for the development and implementation of an ecosystem approach in the management of marine ecosystems. As a follow up, a workshop on the ecosystem approach was held in 1998 in Oslo, Norway. This workshop concluded, amongst others, that clear objectives are needed as part of the development of an ecosystem approach. The workshop further suggested that Ecological Quality Objectives under development within OSPAR could provide a solid basis for defining clear objectives (Anon., 1998). As a result a workshop specifically on Ecological Quality Objectives was organised in 1999 in Scheveningen, The Netherlands. Both workshops were attended by a mixture of policymakers, stakeholders, and scientists.

The basic ecosystem properties included in the OSPAR conceptual framework for EcoQs (Skjoldal, 1999) are:

o Diversity;
o Stability;
o Resilience;
o Productivity;
o Trophic Structure.

Because EcoQs have to address ecosystem properties in relation to human influences, the OSPAR JAMP (Joint Assessment and Monitoring Programme) issues were taken as a basis for covering the latter. Together, these issues make up the conceptual framework shown in Figure 3.1.1. Habitat issues were a late addition.

---

[1] The official text of Statement of Conclusion 2.6 reads:

further integration of fisheries and environmental protection, conservation and management measures, drawing upon the development and application of an ecosystem approach which, as far as the best available scientific understanding and information permit, is based on in particular:

- the identification of processes in, and influences on, the ecosystems which are critical for maintaining their characteristic structure and functioning, productivity and biological diversity;
- taking into account the interaction among the different components in the food-webs of the ecosystems (multi-species approach) and other important ecosystem interactions; and
- providing for a chemical, physical and biological environment in these ecosystems consistent with a high level of protection of those critical ecosystem processes.
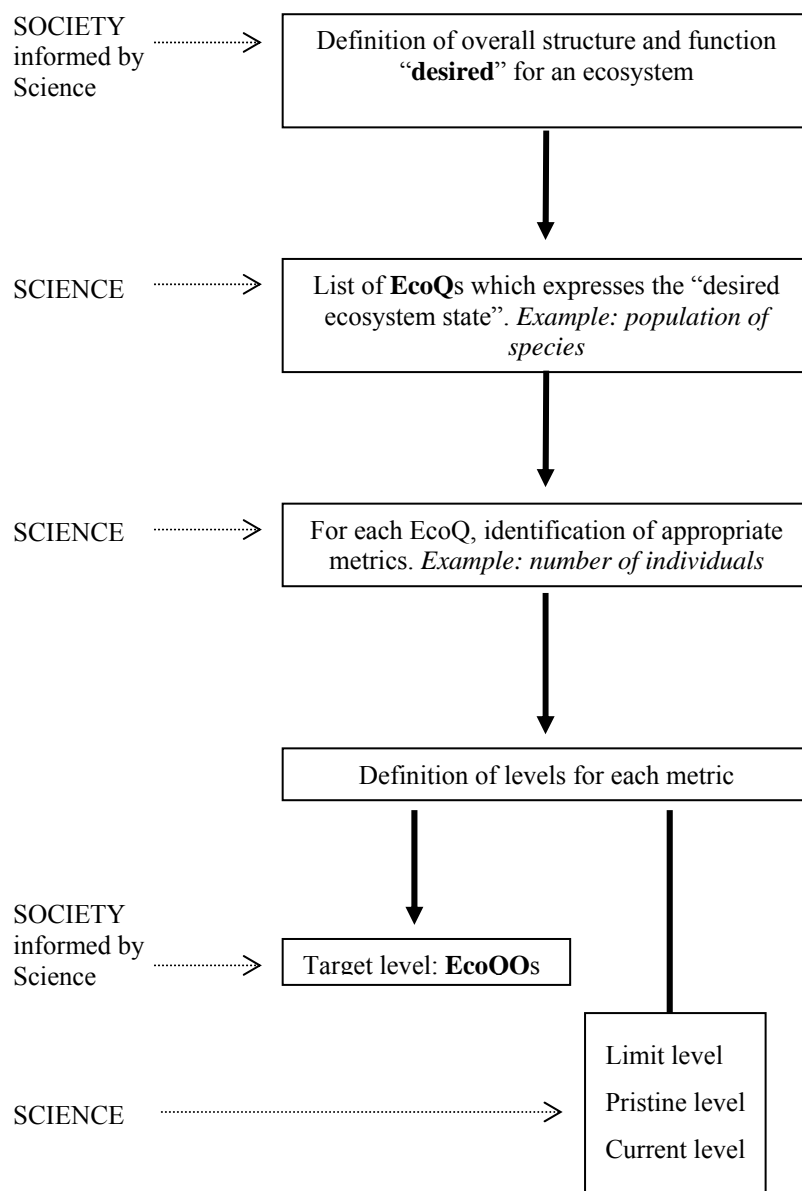
Figure 3.1.1   Conceptual framework for the methodology of describing EcoQ and setting EcoQOs (from Lanters *et al.*, 1999).

Based on a document especially prepared for the meeting (Lanters *et al.*, 1999), the stakeholders, policymakers, and scientists present at the Scheveningen workshop concluded that EcoQOs should be developed for ten issues (Anon., 1999). These ten issues cover EcoQOs at the species, community, and ecosystem levels. They also more or less cover the range from structural (diversity) to functional (processes) aspects of the ecosystem. OSPAR agreed that this list of ten issues would form the basis for future work (OSPAR, 2000a), but would also keep an open eye for further improvement or extension of the proposed list of issues.

Table 3.1.1   The proposed set of ten issues for EcoQOs for the North Sea derived from the Scheveningen workshop (Anon., 1999).

Proposed set of issues	
1	Reference points for commercial fish species
2	Threatened or declining species
3	Sea mammals
4	Seabirds
5	Fish communities
6	Benthic communities
7	Plankton communities
8	Habitats
9	Nutrient budgets and production
10	Oxygen consumption

The set of ten issues were further explored under the guidance of OSPAR. Norway, the Netherlands, ICES and the OSPAR Eutrophication Task Group (ETG) were assigned the ten issues according to the following plan:

1)	Commercial fish species	Norway
2)	Threatened and declining species	Netherlands
3)	Sea mammals	ICES
4)	Birds	ICES
5)	Fish communities	Netherlands
6)	Benthic communities	Netherlands/ETG
7)	Plankton communities	Norway/ETG
8)	Habitats	Norway
9)	Nutrient budgets and production	ETG
10)	Oxygen	ETG

The main objective of the OSPAR Biodiversity Committee was to determine the feasibility of putting some very clear examples of EcoQOs on the agenda of the Fifth North Sea Conference in March 2002. In this process, ICES was responsible only for the elaboration of EcoQOs for marine mammals and seabird species. In the other areas WGECO undertook to provide OSPAR with an independent evaluation of the scientific credibility of the framework and methods being applied, and applied its experience of dealing with ecological reference points to these newer fields of marine science in management contexts (See Sections 1 and 2 of this Chapter).

3.1.2   Terminological issues

Both OSPAR and ICES have been trying to place scientific advice and management decision-making with regard to marine environments and resources into a more rigorous and explicit framework. These efforts, and those of many other groups worldwide, have evolved from the meetings and agreements following from the 1992 UN Conference on Environment and Development (UNCED) in Rio, so it should not be surprising that many terms and phrases are used by both OSPAR and ICES (and other marine conservation and management organizations). Unfortunately, the terms have been evolving partially independently (even within different parts of ICES), so similar words and phrases often mean different things when used by different bodies. This creates potential for confusion and misunderstandings. The involvement of ICES with OSPAR's initiative to develop EcoQOs for the North Sea makes it particularly important that terms be used in consistent and clear manners (ICES, 2000c, 2001d; OSPAR, 2000a). Although there has been a small evolution in the definition of EcoQs and EcoQOs, the main features of their definitions have hardly altered since 1992. Because EcoQOs are currently being developed under the flag of OSPAR, the definitions that came as a result of the Scheveningen Workshop (Anon., 1999) will be used. The following definitions apply throughout this report (ICES usages are those used throughout all ICES advice on fisheries, as summarised in Section 1 of ICES, 2001d):

> **Ecological Quality (EcoQ):** An overall expression of the structure and function of the marine ecosystem taking into account the biological community and natural physiographic, geographic and climatic factors as well as physical and chemical conditions, including those resulting from human activities.

Ecological Quality Objectives (EcoQO): The desired level of ecological quality relative to a reference level.

Reference points: In **ICES** advice regarding fisheries, **reference points** are specific values of measurable properties of systems (biological, social, or economic) used as benchmarks for management and scientific advice. They function in management systems as guides to decisions or actions that will either maintain the probability of violating a reference point below a pre-identified risk tolerance, or keep the probability of achieving a reference point above a pre-identified risk tolerance (ICES, 2001d). There will be multiple reference points for any single property of a system, each serving a specific purpose. In advice on non-fisheries issues, ICES terminology has been somewhat more variable, with **reference value** sometimes used in contexts identical to those where reference point is used in advice on fisheries.

Reference level: In **OSPAR** usage, **reference level** began as the level of EcoQ where the anthropogenic influence on the ecological system is minimal. It became clear that it could be very difficult or impossible to determine such reference levels, when systematic monitoring of properties related to the EcoQ began well after pristine conditions were perturbed. This not only applies to biological conditions, but also to naturally occurring chemical substances. Therefore, OSPAR acknowledged that a pragmatic approach may be required to establish and use reference levels. OSPAR noted that temporal trends could be informative about past conditions, and in some circumstances preliminary reference levels could be taken as the starting point of a time-series. For this reason, the wording "**a** reference level" was preferred over the use of "**the** reference level" in the EcoQO definition (Anon., 1999). It should be emphasised that "reference level" should not be confused with the objective. Although the original meaning of "reference level" as defined in the context of EcoQOs had a different meaning than "reference points" used in the context of fisheries (OSPAR, 2000a), the modified usage by OSPAR leads to the meaning of **reference level** being specific to each application. OSPAR and ICES seem to still differ somewhat because, at least for the present, within OSPAR there appears to be only a single reference level per EcoQO at any time. It appears that the criteria on which the reference level is set can change from EQ to EQ, or over time, leading to changes in the reference level as well, so in that sense reference level does function much like the concept of reference points in ICES advice.

Target Reference Points: In **ICES** usage, particularly for fisheries, target reference points are properties of stocks/ species/ecosystems which are considered to be "desirable" from the combined perspective of biological, social, and economic considerations. Where they address biological aspects of ecosystems, target reference points must in all cases be at least as "safe" as precautionary reference points selected on exclusively biological considerations. Beyond that conservation-based constraint, ICES has stressed that managers, decision-makers, and stakeholders have the responsibility for selecting target reference points (see Section 3.1.3.2). When ICES provides advice relative to **target reference points**, unless otherwise requested ICES assumes that management should be designed to achieve them on average, and hence advice is risk neutral with regard to them, as long as conservation reference points are not placed at unacceptable risk.

Target levels: In **OSPAR** usage, target levels identify states of the EcoQO (or, operationally, values of the metrics of the EcoQO) that management should be trying to maintain with high probability. In this usage, they function in a manner very similar to Target Reference Points as used by ICES. However, the request from OSPAR to ICES, as a scientific advisory body, to provide advice on suitable target levels suggests that target levels are identified through scientific endeavours. This is quite different from the ICES perspective on target reference points (see Section 3.1.3.2), and the difference has not yet been resolved.

Limit Reference Point: In **ICES** usage, a value of a property of a resource that, if violated, is taken as *prima facie* evidence of a conservation concern. By "conservation concern", ICES means that there is unacceptable risk of serious or irreversible harm to the resource. Outside the limit reference point, the stock has entered a state where there is evidence that: productivity is seriously compromised, <u>or</u> exploitation is not sustainable <u>or</u> stock dynamics are unknown.

Management should maintain stocks inside limit reference points with high probability. To account for uncertainty in assessments, ICES uses **precautionary reference points** as a basis for scientific advice, with the intent that management consistent with precautionary reference points should have at least a 95% probability of keeping a property away from its limit reference point. Limit reference points are based on the biology of the stock/species/ecosystem, independent of social and economic considerations. Hence ICES has argued that they should be identified by technical experts, and ICES has selected limit reference points for stocks on which it provides scientific advice. **OSPAR** does not appear to have chosen to include the notion of limit reference points within the EcoQ and EcoQO framework that it is developing.

The request of OSPAR to ICES to develop EcoQOs requires that the sometimes subtle differences in philosophies behind these concepts and terms be understood clearly. The review of terminology above does not find points of specific inconsistency between OSPAR and ICES. However, there are a few more general terms used in very specific and consistent ways in ICES fisheries advice, but in the larger community of those interested in marine ecosystems and

conservation the terms have a variety of meanings. In this report the terms will always be used with the ICES meanings, unless specifically stated otherwise. For that reason it may be helpful to explain those usages here:

Conservation is used in the sense of conserving natural resources. The resources can be used as long as the usage is at rates and in ways that do not place the resource, or the ecosystem in which it is found, at risk of harm that is serious or difficult to reverse in the short, medium or long term. Resources may be being conserved when they are in conditions quite different from their pristine states.

Sustainability is used to refer to the use(s) made of the resource, and not to the state of the resource. A strategy for use of a resource is sustainable when it could be pursued in the long term without causing unacceptable risk of a conservation problem for the resource being used, or the ecosystem in which it is found. Quite often a fishery, for example, is said to be sustainable, when, to be precise, what is meant is that the strategies used to manage and prosecute the fisheries are sustainable. By applying "sustainable" strictly to the use, and not to the resource itself, this is a slightly more restrictive use of the term "sustainable" than is encountered in some general reports on conservation of biodiversity, but is in no way inconsistent with those uses.

For example, the Convention on Biological Diversity (CBD) defines the term Sustainable Use to mean "the use of components of biological diversity in a way and at a rate that does not lead to the long-term decline of biological diversity, thereby maintaining its potential to meet the needs and aspirations of present and future generations." As with the ICES usage, the CBD definition includes the notions of using the resource, but in ways that can be continued in the long term without causing conservation problems.

The final terminological issue relative to this report is our use of **metric** to refer to the biological attribute that is being considered as an indicator of an ecological quality of the system. In our discussions, we routinely used "indicator" and "metric" interchangeably. However, in the written report, WGECO took note that "indicator" sometimes carries a specific meaning as an "indicator species". Therefore, we decided to use **metric** in all cases where we mean something that can be measured quantitatively (or, when appropriate, qualitatively) and is at least be considered as being a suitable way to measure the ecological property that the EcoQ is intended to capture. Where we use **indicator,** we mean for it to be interpreted in the sense of "indicator species".

3.1.3   Conceptual issues

3.1.3.1   Interaction between EcoQ and EcoQO

The requirement for the development of EcoQOs arises from the need to bring forward an "ecosystem approach" to environmental management. This is a key part of the adoption of the Convention on Biological Diversity (CBD) signed at the UN Rio Conference and adopted as a basis for management by the EU and the Intermediate Ministerial Meeting of North Sea ministers.

Unfortunately, the term "ecosystem approach" has been used in a wide variety of contexts and been imparted with a range of definitions, as have the terms EcoQ and EcoQO (Section 3.1.2). From the OSPAR definitions, a sequential framework for developing EcoQs and EcoQOs can be seen (Figure 3.1.1). The starting point for the development of ecosystem approaches to environmental management is to define the "overall structure and function" desired for the ecosystem being considered. The specification of this "desired ecosystem" is a societal decision, although science has some key roles (see Section 3.1.3.2). This desired overall state of the ecosystem must be expressed as a series of clear statements that will constitute the list of EcoQs. Next, it is necessary to identify at least one metric for each EcoQ. The question of the necessary and sufficient number of metrics to ensure conservation of the system, or even achieve the EcoQs specified by society, is not simple (Section 3.1.4). From this list of metrics, one must derive desired levels for various measures of the system, which correspond back to the "desired ecosystem" initially specified by society. The desired values of the metrics comprise the suite of EcoQOs. Consistent with the changing OSPAR definition of "reference level", there is no inherent need for EcoQOs to be set always to the condition where anthropogenic influences are minimal. In fact, this would imply no use of environmental services such as waste treatment or food production. Rather, the "appropriate" values for the EcoQOs are determined by the overall desired ecosystem. The appropriate measures and quantitative values for the EcoQs and EcoQOs will vary among systems and depend on the priority given to various issues. Moreover, it is implicit that the setting of EcoQOs should be done in an integrated manner, to ensure that they are mutually achievable and collectively sufficient to ensure conservation of the ecosystem. However, for pragmatic reasons the initial approach used at the Scheveningen workshop and continued by OSPAR in its request for advice is to develop EcoQOs for various ecosystem components in a variety of different groups (Section 3.1.1). The implications of a number of these issues will be discussed in Section 3.3.

3.1.3.2   Role of science

The different approaches to reference points, reference levels, limits, and targets increases the potential for confusion about suitable roles for technical experts, policymakers, and advocates of many sectors including users and non-users. Although it is inappropriate for ICES to advise on preferred governance approaches among policymakers and public sectors, it is important that the role of science be understood in the larger process of selecting and implementing EcoQs and EcoQOs. Note that the term "technical expert" is used here to make clear that "scientists" includes not just biological, physical, and chemical scientists and collaborating quantitative experts. Social sciences also have an important contribution to make to the role of science.

The selection of properties of ecosystems that are essential to their conservation is the responsibility of technical experts, as is the selection of metrics of those properties. If clients wish to have relative priorities assigned to the general properties or their specific metrics, technical experts also have a key, but not exclusive, role. Technical experts are the appropriate group to assign priorities based on the degree to which conservation of the ecosystem depends on each of various properties of the system, as well as to assign priorities among metrics based on their reliability and sensitivity. Rankings of properties and metrics based on human values is not an issue appropriate for biological and physical scientists, although social scientists may work with policymakers and the public to clarify public opinion on such rankings.

Once a suite of properties needed for conservation of the ecosystem is identified, and metrics of the properties have been selected, several groups have roles in setting various benchmarks along the metrics, and identifying acceptable and unacceptable domains of the properties. It is the responsibility of the technical experts to specify lower (or upper) conservation limits for metrics and properties; that is, values of a metric or states of a property below (or above) which there is increasing risk of harm that is serious or difficult to reverse. (Some properties and their metrics may have both upper and lower limits associated with conservation.) There will almost always be uncertainties with regard to determination of both conservation limits of properties and metrics, and current states of properties and metrics. Technical experts are also responsible for quantifying such uncertainties to the fullest extent possible, and selecting precautionary positions on the properties and metrics such that if management is risk neutral relative to the precautionary reference points, there will be a high probability that the conservation limits will be avoided. How high that probability should be is a societal choice, based on its risk tolerances.

For many plausible candidate metrics, there is insufficient contrast in the historical data (if the data exist at all) to be informative about where the conservation limit may be. In such instances, technical experts have special challenges to determining how to advise on managing risk.

If policymakers or the public wish to know the state of a property prior to substantial anthropogenic perturbations, it is also a question that should be answered by technical experts. That does not mean that the question always is answerable, or that the answer, if possible to provide, is a sound basis for management. The same points apply to questions about the maximum value (or minimum) that a property or metric could assume, if management were intended to achieve the most extreme state possible for that ecological attribute of a system.

Between the states that are determined by conservation limits to be avoided with high probability and the most unaltered or extreme value possible to achieve, policymakers and society have to choose the desired state that management should aim for. Such targets are chosen on the basis of society's values, often as interpreted by policymakers. Technical experts may participate in this exercise as citizens, advocating whatever point of view they may have. However, they have the responsibility to acknowledge that they are merely advocating their particular special interest (even if they believe it is an especially enlightened one), and have no special privileges at the table where competing interests are seeking consensus. It can be difficult to keep these identities distinct, because the technical experts have a role during the negotiations leading to setting management targets: that of warning when targets under consideration would place the conservation limits at unacceptable risk of being violated. Such advice has to be perceived as objective and impartial, which can be hard when the same individuals have been involved in debates over proper values to be the basis for society's choices. Assuming that consensus can be achieved on a set of management objectives that are mutually compatible, the technical experts have a final role to lead the translation of society's values, often expressed qualitatively, into operational management targets, expressed in the currencies of the metrics. This may make it appear that the technical experts are setting the targets, or the EcoQOs, but their role is only as translator of society's choices onto the biological axes that are being used.

3.1.3.3   Approaches to setting EcoQOs

3.1.3.3.1   Approaches used by other Working Groups or experts

WGECO began with draft text on EcoQOs from the Working Group on Seabird Ecology (WGSE) and the Working Group on Marine Mammal Population Dynamics and Habitats (WGMMPH), and OSPAR consultants' reports on

EcoQOs for benthos and threatened and declining species. WGECO began by examining the approaches taken for these four ecosystem components and considered them with a view to developing a generic context for determining EcoQOs. In the cases we examined (all biological systems), it was recognised that it was impossible to know what the pristine state of a system which has minimal anthropogenic influence should be. For contaminants, it is relatively easy to see what the reference level (*sensu* OSPAR prior to 2000) should be, i.e., zero for synthetic substances such as DDT, PCBs, and the appropriate biogeochemically determined level for naturally occurring substances. This is not the case for biological populations or communities.

The benthic reference level proposed (de Boer *et al.*, 2001) is that it should "represent the situation under minimal human impact". The report then advocates the use of values derived from the 1986 data series as a basis for EcoQOs (although it is noted that these should be regarded as minimum/maximum values for the proposed metrics), thereby implying that the situation in 1986 is the acceptable ecological quality. The WGSE and WGMMPH were concerned with EcoQOs for these species groups and the EcoQOs proposed reflect this emphasis. The WGSE considered two possible approaches, the possibility of defining metrics for each species which give a measure of ecosystem health, i.e., using each species as an ecosystem metric, or the development of metrics of possible impacts which use appropriate aspects of seabird ecology. WGSE proposed the latter as being a more sensible approach and so developed EcoQOs relevant to eight ecosystem anthropogenic effects that use seabirds as metrics.

WGMMPH generally concurred with the approach of WGSE, expressing concern, however, that the WGSE approach did not give sufficient prominence to population size, which they considered to be the trait of most relevance to the public. They developed a hierarchical figure, illustrating a series of steps from population size, through life history factors such as productivity and mortality, to a list of human effects from the OSPAR JAMP, and discussed the relationships that could possibly exist among the effects, the life history factors, and ultimately population size. They also discussed the concepts of target and reference levels on EcoQ metrics. In the documentation available at the end of the formal meeting of WGMMPH, specific EcoQs and their metrics had not been identified, however. Rather it was reported that they would continue to pursue the ideas behind the tabulation. It was expected that most or all of the EcoQs and their metrics would be derived from important life history and biological properties of marine mammal populations, and subsequently linkages would be sought to the human effects. This is somewhat in contrast to the approach of WGSE, who began with the ten issues identified by OSPAR, and then sought properties of seabirds considered particularly sensitive to each.

For the "threatened and declining species" the objective is more clear—the rebuilding of populations—although the level to which they should be rebuilt, i.e., 50% of the reference level, requires that the target EcoQO is determined within a societal framework. The key issue here was what criteria triggered inclusion of a species as "threatened and declining"; an issue that although in concept is exclusively scientific, in practice is hotly debated among even scientific experts (see Section 5.4).

3.1.3.3.2 Major influences on WGECO's approach

In Section 3.3, the approach WGECO followed in selecting possible EcoQs and their metrics is explained in detail, and its application is illustrated in Section 5.5. As much as possible, WGECO adhered to the spirit of the EcoQ initiative as it was understood. However, there are a couple of important considerations which arose in discussion.

First, for reasons explained in Section 3.1.3.2, WGECO is not proposing any EcoQOs for any EcoQs. This group, or other groups of scientists, could provide estimates of ecologically defined positions on the metric of an EcoQ, and inform on the ecological consequences of positions along the EcoQ metric that society may be contemplating using as an objective. However, science groups have no basis for actually choosing the position that society desires on the metric.

Second, this Working Group, and ICES in general, has established its scientific credibility through applying rigorous scientific standards for its advice. The scientific concepts and tools of integrated ecosystem management may start off as somewhat more abstract and much more complex than those used in management focused on a single target species in a fishery or a single contaminant. Likewise, the data and models available for use in setting and monitoring status against EcoQs and EcoQOs may be even more incomplete, contain more sources of uncertainty, and be, at present, less well tested. WGECO did not use these realities as an excuse to lower scientific standards for advancing the EcoQ initiative. This does not mean that a good scientific basis cannot eventually be available for supporting integrated ecosystem management, whether or not EcoQs and EcoQOs are the tools that are used. For now, however, it is important to make the reliability of the scientific basis for progress as clear as possible to those outside the community of experts. When scientific advice is requested on a specific issue, including on a specific ecosystem quality however poorly studied, WGECO, and ICES in general, will provide the best advice possible, pointing out uncertainties and potential weaknesses. In this case, however, WGECO interpreted its task as being asked to make what progress was possible on identifying community-scale EcoQs and suitable metrics for them, maintaining the usual scientific standards of WGECO and ICES.

3.1.4 Issues regarding implementation

3.1.4.1 Lessons learned from past experience

The ongoing development of EcoQOs for the North Sea in various fora, as well as the specific OSPAR requests to ICES to provide recommendations for "appropriate" EcoQ indices for marine mammals and seabirds, evoked a discussion on the added value of this approach, from a scientific standpoint, compared to existing management objectives.

Several existing policies to regulate the effect of anthropogenic impacts on the marine environment have been successful, for instance in diminishing nutrient loads and various sources of pollution. However, at present fisheries are broadly, and probably rightly, seen as having by far the most important impact, not only on commercial fish stocks but also on the ecosystem at large (OSPAR, 2000b). Most target species of North Sea fisheries are overfished, even though in practice, the nature of the overfishing problem is well known. Fisheries science has developed over many years to provide a rigorously defendable advisory framework, wherein the advice provided meets high standards for objectivity, peer review, and consistency (Chapter 1, Sections 1.2.1 and 3.1.3). The advice is primarily based on evaluating the necessary and sufficient conditions for conservation and sustainable exploitation of commercial stocks, using carefully screened data sets and assessment models. Studies of the advice have found patterns of systematic overestimation of future biomass and underestimation of exploitation rates in many fish stocks (van Beek and Pastoors, 1999), indicating that the models and/or data were not perfect. However, even where quantitative details of the scientific advice on fish stock management have been inaccurate or imprecise, technical experts have consistently advised management actions that would have moved the fisheries in the direction of greater sustainability (Serchuk *et al.*, 2000). Nonetheless, overharvesting has continued and for many species the situation has become worse since the Common Fisheries Policy (CFP) was adopted in 1983.

After so long a period with limited progress on eliminating overfishing, it is important to consider what factors have contributed to the lack of progress on a clearly identified and scientifically tractable objective (reduce overfishing). Limitations on fisheries science, the current management system itself, and the current decision-making environment for fisheries are thought to have contributed to the ongoing problems. Limitations on fisheries science may have contributed to continued overfishing directly through the inaccuracies referred to above and indirectly through creating openings for opponents to argue for deferment of action pending greater certainty. The TAC-based management system as presently applied may be intrinsically unsuitable to control fishing mortality on an annual basis. The failure of the system may be partly attributed to TACs having been set too high (EC, 2001), partly to ineffective enforcement and intentional failure of harvesters to comply with management plans, and partly to the multispecies nature of fisheries, which cannot hit several TAC targets simultaneously. In decision-making about fisheries, opponents of fishery restrictions are well organized at least at the local level, know the political system well, and have exploited uncertainties and even small errors in assessments to discredit advice and delay implementation. Given the institutional problems in the policy setting and management of fisheries, even perfect assessments would not guarantee an effective TAC management regime (Daan, 1997).

In the Green Paper on fisheries, the EC now suggests that the solution for failing TAC management may lie in making multiannual and multispecies TACs (EC, 2001). In such a management system, however, any scientific predictions of such quantities will require even more complex models and analyses. These will have an even higher degree of uncertainty than the annual species-specific catch options currently calculated by assessment working groups, and greater opportunity for errors that may not be detected before the advice is provided. Thus, while not making major improvements to other management system and decision-making factors that contribute to overfishing, the scientific advisory challenges have been made greater.

These developments have two important implications. First, to the extent that weaknesses in past scientific advice contributed to the failure of the CFP to achieve sustainable harvesting, future scientific advice has the potential to contain even more such weaknesses. Some steps to use lessons from the past to shore up these potential weaknesses are discussed in Section 3.1.4.2. Second, to the extent that the management system and decision-making process for fisheries are at fault, they require major overhauls. It is in this pessimistic context that the application of the EcoQ and EcoQO initiative to fisheries problems must be viewed. In promoting the ecosystem approach, the Inter-Ministerial Meeting on the North Sea and OSPAR have initiated development of an integrated policy for the conservation of the marine environment. This policy will be debated and enacted in a public opinion climate strongly influenced by public and political frustration over the ineffectiveness of the Common Fisheries Policy to control fishing pressure (Green Paper), as well as concern over the future consequences for the marine ecosystem, should the present situation be allowed to continue indefinitely.

The integration of all relevant management policies, including fisheries, within a single framework is an intrinsic component of an ecosystem approach to management. Such integration makes setting fisheries policy part of a much larger debate, where the legitimacy of many more stakeholders and concerns is indisputable. Placing debates on fisheries policy in this larger framework may mobilize social and political support for conservation issues, and alter the

management and decision-making climate that has failed to prevent overfishing in the last few decades. Even without structural changes the greater support may strengthen the will of policymakers to make effective decisions to reduce overfishing, and the ability of managers to implement and enforce those decisions. However, without structural changes to the management systems that address directly the reasons why the existing legal framework failed to restrict overfishing the benefits of adopting a much broader approach of defining a coherent set of EcoQs and EcoQOs may not be achievable. It is of great concern that EcoQs and EcoQOs are not mentioned in the fisheries Green Paper, which suggests that OSPAR and EU may be on different tracks with their policy development. The different tracks invite questions about the degree of commitment of fisheries managers to move their policy development and management into this larger and more socially inclusive framework of ecosystem management. The institutional changes needed to ensure that this transition occurs are also discussed in Section 3.1.4.2.

In summary, unless fisheries management is brought within the framework that OSPAR is developing, it will not be possible for OSPAR to achieve the goals which motivated it to pursue the EcoQ framework. However, even if fisheries were to come within the framework, many of the reasons why overfishing has continued would not be addressed.

3.1.4.2 Applications of lessons from history to the Advisory and Management System needed to implement EcoQ-based management

As noted in Section 3.1.4.1, the management system within the marine environment has failed in a number of areas. The greatest area of failure that has had an effect at a basin-wide scale has been in fisheries management (OSPAR, 2000b). If ecosystem-based management is to be implemented, consideration of the effects of all human activities on the ecosystem needs to be integrated at the highest policy level. At present, the management of fisheries in the North Sea (and in the wider EU area) is carried out by fisheries ministers who are responsible both for conservation of fish stocks and for promotion of the fishing industry. Policymakers and managers for fisheries are responsible for setting (and accountable for achieving) both conservation objectives for fish stocks and socio-economic objectives for fisheries. Adequate structures or mechanisms are not in place to reconcile discrepancies that arise now between either the conservation and socio-economic objectives within fisheries, or in future between conservation objectives for fish stocks and the more encompassing integrated ecosystem objectives. The decoupling of those responsible for setting and delivering conservation objectives from those responsible for setting and delivering socio-economic objectives is one possible step towards a system where more integrated ecosystem management could be pursued. This would still not resolve the problems presented by the absence of mechanisms to reconcile discrepancies among objectives set for fisheries conservation and those set for integrated ecosystem management, were any to occur (Symes and Pope, 2000). In fact, it might reveal a need for a mechanism to reconcile discrepancies between objectives set for conservation of fish stocks and socio-economic objectives set for fisheries. If the current fisheries policy and management framework in the North Sea were merely provided with objectives relating to the ecosystem derived by OSPAR, institutional changes to increase the accountability of managers to meet those additional objectives might be needed as well, in order to have a high likelihood of achieving more integrated ecosystem-based management and better management of fisheries.

Applying the past experience of WGECO, a number of needs and opportunities for improvement of the science and advisory systems can be identified. If ICES is to be involved in the monitoring and assessment of different EcoQs, it is important to establish a peer review and advisory framework that deals explicitly with quality control of data collection and analysis. As noted in Section 3.1.4.1, despite strict protocols, great collective experience, and high vigilance, occasionally poor data and some errors in stock assessments escape the review by both working groups and advisory committees. Although it is possible at this stage to define and propose metrics that meet the available selection criteria and, combined, may provide a broad picture of the health of the system (Section 5.5), any metric may be calculated from a variety of available data sets that have not been collected for this particular purpose. Moreover, subtle variations in algorithms for calculating indices may sometimes have a significant influence on their performance. Given that EcoQs and EcoQOs, once adopted, are altered only periodically, recommending a particular metric is technically demanding and more complex than it may initially appear. Once a metric and the reference levels associated with it have been selected, review and advisory groups with the skills of the best assessment working groups, but even greater breadth of knowledge and expertise, will be essential if management based on the EcoQOs is to have a sound scientific foundation.

There are clearly far more potential metrics of EcoQs that could be used in management of the North Sea than are practical, given available funds for monitoring and assessments. OSPAR will have to make some choices among them, but once made, there are a number of science activities that must be done. Scientists should carry out a sensitivity analysis of various methods and data sets to select on technical grounds the optimal combination for future use. This step alone may require further interaction with OSPAR, if the detailed technical review reveals unforeseen, but crippling technical problems for some preferred metrics of ecological quality. Once EcoQ metrics, data standards and calculation algorithms all have been decided upon, relevant data sets for each of them must be collected and analysed periodically. Both processes require quality control to ensure that any advice derived from such data is perfectly defendable.

There is still considerable uncertainty about the effectiveness with which such metrics may in practice measure the response of the system to human impact. Therefore the research community should work with the science advisory and management framework explicitly to explore the occurrence of true hits as well as false alarms and misses in historic series of the EcoQ metric and human activity. Also, it is important to ascertain that the metrics match the set of potential impacts that management measures can address, and to evaluate the performance of EcoQO-based advice over time in improving management decision-making and actions.

Once the metrics have been selected, monitoring and analyses completed, the results subjected to peer review, and advice developed, the scientific advice will be given to a management system which has thus far proven unable to solve the relatively simpler problem of controlling overfishing, given advice on the fishery and target stocks. Even with the structural changes discussed above, there are specific problems of science advice that should be addressed:

1) The selection of "appropriate" EcoQOs is not straightforward (Section 3.1.3), partly because what is "appropriate" cannot be singularly defined scientifically, and partly because there is incomplete scientific knowledge about what aspects of an ecosystem are necessary and sufficient for its conservation. Compared to single-species fisheries advice, where keeping spawning biomass large, and exploitation rates low, is likely (but not guaranteed) to keep harvesting sustainable and to conserve stocks, guides to successful ecosystem management are less clear. Given the complexity of marine ecosystems, there are many properties that one might argue need to be conserved and a nearly infinite number of potential metrics of these properties. It is clear from a pragmatic point of view that we have to be selective, and have to select wisely. Although it is relatively easy to formulate important selection criteria for EcoQ metrics (Section 5.4), applying these over a wide scale of potential metrics is by no means straightforward.

2) More importantly, the approach chosen by OSPAR deviates from the existing one for commercial stocks, because in the OSPAR framework the EcoQO (the target) is to be set relative to the current level and to a reference level that should reflect a situation when anthropogenic impact was minimal (with allowance for a pragmatic approach), rather than a limit reference point (LRP) referring to conditions considered not sustainable and posing unacceptable risk to the resource (Section 3.1.2). In fact, for many potential EcoQ metrics it will be hard, if at all possible, to define a level associated with "unsustainability" or otherwise with an unacceptable threat to the ecosystem. In the EcoQ system, the possibility of large numbers of metrics combined with poorly determined conservation limits on many of them will make any scientific advice even easier to contest by stakeholders and also by other experts. Current fisheries advice formulated in the sense of keeping the impact below some unsustainable level is obviously much easier to defend than EcoQ-based advice that points to some current and historic values whose distances from a LRP are known only vaguely or not at all. The resultant lack of defensibility might well further reduce rather than enforce the impact of scientific advice on management and therefore could easily undermine the advisory role of ICES.

3) By definition, any broad EcoQ metric for a community reflects the ecosystem response to a broad set of human impacts, and therefore the contribution of each activity to its present value may not be singled out easily. In fact, any particular value of a metric of an EcoQ may arise from completely different combinations of different impacts. This will make it much more difficult to predict how the metric will respond to various options to reduce one particular impact, and to assign responsibility (and associated costs) among possible contributors, when a metric does indicate a conservation problem. On these grounds, EcoQs and their metrics selected because they are responsive to a specific threat seem particularly useful (although see Section 3.1.3.3.1).

Although the approach seems promising in principle, embarking on giving advice on EcoQOs will set high demands on developing a rigorous and defendable advisory framework, which will take considerable time. Therefore, it would seem wise to concentrate on developing a suite of EcoQ metrics first and to test their performance particularly with a view to defining potential LRPs before endeavouring recommendations on EcoQOs. It is likely that management systems, as well as science advisory systems, must also adjust to new and greater demands on their effectiveness, if they are to be able to enact and enforce management measures based on the best ecosystem advice possible.

We cannot know now the detailed organisation and procedures for the management system that will actually create and implement the management policies and plans based on the scientific advice regarding status of ecological features relative to their target levels, as measured by the metrics and EcoQOs. However, that process must function much more effectively than the current one, for progress to be made on the pieces (the individual EcoQs) and for this process to actually result in effective ecosystem management, leading to improved ecosystem quality.

3.1.4.3 Practical considerations regarding making EcoQs work together for integrated management

The OSPAR decision to proceed with identifying EcoQs separately for ten issues permits possibly hundreds of EcoQs to be proposed, in order to guarantee that the entire marine ecosystem and all the processes that operate within it are covered. Although this decision was considered to be pragmatic (Scheveningen Workshop, Anon., 1999), each EcoQ would have at least one EcoQO to be monitored and managed. Currently, fisheries managers struggle to address adequately targets for 14 annually assessed commercial fish and benthic species in the North Sea, along with the additional seven non-assessed species, or species groups, for which TACs are set. Add to these the need to account

simultaneously for EcoQOs for threatened and declining species, seabird and marine mammal species, fish and benthos communities, habitats, and two ecosystem process issues, and the task of managers becomes much more complex. Where management actions will be necessary, some may be difficult, costly, and/or controversial, and for reasons of logistics or politics, it may not be possible to implement them all at once. This creates at least two classes of problems: assigning priorities and achieving intercompatibility.

The requirement to rank these EcoQs and EcoQOs so as to be able to choose which to pursue aggressively and which to defer, seems inevitable. Where much effort has been invested in gaining social consensus on EcoQOs on which different sectors of society placed different initial values, and the achievement of which will demand differential subsequent costs, opening a second debate on the priority of that EcoQO relative to others may be divisive. It needs to be clear in advance whose task it will be to carry out these ranking and reconciliation exercises. What will happen to the EcoQOs which are ranked low or are incompatible?

As the number of EcoQOs increases, so does the risk of redundancy or, more seriously, mutual incompatibility. In attempting, for example, to restore commercial fish stocks, and fish and benthic communities to some improved state, the population dynamics for some seabird and marine mammal species maybe affected in such a way as to, at the very least, inhibit future population growth, if not cause actual population declines. In considering such potential conflicts, the logic behind the different objectives needs to be carefully maintained. The goals for commercial fish stocks and fish and benthos communities appear, at the very least, to be to return the system to a state characteristic of several decades ago. Some seabird species are currently at population sizes many times higher than they were at the start of the twentieth century. Much of this increase has been attributed to fishing activity: the provision of additional food resources at key times of the year through discarding, the increase in the abundance of small fish in the assemblage through size-selective fishing, and the removal of large predatory fish that may have competed with seabirds. Changes within the fish components of the ecosystem to a greater proportion of larger fish and fewer discards may render the North Sea a much more inhospitable place for some species of seabirds. Are EcoQOs for seabirds likely to reflect this, and allow for significant declines in some of our most abundant seabird species? Or will they be set so as to try and conserve the current state?

These difficulties are nearly unavoidable, if EcoQs for the ten EcoQ issues are developed and implemented independently. This decision may prove to have been pragmatic from the point of view that it by-passed the enormous hurdle of determining one (or at most a few) holistic ecosystem objectives, if such even exist, and so allowed the process to proceed quickly. However, the same hurdle may simply be encountered later, when it comes to putting the process into practice. At that point it will be necessary to gain social consensus on ranking which EcoQOs to pursue most aggressively, and on compromises to reconcile incompatible EcoQOs. Because these are human issues, clearly social scientists need to be more involved in the EcoQ and EcQO initiative.

To balance this pessimistic view, there are some potential steps forward. Short of the grail of one (or a very few) all-encompassing EcoQ and EcoQO, some simplification of the implementation task can be achieved by recognizing opportunities, if they exist, for one EcoQ to address more than one of the ten issues. This may be practical, regardless of whether one believes that a single well-chosen community-scale EcoQ may protect many species of fish, seabirds, marine mammals and benthos, or that an EcoQ for a well-chosen species, sensitive and vulnerable to several threats, may ensure the ecological quality of many other species and the larger community of which it is part. Also, a policy framework is developing that may guide ranking and reconciliation of EcoQs. The 1997 Intermediate Ministerial Meeting on fisheries laid down some guiding principles that require the development of an ecosystem approach to management, taking account of critical ecosystem processes, and involving a multispecies approach. This will be difficult or impossible to realize without giving priority to EcoQOs that are related to OSPAR's communities and ecosystem process issues, even if they are difficult to make operational.

3.2 Ecosystem Properties and EcoQ Metrics

3.2.1 Background

The Convention on Biological Diversity (CBD), signed at the 1992 UNCED in Rio, provides the principal framework for international efforts to protect natural resources. The CBD defines biological diversity as *"the variability among living organisms from all sources including,* inter alia*, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems"*. This definition recognises, therefore, two components of biological diversity: the biological composition (itself divided into three levels – diversity among ecosystems and habitats, diversity of species within an ecosystem or habitat, and genetic variation within individual species) and the preservation of the ecological complexes of which they are part, that is to say ecological functionality.

The 1995 Jakarta Mandate on Marine and Coastal Biological Diversity (Conference of the Parties decision II/10) highlighted five (now six) thematic areas, second amongst them being sustainable use. The text specifically calls for "*the present mono-species approach to modeling and assessment should be augmented by an ecosystem process-oriented approach, based on research of ecosystem processes and functions, with an emphasis on identifying ecologically critical processes that consider the spatial dimension of these processes*". The ecosystem approach is further defined (Conference of the Parties Decision V/6) as "*the application of appropriate scientific methodologies focused on levels of biological organisation, which encompass the essential structure, processes, functions and interactions among organisms and their environment. It recognises that humans, with their cultural diversity, are an integral component of many ecosystems*". This again emphasises the need to consider not just protection of the full inventory of taxa present but also protection of ecological processes and explicitly the spatial elements of these processes.

This section provides major concerns to be addressed when specific EcoQs and their metrics are sought on each of the three key properties of the system.

3.2.2 Biological diversity

Most metrics of biological diversity can be derived from observations collected routinely during surveys. However, they never reflect the "true" diversity within the ecosystem, but rather the diversity as observed in the image of the community as viewed through the sampling gear. This picture is unavoidably distorted by species-specific differences in catchability, the absence of information on grounds that are difficult to sample, differential ease or vigilance in species identification, etc. If such metrics are used as an EcoQ, the inherent assumption is that any relative change in the survey metric greater than the sampling variance mirrors a true relative change in the ecosystem. In practice, surveys carried out with different methodologies (or different geographical extensions) may be expected to show differences in the same metric. Clearly, our ability to make conclusive statements about perceived changes in an EcoQ would be greatly enhanced if at least two independent surveys could be used to calculate the same metric. If these estimators would show similar annual deviations and trends, our confidence in measuring the true EcoQ of the system would obviously be increased.

3.2.3 Ecological functionality

Metrics of ecological functionality are even more problematic, because they can only be based on integrated sets of observations from different sampling programmes, each of which may be biased in specific ways. For many aspects of functionality, additional tropho-dynamic modelling is required to obtain the functional responses of the ecosystem and its components. Consequently, metrics of ecological functionality reflect modelling results rather than direct observations. In practice, any metric will be at least partly influenced by model assumptions even when model inputs are regularly updated with new observations, and the interpretation will often be open to scientific debate. Also, it is much more difficult to get independent confirmation, unless a suite of models with alternative assumptions is available and the robustness of model outcomes has been tested and found to be high.

3.2.4 Spatial integrity

Ecosystems may be defined at many spatial scales, but within the OSPAR context they apply to relatively large scales ("Large Marine Ecosystems"), that integrate over many sub-systems (pelagic vs. demersal; shallow vs. deep water; etc.). In fact, the spatial integrity of the different sub-systems could be viewed as an important element of total ecosystem quality. Spatial statistics are a specialized field (Ripley, 1988), and metrics derived from that field have not worked their way into most ecological practice. However, attention must be drawn to the fact that many metrics specifically apply to particular sub-systems (metrics derived from trawl surveys, for instance, provide specific information on demersal fish communities in muddy and sandy areas that can be trawled). Such restrictions may in fact favour the ability to assess some impacts on spatial integrity aspects of EcoQs. For example, changes in a metric directly related to the spatial impact of bottom trawling are effectively derived from the same suite of species as represented in the survey. If EcoQs and EcoQOs are to be effective tools for conservation of spatial integrity of ecosystems, however, an integrated and comprehensive set of quality metrics for spatial integrity are required, covering the entire suite of impacts caused by human activities.

3.2.5 Metrics

WGECO drew upon the group's collective experience to generate a list of key ecosystem properties relating to Biological Diversity, Ecosystem Functionality, and Spatial Integrity (Table 3.3.4.1). For each property, it went on to list at least a few key metrics. For some properties, there are very large numbers of possible metrics, often differing in only minor details. These lists are not exhaustive but cover examples of the most widely used metrics in each family. Nor did WGECO conduct an exhaustive critique of the relative merits of alternative metrics for various properties, a task which has been done many times before both by this group (ICES, 1995, 1996a), and in publications (for reviews see Hollowed *et al.*, 2000; Rice, 2000). Rather, where possible WGECO chose metrics that were either in widespread use,

or were recommended by recognized experts for certain fields of study, expecting that among equal alternatives, users of EcoQOs would prefer metrics with both of those features. WGECO thought that the most important task was to develop a rigorous and sound approach for identifying particularly promising or dangerous types of metrics, identify important community properties for which promising metrics were not available, and refine the selection subsequently. The selection of metrics for use in actual management would be done subsequently, by appropriate expert groups, using the approaches identified here.

Aside from spatial integrity, WGECO is satisfied that no really major aspect of biological diversity or functional integrity would be missed by the properties and their metrics as tabulated in Section 3.3. WGECO also specifically considered and rejected calling the list of properties and their metrics either necessary or sufficient to, individually or in combination, ensure conservation of ecosystems, were they implemented in an EcoQ/EcoQO framework. Rather, each metric should be evaluated on its merits, with a watchful eye for redundancies, potential synergies, and gaps among promising metrics.

WGECO specifically assumes that high standards of quality control are applied at all stages of collecting data and conducting analyses to produce values of a metric (whether reference values or estimates of the present state of the system). Even the best metrics cannot withstand poor practice. Some metrics are especially vulnerable to distortion by even minor weakness in data sets or analysis approach, and such vulnerability must be considered when selecting metrics for use in reflecting EcoQs.

3.3 Evaluation

3.3.1 The evaluation method

In order to provide a unified framework for comparison of approaches, WGECO developed a cross-tabulation approach. We began by listing the ecological qualities that might be threatened by anthropogenic activities. These were considered in three categories: issues relating to biodiversity of species, to ecological functionality, and to spatial integrity of ecosystem properties. For each of these, we then listed a number of classes of metrics of that property. Each of these was then independently ranked by WGECO members against the eight criteria developed from those used by WGSE, WGMMPH, and Piet (2001) in the draft EcoQOs for fish (see Chapter 2 Section 2.2.1). These criteria were designed to cover the utility of the metric both as an accurate measure, a property responsive to management action, and its communicability. All metrics that were considered could provide ecological information of great utility in the consideration of ecological dynamics and processes. For use as EcoQ metrics, however, the key issue was to determine which of the metrics at this time could form a basis for management given current levels of knowledge. In addition to selecting metrics, we also highlight areas where further metric development is required, either because no metric currently exists or because those available do not fully meet the criteria and so require additional development.

3.3.2 Criteria for good Ecological Quality metrics

The concept of ecological quality objectives (EcoQOs) has been discussed in a number of documents and at a number of recent meetings (Anon., 1999; Lanters *et al.*, 1999; ICES 2001a, 2001b; Kabuta and Enserinck, 2000; Piet, 2001). Several key features of EcoQ metrics may be derived from these discussions. These may be summarised as follows:

Metrics of EcoQs should be:

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

In addition, an EcoQ metric may:

•   Relate to a state of wider environmental conditions.

These eight properties were all deemed desirable in a metric of EcoQ but were not all regarded as essential properties. The eighth was considered to refer to the information content of the metric rather than being a necessary quality. We therefore did not employ these criteria in our screening process.

3.3.3 Properties and metrics considered for fish and benthic communities

In the following annotated list, a number of properties of fish and benthic communities are reviewed and for each property one or more potential metrics are proposed. In all cases our assumption, in discussing a metric, is that it has been correctly calculated based on an appropriate data set.

3.3.3.1 Biodiversity of species

3.3.3.1.1 Biomass

Sum of weights across species from survey

The total biomass of organisms sampled, standardised for effort, from a region is an informative measure of its long-term productivity, and changes in long time-series data sets show a particularly useful broad scale change.

3.3.3.1.2 Size structure

Slope size-structure

Sheldon *et al.* (1972) showed a log-linear relationship between fish biomass and size. In spite of the differences in numbers and size between species, the community as a whole shows a log-linear decrease of biomass with increasing size. The slope of this relationship is assumed to reflect the efficiency of energy transfer and the mortality rate and can be used as a metric of the size-structure. Although several alternatives have been suggested since its introduction (Borgmann, 1987; Boudreau and Dickie, 1992; Boudreau *et al.,* 1991; Thiebaux and Dickie, 1992, 1993; Sprules and Goyke, 1994), the conceptual basis is widely recognized (Rice and Gislason, 1996).

The general formula for the log-linear relationship between size and biomass is:

$\ln(y) = a * \ln(x) + b$
where: x = size, y = biomass or number, a = slope, b = intercept.

A disadvantage is that slope and intercept are not independent, which makes it difficult to interpret a time-series of either one. Also, an arbitrary choice must be made about the minimal size of fish that should be incorporated in the linear regression; depending on the mesh-size of the gear, certain size classes will be underrepresented and thus disturb the relationship.

Rice and Gislason (1996) studied the log-linear relationship for the North Sea fish community (1975–1995) and observed a change in slope caused by a decrease in large fish. This change was attributed to the impact of fisheries. Gislason and Lassen (1997) showed that a linear relationship between fishing effort and the slope of the size spectrum can be expected. WGECO (ICES, 1998) reported that there is now sufficient theoretical and empirical evidence to be confident that changes in fishing mortality should result in a long-term change in the slope of the size spectrum. Provided that growth and relative recruitment of the constituent species do not change, the change in the slope should be directly proportional to the change in exploitation rate of the community.

Length-frequency distribution

The length-frequency distribution of the community is determined by summing up the number of individuals caught per size class. In most cases these size classes will be cm-classes. A relevant metric to represent the length-frequency distribution may be the total number or weight of the community above a specific length threshold. Another relevant metric that may be derived from the length-frequency distribution is the percentage composition of groups that cover certain size ranges.

Multi-dimensional ordination

For studies involving complex tabular data (commonly *i* rows as sampling sites, *j* columns containing species or size-classes and cell entries of (transformed) abundances of species or size-class *j* at site *i*), ordination methods can be used to reduce this complexity to a small number of (usually) orthogonal (i.e., not correlated) gradients (reviewed in Jongman *et al.*, 1987). Several ordination methods exist such as Principal Components Analysis (PCA), Correspondence Analysis (CA), and Non-metric Multidimensional Scaling (MDS). Of these methods, MDS has become the preferred technique for ecological ordinations of fish communities because of its increased robustness in the face of irregular distributions of abundance and high sampling variance (Clarke and Ainsworth, 1993; McRae *et al.*, 1998). Although this technique may reveal patterns or trends that would otherwise remain obscured, interpretation or linking them to useful management information proves difficult, and communication to non-scientists challenging, to say the

least. Although ordinations are listed under size structure, ordinations on the basis of species abundances as well as frequencies of size classes are common, so there could be ordinations of species identities.

3.3.3.1.3   Species identities

Species presence/abundance

There are several informative measures of community structure that do not take into account the species identities of the community. It is conceivable therefore that changes to species presence or absence may go undetected unless reference is made to lists of species relative abundance.

Index of rare species

Variability in abundance of the uncommon species in a survey can illustrate underlying patterns of change that are not evident from analysis of the dominant parts of the community. For example, the presence of unexpected migrants or the decline in population size of less common species can be used as metrics of previously unobserved adverse human impact. Daan (2001) proposed a spatial and temporal diversity index that was based on species rarity.

Index of declining or increasing species

A variety of metrics are available based on the proportion of species in the community which are showing increases or decreases in abundance (biomass). These measures are at best coarse and may provide little information about causes of the changes, but are readily interpreted and understood by non-specialists.

Presence of indicator, charismatic, sensitive species

Societal concerns about the environment often focus on a limited number of organisms that are in some way "attractive". Such charismatic species, including dolphins, killer whales, large sharks, and a variety of seabirds, are often viewed as sentinels of the health of the ecosystem. The scientific justification for such a view varies with the species, but as many are higher predators and long-lived they will often be more sensitive to human impacts. Indicator and sensitive species are selected on the grounds of criteria that explicitly use their known response to impacts. Many examples of such indicator taxa exist in the pollution literature (Pearson and Rosenberg, 1978) and a limited number of benthic taxa have also been suggested as being vulnerable to direct effects of fishing (Lindeboom and de Groot, 1997). Development of this approach is often more difficult than it at first appears as lists of sensitive/indicator taxa are rarely transferable between regions and developing the list from the impacted system studied leads to circularity.

Non-indigenous species

The presence of non-indigenous species, used here to mean species introduced by anthropogenic activities rather than natural invasions/range expansions, is by definition a failure to maintain "natural levels of biological diversity". For larger organisms, the presence of non-indigenous species is easily recorded; for lower organisms, our lack of knowledge of pristine fauna makes this more difficult (Eno *et al*., 1997).

Species turnover/loss rates

The rate at which species composition changes from year to year in samples taken in a consistent manner and location is a widely used metric in terrestrial conservation biology. It requires consistent and reliable sampling where sampling is expected to detect most of the species that are present. Measures of turnover rates are most effective at local scales, and may be less effective at the scales of large marine ecosystems when many samples are pooled.

3.3.3.1.4   Species diversity

The concept of species diversity has a long history in the ecological literature; countless different metrics have been devised and utilised in numerous different studies covering taxa from just about every phylum in the plant and animal kingdoms (Brown, 1973; Connell, 1978; Davidson, 1977; Death and Winterbourn, 1995; Eadie and Keast, 1984; Heip *et al*., 1992; Huston, 1994; MacArthur and MacArther, 1961; Magurran, 1988; May, 1975; Rosenzweig, 1995; Washington, 1984). Despite this long tradition, and perhaps in part due to the proliferation of different metrics, species diversity as a concept has been questioned (Hurlbert, 1971). Hill (1973), however, argued that much of the perceived difficulty with the concept lay in the fact that it combined the two characteristics of richness and evenness. The theoretical underpinning of the concept has been discussed (May, 1975, 1976). The ability of the different indices to actually detect environmental and anthropogenic influences has on occasion been questioned (e.g., Robinson and

Sandgren, 1984; Chadwick and Canton, 1984); in general, however, these problems have usually been associated with inadequate sample size (Soetaert and Heip, 1990).

WGECO considered several species diversity metrics as candidates on which EcoQOs could be based. The simplest representation of the species relative abundance data, on which any metric of species diversity is based, is the straightforward graphical representation of relative abundance on species abundance ranking. The most commonly used representation of this type is the *k*-dominance curve (Lambshead *et al.*, 1983; Clarke, 1990). This index was endorsed by WGECO because of the simple, easily comprehensible way that it conveyed the information, avoiding the problems of trying to convey both aspects of species diversity in a single numeric parameter. Well-defined statistical methods for determining differences between samples have been developed (Clarke, 1990). The *k*-dominance curve was the only metric to receive a positive score for all selection criteria.

Hill's N numbers

Hill (1973) suggested that several of the most commonly used diversity indices were mathematically related, forming a family of indices varying in their sensitivity to species richness and species evenness (Peet, 1974; Southwood, 1978). These indices are all affected by sample size, which is a major disadvantage with regard to monitoring change in marine ecosystems where sampling is logistically difficult and expensive. As the Hill number notation increases, the index moves from being a measure of species richness to one of species dominance. Low N number metrics, e.g., N0 and N1, are consequently the most affected by variation in sample size. When the problem of variable sample size can be addressed, these metrics have been used to demonstrate long-term temporal and spatial trends in species diversity that have been associated with differences in fishing activity (Greenstreet and Hall, 1996; Greenstreet *et al.*, 1999).

Taxonomic Diversity Indices

Taxonomic diversity indices were developed by Warwick and Clarke (1995, 1998). They are closely related to the Shannon-Weiner Index, but they also provide additional information with respect to the level of phylo-genetic relationship present in samples. As such they were considered to convey some information on the genetic diversity aspect of biological diversity. They have been demonstrated to be relatively sample-size independent, and to be sensitive to ecological perturbation in circumstances where other species diversity metrics, such as the Shannon-Weiner, or Simpson's Indices, fail to respond. They are, for example, particularly sensitive to situations where a group of particularly vulnerable, closely related species may be in decline and being replaced by alternative, unrelated species. The impact of fishing on elasmobranch fish species is an example of this (Rogers *et al.*, 1999). However, in circumstances where Hill's N1 and N2 are varying, these taxonomic indices may convey little additional information (Hall and Greenstreet, 1998).

Theoretical Distribution Metrics

Log-Series and Log-Normal: Parameters derived from these distributions have the advantage of being relatively sample-size independent (Kempton and Taylor, 1974). Also, there has been considerable debate in the ecological literature regarding the theoretical reasons as to why distributions of species relative abundance should follow either one of these models (Fisher *et al.*, 1943; Preston 1962, 1980; Kempton and Taylor, 1974; May, 1976). One major difficulty with using these indices lies in the necessity to fit the data to the distributions, to estimate parameters of the distribution for subsequent use. Generally this tends to require a substantial amount of data, rather negating the advantage of sample-size independence. Often fitting the data to the distribution proves to be difficult, and in testing the significance of any fit, one hopes not to disprove the null-hypothesis, which is unsatisfactory from a statistical perspective.

Species-Effort Index

Many scientists have argued on theoretical grounds that species richness (e.g., N0) is the most important aspect of species diversity, but the sampling effort required to estimate this adequately from the data normally available from fish or benthic surveys is usually prohibitive. WGECO considered that a species-effort index derived from the parameters of the function describing the rate of increase in the number of species recorded as samples from a survey and that are increasingly aggregated may offer a solution. This function is exactly equivalent to the species-area relationships of the form $S=cA^z$, which describes species richness in habitats of varying size, e.g., islands, continents (Rosenzweig, 1995). The two parameters, c and z, could perhaps be derived from a much smaller number of trawl samples to provide a relatively sample-size independent estimate of species richness.

3.3.3.1.5 Life history composition

There is extensive theoretical literature that distinguishes K-strategists from r-strategists, that is, species whose life history characteristics adapt them to living in undisturbed, stable environments vs. those adapted to living in frequently

disturbed, variable environments. Particular life history characteristics can be used to place species somewhere along this continuum, and thus provide an indication of vulnerability to disturbance by additional fishing mortality. Correspondingly, the life history character composition of communities may provide a metric of the past impact of fisheries on that community. Possible life history characteristics that might be used as such metrics include:

- maximum size (cm);
- size above which 50% of the population is mature (cm);
- maximum age (year);
- age above which 50% of the population is mature (year);
- fecundity expressed as number of eggs per female or number of eggs per body weight;
- parameters k and $L\infty$ of von Berthalanffy growth curve.

Values for one or more of the above parameters are available for many species from the literature. This list, however, is far from comprehensive and for several of the parameters values are available for only a few species. Community metrics based on these parameters are calculated per year by weighting the community species' biomasses with the value of that particular life history parameter.

Other potential metrics might be derived from sex ratio, lifetime reproductive output, or growth rates.

3.3.3.2 Ecological functionality

3.3.3.2.1 Resilience

The concept of resilience refers to food webs as a whole (Pimm, 1982; Cohen *et al.*, 1990). The concept addresses the ability of the web as a whole to retain its overall configuration when stressed, or to return to its original configuration when perturbed. Food webs can suffer several types of stresses and perturbations, including invasions by new species, loss (extinction) of species in the web, and large, abrupt increases or decreases in abundances of one or more species. There is much theoretical detail about what properties of food webs do (or do not) make food webs (and the ecosystem that they represent) amplify or damp stresses and perturbations, and about what constitutes an important response by the food web. For use as a general metric of food web (ecosystem) quality, however, the diverse expert argumentation consistently suggests that "healthy" food webs (ecosystems) maintain their general configuration when moderately stressed or perturbed, whereas badly altered ones may undergo dramatic restructurings by the same degree of stress or perturbation. There are, of course, the usual problems with potential circularity of the concept, and concerns that Null Hypotheses are often poorly formed when the concept has been tested with models or in the field. Theory about **resilience** of food webs has identified a number of potential metrics. The ones considered by WCECO include:

Return time of properties of food webs

This refers to the number of time steps required by a food web to return to its original configuration when perturbed in some specified way. Stable food webs should have short return times, and return times increase as food webs lose properties that confer stability. The parameter for which return time is measured depends on the model or study, and selection of the parameter can affect the results. If the metric is used as a measure of ecological quality, it is also necessary to decide whether the state to which the food web (ecosystem) should return is a recent state, or a state thought to persist historically.

Invasibility

The likelihood that a new species can establish itself if introduced into an existing food web. Sometimes the measure differentiates cases where a successful invader can be established without loss of any species in the original food web. At other times the measure includes the degree to which membership of the previous web was changed by a successful invader. Invasibility depends, of course, on the characteristics of the "species" introduced, so this property is usually explored through intensive simulations. Such simulations have demonstrated that some configurations of food webs are more likely to allow invading species to be established than others, and some configurations of food webs are more likely to lose existing species when an invader is established than others. Field studies sometimes have confirmed predictions from theory, and other times have not. It is generally argued that as communities co-adapt to particular environmental conditions, invasibility of food webs should decline, and when food webs are stressed invisibility may increase.

3.3.3.2.2 Productivity

Although there are many ways to measure productivity, the basic concept is the amount of new material produced by some level of biological organization. Productivity has been discussed sometimes at the scales of individual (growth), but more generally at the scale of species (increase in numbers and/or biomass), and ecosystems. At the scale of

ecosystems, primary productivity (fixation of carbon by plants) is generally differentiated from secondary productivity (passage of carbon [or other currency] through the food web). System productivity is also often partitioned into "new" production, due to nutrients taken from inorganic sources, and "regenerated" production, due to recycling nutrients already in the food web. There is again much theoretical detail in this area (Cushing, 1995; Steele, 1998). In the context of maintaining ecological quality, however, the property is considered quite broadly. Ecosystems that are highly productive, producing lots of biomass, energy, and/or individuals are considered to be in "good" condition with high ecosystem quality (unless excessive nutrient inputs cause eutrophication). As the quality of the ecosystem (or any of its components) is degraded, its productivity can decrease, and less "stuff" is produced.

Secondary production occurs in the water column (zooplankton) and on the seabed (benthos). On-site measurements of secondary production in the North Sea of all seabed animals have not been made, also due to the lack of adequate methods. Only sporadic measurements have been executed into the secondary production of specific species. The fish community in the North Sea is situated on the third and fourth trophic levels and as such is dependent on the production of the underlying levels. The total fish production can best be determined based upon stock assessments of all the fishes occurring in the North Sea. However, stock assessments have only been made of a number of commercially important species, but they do form a significant share of the total fish biomass. An estimation of the total fish production is the sum of the somatic fish production and the production of gonads.

P/B ratio

The ratio of production of some part of an ecosystem to the standing biomass of the same part of the ecosystem. This can be measured for a population, a suite of species, a trophic level, or any other grouping that researchers can quantify and justify.

Carbon per unit area/time/volume

In general, productivity is expressed as the fixation of amount of carbon per area per time unit (e.g., a regular expression for primary production is for instance g C per m^2 per year).

Partitioning of production between somatic and gonad material

This in effect follows on from the discussion on life history characteristics above. As the community shifts towards domination by r-strategist species, the partitioning of production between gonadal tissue and somatic tissue should shift from investment in somatic material to investment in gametes. This follows on from the nature of the two types of strategists. K-strategists invest in growth because they intend to remain for a long time in a stable home. Conversely, r-strategists tend to have small body sizes. Instead, they mature early so that, from that point, they cease investing heavily in growth, directing their resources to producing gametes instead. This buffers them from perturbation in the environment, ensuring that they can recolonize an area, or colonise an alternative area. Consequently, in a community disturbed by fishing, one might expect a shift in the ratio of gamete:somatic production.

3.3.3.2.3 Trophic structure

Trophic structure is a general term for the feeding relationships among species in a community and ecosystem. Theory on trophic structure has a long history and can be quite complex (Pimm, 1982; Cohen *et al.,* 1990; Hall and Raffaelli, 1991; Rice, 1995; Thingstad, 1998). In general, however, trophic structure is thought to be a major component of how communities and ecosystems maintain their integrity. Abundance of individual species within a trophic system may change due to human perturbations, environmental forcing, or the trophic (predator-prey) relationships themselves. The trophic structure is some consolidated or emergent statement about how the relationships among the species respond to those changes in abundance, whether tracking them proportionately, amplifying them, or buffering them. Trophic structure is often expressed for aggregates of species, often grouping species into levels sharing a common number of trophic transfers: primary producers being the first level, their grazers being a second level, predators on grazers being a third level, etc. Because feeding is strongly size dependent in marine ecosystems (see Size Structure), these groupings are generally severe abstractions of reality. Nonetheless, they form the basis for most analyses of trophic structure.

By representing the relationships among predators and prey, trophic structure is considered fundamental to ecosystem functioning. Human actions that alter trophic structure are generally considered to degrade ecosystem quality, particularly if the change simplifies the structure in some way, such as reducing linkages among species or the proportion of total biomass at any level.

Distribution of production among trophic levels, size classes, taxonomic groups

This represents a class of metrics that are simply the frequency distribution of productivity (measured as biomass, calories, etc.) across a number of groups of species to another, where the grouping criterion could be trophic level, size classes, etc.

Connectance

The connectance index in a food web is the ratio of the number of actual predator-prey links to the maximum number of possible links, where different modellers have applied slightly different approaches to determining the theoretical upper limit. Christensen *et al*. (2000), for example, estimated the number of possible links as $(N-1)^2$, where N is the number of food web groups.

Path length

This is a measure of the distance, measured as number of linkages, between selected species (or nodes, if species are aggregated in a food web model). Different researchers have used the mean number across all linked species, or the distance from primary producers to top predators, as the maximum number of steps possible in a model as the metric for estimating the path length of a food web. Christensen *et al*. (2000) estimated path length as the average number of groups that an inflow or outflow passes through in their models.

Ratios of trophic levels

This represents a class of metrics that are simply the ratio of biomass or productivity (measured as biomass, calories, etc.) of group of one species to another, where the grouping criterion could be trophic level, size classes, etc. There are as many possible metrics of this property as there are ways to group species and things which reflect their role in the ecosystem. Intended usage, data availability, and professional experience will guide the selection of grouping criteria and things to express as ratios.

3.3.3.2.4 Throughput

This property reflects the rate at which energy or biomass is passed through the ecosystem. It is influenced by ecological efficiencies of the species in the web, the numbers of linkages among species, and mortality rates. It is an important property of ecosystems, but to use it would require data not likely to be available without significant preparatory work, and probably much new directed research. Therefore, WGECO did not give prominence to metrics of it, such as:

Internal consumption to yield

The ratio of energy lost to the system through respiration and bioenergetic needs of the individuals in the web to the energy removed by the fishery.

Ulanowicz index

In his textbook on bioenergetic ecological models, Ulanowicz (1997) has a specific index that reflects throughput of energy in a food web. The Working Group was aware of the index, but lacking energetics data this metric was not pursued.

3.3.3.2.5 Body well-being

Condition factor

In fish ecology, condition is believed to be a good metric of the general "well-being" or "fitness" of the population under consideration (Adams and McLean, 1985). This can also be expected to apply at the level of the community. Several condition indices are used in fishery science as metrics of the length-weight relationship of a population. However, the conversion of a two-dimensional length-weight relationship into a single statistic results in a loss of information and, in many cases, an inaccurate representation of that relationship. After review of the most common condition indices by Bolger and Connolly (1989), Cone (1989) propagated the calculation of estimates of ordinary least squares regression parameters as the most accurate method of examining length-weight relationships for fish populations. However, since regression parameters are commonly heterogeneous and slope and intercept are often inversely related, valid interpretation of the results is difficult (Bolger and Connolly, 1989). A disadvantage of an alternative, the estimated weights of fish of a particular species and length from regression equations specific to the groups under consideration (De Silva, 1985), is the dependency on the arbitrary choice of the length.

For the community, one possibility would be to use the average condition of a theoretical community of fixed size-structure and species composition over time as an index of body condition. For each individual in this community, the condition is expressed as the weight calculated from the species-specific length-weight relationship per year and the mid-range length of the size-class. Considering that length-weight relationships are only determined annually for a subset of (commercial) species, this theoretical community will consist of a subset of species that are present in the actual community. Another possibility would be to use the full frequency distribution of condition factors (calculated correctly) across a suite of species, and compare the distributions themselves across space or time, or compare their ordinations.

Incidence of disease, pathogens, parasites, contaminants

Considerations relating to the types and incidence of diseases and parasites are similar to those relating to body burdens of contaminants and other measures of body condition. If lower environmental quality affects the biological health of individuals, their resistance to disease and parasites may be lowered. Hence, it is possible that metrics based on the incidence of disease or parasites across a full community could be developed. Such a metric would require data not available to this meeting (and possibly not at all) and hence it was not explored at this meeting.

3.3.3.3   Spatial integrity

No specific metrics were identified for this property (see Section 3.2.4), but the property was scored during the evaluation process.

3.3.4   Results of the evaluation

The resulting scores were discussed and Table 3.3.4.1 represents the consolidated results of this consensus building phase. Metrics were graded on a three-point scale: 2: fully matched to criteria, 1: of some utility against these criteria, and 0: fails to address at least some aspect of these criteria. As in any exercise of this nature, there were some areas of divergent opinion and a number of concerns that are summarised below.

In the tables evaluated, there were 320 cells and complete unanimity of scores was achieved in 30% of the cells (95/320) for fish and 40% (127/320) for benthos. WGECO then proceeded to remove all metrics that had been scored unanimously with a zero for any of the first seven criteria. It had been decided *a priori* that the first seven criteria were to be of equal weight, while the eighth was considered to refer to the information content of the EcoQ rather than a necessary quality.

This first selection left 21 measures in the fish matrix and 14 in the benthos matrix. This was still considered to be too many to be of use operationally and a second sifting was applied. We now removed all metrics having a *modal* score of zero for any of the first seven criteria. This restricted the list for fish to seven metrics, although three cover one property (size structure) (Table 3.3.4.1). For the benthos only one measure, presence of sensitive/charismatic/indicator taxa, remained (Table 3.3.4.1). WGECO then proceeded to consider if these strict criteria had excluded any metrics that tracked crucial properties and almost met the selection criteria (Section 6) and to develop recommendations where key ecological qualities had no metric (Section 3.3.6).

3.3.5   Metrics not considered further

In this section, we identify the principal reasons why various metrics were not considered further (i.e., the criteria they failed to meet).

Biomass: Total sample biomass did not meet the criteria as it was generally regarded as being insensitive to human impacts and subject to high levels of "noise" (natural variation) and for benthos there is a lack of historical data at the appropriate scale.

Size structure: Percentage size composition was the only metric of this group dropped from the fish table. It was considered to have lower sensitivity to human impacts than the other measures and so was dropped. In the benthic table all the metrics failed to meet the selection criteria primarily due to lack of existing data, confounding effects of sampling protocols, and communicability.

Species identities: Indices failing to meet the criteria in this category were generally regarded too insensitive to human impacts and subject to high levels of "noise" (natural variation).

Species diversity: The excluded metrics tended to fail on the criteria of "a high response to the signal from human activity". Many of these metrics are affected by environmental variability. Problems of sample size variability also tend to mask the signal. There was also concern that the "linkage in time" of many of these metrics was poor. Lag-times

were too long, so that the delay between event and response was such that managers may not be able to take remedial action quickly enough. The theoretical linkage between fishing activity and diversity is also poorly understood. How does fishing affect species diversity, and exactly what type of change in activity is required to achieve a particular response?

Life history composition: A variety of life history metrics were considered and most were rejected for fish and all for benthos. The principal reasons were the extent of noise in the data, lack of a tight effect-to-response relationship, and difficulties of having sufficient data for assessing them.

Ecological Functionality: A host of metrics were reviewed for the properties considered relevant to ecological functionality. Productivity was the strongest candidate metric in this group, but all failed to meet the criteria. The main reasons for failure were difficulties in accurately measuring (deriving) the values, a particular concern for those only derivable from models, lack of a strong response to human effects, and a lack of historical values. These issues are addressed further in Section 3.3.6.

Spatial Integrity: WGECO was unable to propose a metric which adequately addressed this issue. There was considerable consensus that this was an important issue and it is considered further in Section 3.3.6.

3.3.6 Gaps

3.3.6.1 Metrics of biological diversity

Much of the reasoning behind the OSPAR EcoQ issues and the development of EcoQOs is driven by the commitments made by most European governments and the EC to the Rio Convention on Biological Diversity. Most biological sampling programmes undertaken in the North Sea invariably record information on species identity and abundance. Therefore, it should be within the power of fisheries scientists and marine ecologists to say rather a lot about species diversity. However, no single diversity index survived the criteria for the selection of metrics on which EcoQOs could be based. This highlights a major failing of the currently available range of diversity measures as operational metrics in the opinion of WGECO.

There is an extensive literature on the subject of species diversity, including theoretical and applied studies (Section 3.3.3.1). These studies have identified a number of shortcomings of the indices that are relevant to their use as management tools and triggers. Diversity indices encapsulate two characteristics of species relative abundance: the number of species and the distribution of individuals among species. Thus when the value of an index changes, it is rarely clear what has happened without further investigation. Species diversity indices vary considerably from year to year, so the signal-to-noise ratio is often low. Most of the metrics in use are sensitive to sample size and to vigilance of observation, weakening further the signal-to-noise ratio. In addition, the relationship between fishing and the species diversity of benthic and fish communities in the North Sea is poorly understood. For example, both positive and negative responses of diversity to fishing have been found (Greenstreet *et al*., 1999; Rogers and Ellis, 2000; Piet, 2001). Therefore, it would not be possible to advise managers of the adjustments to fishing effort that would move a diversity index towards a chosen value.

It would be inappropriate to suggest that any particular species diversity metric would provide an adequate metric of EcoQ in this respect, or therefore provide a sound basis for an EcoQO. Nevertheless, species diversity remains an important characteristic of the communities that make up the North Sea ecosystem, and work should be done to develop metrics of species diversity free from these shortcomings.

3.3.6.2 Metrics of ecological functionality

In their efforts to implement ecosystem-based management in the North Sea (and elsewhere) OSPAR and associated participants have made commitments to conserve ecological functionality as well as biological diversity (Section 3.1.3). WGECO supports this conceptual commitment, but found almost no metrics that could meet reasonable standards for use in management applications at present, nor were any of the ones considered by WGECO thought likely to meet them in the near future. This is a major gap, which requires both some explanation and constructive suggestions for making progress. WGECO identified three aspects of ecological functionality that could be considered separately. For each aspect, the prospects for development of community metrics were different.

1) *The well-being of all the individuals in the community, when viewed collectively*. The community-wide distribution of biological condition has been designated as a promising metric, but no similar metrics were identified for community-wide distributions of body burdens of contaminants or incidence of diseases, parasites, etc. WGECO does view the community-wide level of contaminants, disease, etc., and how concentrations or incidence vary among species and individuals within a community to be an important attribute of ecological quality of the community, particularly when biomagnification and bioaccumulation compound risk or impede rehabilitation. Nonetheless, that does not mean

that there is some community-scale metric of level of contaminant that would be more sensitive to perturbation or more informative to managers than contaminant levels or disease incidence in well-chosen indicator species. ICES has previously provided advice on the selection of indicator species for contaminants (ICES, 1989). The only addition to the past advice on selection of indicator species when one is advising on community-scale indicators of contaminants or disease is the representativeness of the species being used. At the community scale, species which are widespread and highly mobile within the ecosystem of concern should accumulate contaminant and disease burdens more representative of the "community" than a species that is sedentary and patchily distributed, so contaminant levels reflect quite local conditions.

2) *The responses of biological processes to physical forcing.* Great strides have been made in linking physical oceanography to dynamics of marine populations and communities, especially processes like recruitment and growth in fish stocks (Harrison and Parsons, 2000; McKinnell *et al.*, in press; ICES, 2000a). An ecosystem approach should take these linkages into account as fully as possible. Such considerations do not create the need for new community-scale EcoQs and EcoQOs for management, however. In general, the same metrics currently used in single-species fisheries management, for example, can continue to be used. What changes is that the estimation of current states of the population, projections of states in the near- and medium-term future, and possibly even the values of the reference points used in advice can all be improved. Some research on oceanographic forcing of biological systems is indicating that ecosystems may undergo relatively abrupt regime shifts (Francis *et al.*, 1998; Reid *et al.*, 2001), which could affect properties like productivity and resilience of the full system. It is not yet known how to accommodate fully regime shifts in single-species reference-point-based advice and management. However, there is no reason to expect that the setting of some EcoQ and fixed EcoQO for a community property will be an effective strategy for bringing regime shifts into ecosystem management. Such a strategy has the risk of making management less responsive to oceanographic regimes, if they are important, rather than more responsive, by giving special status to some historic configuration of the ecosystem, instead of considering the ecosystem quality objective best for each regime.

4) *Tropho-dynamic processes.* These are intrinsically dynamic relationships among organisms, species and their environments and habitats. This stands in contrast to biological diversity, which is more of a structural property and, although dynamic over time, has a meaning when considered statically at a moment in time (or a sampling interval). Given that tropho-dynamic relationships only have meaning dynamically, they are less tractable to direct monitoring, and tropho-dynamic models are virtually essential in calculating values of metrics. Tropho-dynamic modelling has been an active science field for some years (reviewed in Hollowed *et al.*, 2000), and WGECO has been following the field closely (ICES, 2000a). There is no shortage of tropho-dynamic models, and for over a decade ICES has been using multispecies models of predator-prey interactions (ICES, 1996b) as a contribution to the basis for scientific advice on fisheries. However, ICES has intentionally used these multispecies models to improve estimates of specific parameters of assessment models, with advice continuing to be based on single-species properties that are again estimated better.

With regard to integrated properties of the multispecies models, ICES has viewed results as matters of research interest and tools for framing ecological hypotheses (ICES, 1988, 1990), but not as suitable bases for management advice. When considering tropho-dynamic models of even greater portions of ecosystems, WGECO sees no reason to change its past conclusion (ICES, 1998, 2000a) that none are presently suitable to use as the basis for management advice. Various tropho-dynamic models can produce many outputs that may appeal as bases for advice, but the appeal is deceptive. Tropho-dynamic ecosystem models are still research tools at best. They have not been tested with the rigour routinely applied to models that are used by ICES in formulating management advice, nor do the data used in parameterization withstand the review given to data accepted for analyses by most ICES Working Groups.

Many things have to improve before ecosystem tropho-dynamic models should be viewed as suitable sources for advice on specific management problems, for use in setting EcoQOs. Databases of feeding relationships of marine predators have to cover many more species in the ecosystem, and must be updated on time scales at least matching the time scale on which advice is required regarding properties derived from the tropho-dynamic relationships. Data on energetic requirements of predators and energetic values of prey, and how they vary in space, time, with size, and with abundance of other species are often even weaker than the diet data, and require even more augmentation. Better data for parameterization will not be sufficient for tropho-dynamic models to be suitable for use in advisory contexts, however. The models themselves have to be improved through addition of important processes, such as environmental forcing of system dynamics (see above) and food-dependent life history dynamics (Pimm and Rice, 1987; Rice, 1995), and effective treatment of uncertainty about data and formulations of relationships. More importantly, the models have to undergo a level of testing and validation with a much greater rigour than has been customary when the models are used in exploratory modes. The workshop on testing ecosystem models that was recommended last year (ICES, 2000a) is an important step in the right direction. However, based on the results of the Planning Group meeting (ICES, 2001c), it appears that the existing ecosystem models are still far from being amenable to the type of testing necessary for their use as a basis for management advice, let alone being ready to pass such tests.

With a continuing pessimistic view of the value of ecosystem models to improve management advice directly, particularly in terms of providing currency for effective and reliable EcoQOs, WGECO may be becoming perceived as de-emphasising the importance of tropho-dynamics in understanding ecosystem processes, and in making management of marine ecosystems truly effective. Rather, the opposite is the case. WGECO considers these relationships very important for conserving ecosystem functionality, and certainly sufficiently important that models of the relationships need to be tested as rigorously as models used for the comparatively much simpler problems of tracking, forecasting, providing information about, and supporting scientific advice on single-species dynamics. These are new challenges to ecosystem modellers, but challenges they must rise to meet, if tropho-dynamic aspects of ecosystem functionality are to be convertible into EcoQOs.

It will take time to meet these challenges, and for the interim, it may be more effective to look at much simpler attributes as candidates to be surrogate metrics of tropho-dynamic aspects of ecosystem functionality. WGECO noted that things as simple as the mean and distribution of mouth-gape sizes of predators might be informative about tropho-dynamics at the community scale. This trait, and other similar traits, should be explored in the context of a possible metric for use in ecosystem management, while the longer-term work on raising both tropho-dynamic ecosystem models and their testing approaches to another plane of rigour and reliability, is pursued.

3.3.6.3   Metrics of spatial integrity

Several statistical measures of spatial pattern exist (e.g., Ripley, 1988) and there are many measures used by researchers in studies of spatial structure of populations and meta-population dynamics (Cooper and Mangel, 1999; Caroll and Lamberson, 1999; Policansky and Magnussen, 1998). Between these two sources, there would be no shortage of metrics that address in some way ecological issues of spatial structure and/or function. This does not mean that the possible metrics are good candidate metrics for **community**-scale measures of **spatial integrity. First**, many of them have only been used in single-species applications, and even their computation at the scale of a community may not be straightforward, or possible at all. Where it turns out to be possible to compute the metrics of spatial pattern or meta-population relationships at the community scale, the ecological interpretability of the results remains to be established. **Second,** to the knowledge of the Working Group, the usefulness of most of the spatial metrics has not been tested and demonstrated to be effective in management contexts. This does not mean that we believe that they are not useful when advising managers, simply that their hit, miss, and false alarm rates in management applications are largely unknown. Nor for most or all metrics will their linkage to management actions and their time sensitivities to perturbations be known. **Third**, even if there are metrics of spatial pattern or meta-population relationships that are computable and applicable in management contexts, it is far from clear how to know the degree to which the metric(s) reflect the fairly abstract property of **spatial integrity.**

Is there a reason to be concerned about the absence of community-scale metrics of spatial integrity? Spatial pattern, particularly habitat fragmentation, is a dominant concern in the management of many terrestrial and coastal ecosystems (Eggleston *et al.*, 1999; Olsen, 1999). In marine ecosystems, it should be much less of a concern, because larval distribution processes for many marine fish and invertebrates spread eggs and larvae very widely in the ecosystem. We stress that this is not an absolute exemption from concern, however, because recruitment processes of some important marine plants such as eelgrass may be very local, and some marine invertebrates such as dogwhelks also spread very slowly. Particularly where plants constitute an important part of the marine habitat, spatial integrity may be an important consideration. Also aside from recruitment processes, spatial relationships may be crucial to interactions among predators, prey, and competitors (Rothschild and Osborn, 1988).

Not only are there ecological reasons to conclude that spatial pattern/integrity contributes to ecological quality, there are management issues with intrinsic spatial components. The design of marine protected areas to achieve biological and conservation objectives should be informed by EcoQOs reflecting spatial integrity, if any could be developed. Although WGECO has stressed many times that reducing fishing effort is an essential step to reducing impacts of fishing (ICES, 2000a), for a given amount of fishing effort, changing the spatial pattern of fishing may contribute to changing ecological quality. This would again give value to informative measures of spatial integrity, were any to be found. Finally, a number of coastal zone management issues have an inherently spatial component, and informative metrics of spatial integrity could again be helpful in managing for improved ecological quality.

If there are ecological and management reasons to be interested in metrics and EcoQOs for spatial integrity, what should be done to rectify their present absence? First, the ICES science community must familiarize themselves more fully with the research field and literature on spatial statistics and meta-population dynamics, and increase the participation of experts in that speciality. Advances from the growing field of landscape ecology (Kareiva and Wennergren, 1995; Gray, 1997), to this point pursued largely for terrestrial systems, also need to be brought into marine applications as focused research and not vague platitudes. Knowing more about the ecological information in spatial metrics whose operational management relevance has not been explored, will be only a small step forward. It is critically important that the functional utility of these metrics to support management decision-making also be explored in a focused way. This will require new types of research on these metrics, as discussed below.

Table 3.3.4.1 WGECO group grading of various ecosystem metrics for properties covering key ecological qualities. Metrics were graded on a three-point scale: 2: fully matched to criterion, 1: of some utility against this criterion, and 0: fails to address at least some aspect of this criterion. See the text for a description of the metrics and justification for the criteria: (a) species biodiversity fish communities, (b) species biodiversity benthic communities, (c) ecological functionality in general, and (d) spatial integrity. Where there was unanimity in grading, a single value is presented; otherwise the range of scores is given.

Properties	Possible metrics	Comprehensive and communicable	Sensitive to manageable human activity	Tight linkage in time to that activity	Easily and accurately measured	High response to signal from human activity compared with variation induced by other factors / low miss rate	Measurable in a large proportion of the area to which the EcoQO is to apply	Measured over enough years to provide baseline of information and allow realistic setting of objectives	Representative of relevant aspect of EcoQ. May relate to wider environmental condition
A. SPECIES BIODIVERSITY FISH COMMUNITY									
Biomass									
Size structure	Slope of size spectrum	0–1	1–2	0–1	2	0–2	2	2	1–2
	Length frequency distribution	0–2	1–2	0–2	1–2	0–1	2	1–2	2
	Mean length/weight of all organisms sampled	1–2	1–2	1	2	0–1	2	2	2
Species identity	Presence of indicator, charismatic, sensitive species	1–2	1–2	0–1	1–2	1–2	1–2	1	2
Species diversity	k-dominance curves	1	1–2	0–1	1–2	0–1	1–2	1–2	2
Life history Comp	L_{max} (weighted mean, full distribution)	0–2	1–2	0–1	1–2	1	2	1–2	2

Properties	Possible metrics	Comprehensive and communicable	Sensitive to manageable human activity	Tight linkage in time to that activity	Easily and accurately measured	High response to signal from human activity compared with variation induced by other factors / low miss rate	Measurable in a large proportion of the area to which the EcoQO is to apply	Measured over enough years to provide baseline of information and allow realistic setting of objectives	Representative of relevant aspect of EcoQ. May relate to wider environmental condition
B. SPECIES BIODIVERSITY BENTHOS									
Biomass									
Size structure									
Species identity	Presence of indicator, charismatic, sensitive species	1–2	1–2	0–2	1–2	1–2	1–2	1	2
Species diversity									
Life history Comp									
C. ECOLOGICAL FUNCTIONALITY									
Resilience									
Productivity									
Trophic structure									
Throughput									
Body well-being	Mean and distribution of body burden (contaminants)	1–2	1–2	1–2	1–2	1–2	1–2	0–2	2

D. SPATIAL INTEGRITY

As explained in the accompanying text, no testable candidate indicators for Spatial Integrity could be found by the Working Group.

3.4 Framework considerations

The ICES approach to fisheries advice and the OSPAR approach to ecosystem management differ because OSPAR focuses on one goal, achieving a desired state of the Ecological Quality Objective. The OSPAR approach gives no role to limit and precautionary reference points, which ICES defines relative to undesirable states to be avoided with high probability. The ICES approach includes explicit provisions for uncertainties from several sources, whereas the OSPAR approach, although acknowledging uncertainty and change, does not provide direction for how it should be handled within the EcoQ and EcoQOs. Perhaps most importantly, the OSPAR approach *de facto* asks the scientific community to address political and social objectives, tasks which the ICES approach explicitly reserves for managers and their consultation mechanisms. WGECO expects that there will be problems with implementation of the OSPAR EcoQ and EcoQO framework in future as well, that may be amplified by these differences in approach to scientific advice. Although the OSPAR framework is developed as an overall framework for safeguarding the ecological health of marine ecosystems independently of the human activity threatening the system, many EcoQOs cannot be achieved without substantial cooperation by the fishing industry, and major changes in approaches to fisheries management (see Section 3.1.4.2). To facilitate such cooperation, the advisory approaches in support of the two frameworks should be as similar as possible. In that context, it is of particular concern that the Green Paper on the Common Fisheries Policy, although mentioning ecological quality as a concept, gives no role to EcoQs or EcoQOs in any stage of developing or implementing fisheries management policies and practices.

References

Adams, S.M., and McLean, R.B. 1985. Estimation of largemouth bass, *Micropterus salmoides* Lacepede, growth using the liver somatic index and physiological variables. Journal of Fish Biology, 26: 111–126.

Anon. 1998. Workshop on the Ecosystem Approach to the Management and Protection of the North Sea. Oslo, Norway, 15–17 June 1998. Nordic Council of Ministers, Copenhagen, TemaNord 1998:579. 100 pp.

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

Bolger, T., and Connolly, P.L. 1989. The selection of suitable indices for the measurement and analysis of fish condition. Journal of Fish Biology, 34: 171–182.

Borgmann, U. 1987. Models on the slope of, and biomass flow up, the biomass size spectrum. Canadian Journal of Fisheries and Aquatic Science, 44: 136–140.

Boudreau, P.R., and Dickie, L.E. 1992. Biomass spectra of aquatic ecosystems in relation to fisheries yield. Canadian Journal of Fisheries and Aquatic Science, 49: 1528–1538.

Boudreau, P.R., Dickie, L.E., and Kerr, S.R. 1991. Body size spectra of production and biomass as system-level indicators of ecological dynamics. Journal of Theoretical Biology, 152: 329–340.

Brown, J.H. 1973. Species diversity of seed eating desert rodents in sand dune habitats. Ecology, 54: 775-787.

Caroll, J.E., and Lamberson, R.H. 1999. Sources, sinks and selectivity. Natural Resources Modelling, 12: 1–35.

Chadwick, J.W., and Canton, S.P. 1984. Inadequacy of diversity indices in discerning metal mine drainage effects on a stream invertebrate community. Water, Air and Soil Pollution, 22: 217-223.

Christensen, V., Walters, C.J., and Pauly, D. 2000. Ecopath with Ecosim: a User's Guide, October 2000 Edition. Fisheries Centre, University of British Columbia, Vancouver, Canada, and ICLARM, Penang, Malaysia. 130 pp.

Clarke, K.R. 1990. Comparisons of dominance curves. Journal of Experimental Marine Biology and Ecology, 138: 143-157.

Clarke, K.R., and Ainsworth, M. 1993. A method of linking multivariate community structure to environmental variables. Marine Ecology Progress Series, 92: 205–219.

Cohen, J.E., Briand, F., and Newman, C.M. 1990. Community Food Webs: Data and Theory. Springer-Verlag, Berlin. 308 pp.

Cone, R.S. 1989. The need to reconsider the use of condition indices in fishery science. Trans. Am. Fish. Soc., 118: 510–514.

Connell, J.H. 1978. Diversity in tropical rainforests and coral reefs. Science, 199: 1302–1309.

Cooper, A.B., and Mangel, M. 1999. The dangers of ignoring metapopulation structure for the conservation of salmonids. Fishery Bulletin, 97: 213–226.

Cushing, D. 1995. Population Production and Regulation in the Sea. Cambridge University Press. 354 pp.

Daan, N. 1997. TAC management in North Sea flatfish fisheries. Journal of Sea Research, 37: 321–341.

Daan, N. 2001. A spatial and temporal diversity index taking into account species rarity, with an application to the North Sea fish community. ICES CM 2001/T:04.

Davidson, D.W. 1977. Species diversity and community organization in desert seed-eating ants. Ecology, 58: 711-724.

Death, R.G., and Winterbourn, M.J. 1995. Diversity patterns in stream benthic invertebrate communities: the influence of habitat stability. Ecology, 76: 1446-1460.

de Boer, W.F., Daniels, P., and Essink, K. 2001. Towards ecological quality objectives for North Sea benthic communities. A report to RIKZ, Haren, NL. (contract RKZ 808) from Koeman en bijkerk bv, Haren, Nl, report no. 2001–11. 62 pp.

De Silva, S.S. 1985. Body condition and nutritional ecology of *Oreochromis mossambicus* (Pisces, Cichlidae) populations of man-made lakes in Sri Lanka. Journal of Fish Biology, 27: 621–633.

Eadie, J.McA., and Keast, A. 1984. Resource heterogeneity and fish species diversity in lakes. Canadian Journal of Zoology, 62: 1689-1695.

EC. 2001. Green paper on the future of the Common Fisheries Policy. Brussels, COM (2001) 135.

Eggleston, D.B., Elias, W.E., Etherington, L.L., Dahlgren, P.L., and Posey, M.H. 1999. Organism responses to habitat fragmentation and diversity: Habitat colonization by estuarine macrofauna. Journal of Experimental Marine Biology and Ecology, 236: 107–132.

Eno, N.C., Clark, R.A., and Sanderson, W.G. (Eds.). 1997. Non-native Marine Species in British Waters: a Review and Directory. Joint Nature Conservancy Council, Peterborough, UK.

Fisher, R.A., Corbet, A.S., and Williams, C.B. 1943. The relation between the number of species and the number of individuals in a random sample of an animal population. Journal of Animal Ecology, 12: 42–58.

Francis, R.C., Hare, S.R., Hollowed, A.B., and Wooster, W.S. 1998. Effects of interdecadal climate variability on the oceanic ecosystems of the NE Pacific. Fisheries Oceanography, 7: 1–21.

Gislason, H., and Lassen, H. 1997. On the linear relationship between fishing effort and the slope of the size spectrum. ICES CM 1997/DD:05.

Gray, J.S. 1997. Marine biodiversity: Patterns, threats and conservation needs. Biodiversity and Conservation, 6: 153–175.

Greenstreet, S.P.R., and Hall, S.J. 1996. Fishing and the groundfish assemblage structure in the northwestern North Sea: an analysis of long-term and spatial trends. Journal of Animal Ecology, 65: 577–598.

Greenstreet, S.P.R., Spence, F.E., and McMillan, J.A. 1999. Fishing effects in northeast Atlantic shelf seas: patterns in fishing effort, diversity and community structure. V. Changes in structure of the North Sea groundfish assemblage between 1925 and 1996. Fisheries Research, 40: 153–183.

Hall, S.J., and Greenstreet, S.P.R. 1998. Taxonomic distinctness and diversity measures: responses in marine fish communities. Marine Ecology Progress Series, 166: 227–229.

Hall, S.J., and Raffaelli, D. 1991. Food web patterns: lessons from a species-rich web. Journal of Animal Ecology, 60: 823–842.

Harrison, P.J., and Parsons, T.R. (Eds.). 2000. Fisheries Oceanography: An Integrated Approach to Fisheries Ecology and Management. Blackwell Science, Oxford. 359 pp.

Heip, C., Basford, D., Craeymeersch, J.A., Dewarumez, J.-M., Dorjes, J., de Wilde, P., Duineveld, G., Eleftheriou, A., Herman, P.M.J., Niermann, U., Kingston, P., Kunitzer, A., Rachor, E., Rumohr, H., Soetaert, K., and Soltwedel, T. 1992. Trends in biomass, density and diversity of North Sea macrofauna. *ICES Journal of Marine Science*, 49: 13-22.

Hill, M.O. 1973. Diversity and evenness: a unifying notation and its consequences. Ecology, 54: 427-432.

Hollowed, A.B., Bax, N., Beamish, R., Collie, J., Fogarty, M., Livingston, P., Pope, J., and Rice, J.C. 2000. Are multi-species models an improvement on single-species models for measuring fishing impacts on marine ecosystems? *ICES Journal of Marine Science*, 57: 707–719.

Hurlbert, S.H. 1971. The non-concept of species diversity: a critique and alternative parameters. Ecology, 52: 577-586.

Huston, A.H. 1994. Biological Diversity: The Coexistence of Species on Changing Landscapes. Cambridge University Press. 681 pp.

ICES. 1988. Report of the Multispecies Assessment Working Group. ICES CM 1988/Assess:23.

ICES. 1990. Report of the Special Meeting of the Multispecies Assessment Working Group. ICES CM 1990/Assess:28.

ICES. 1995. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 1995/Assess/Env:1.

ICES. 1996a. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 1996/Assess/Env:1.

ICES. 1996b. Report of the Multispecies Assessment Working Group. ICES CM 1996/Assess:3.

ICES. 1998. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 1998/ACFM/ACME:01.

ICES. 2000a. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 2000/ACME:02.

ICES. 2000b. Report of the Working Group on Cod and Climate Change. ICES CM 2000/C:11.

ICES. 2000c. Report of the ICES Advisory Committee on the Marine Environment, 2000. ICES Cooperative Research Report, 241: 150–152.

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

ICES. 2001b. Report of the Working Group on Marine Mammal Population Dynamics and Habitat. ICES CM 2001/ACE:01.

ICES. 2001c. Report of the Planning Group for a Workshop on Ecosystem Models. ICES CM 2001/ACME:06.

ICES. 2001d. Report of the ICES Advisory Committee on Fishery Management, 2000. *ICES Cooperative Research Report*, 242: 1–2.

IMM. 1997. Statement of Conclusions from the Intermediate Ministerial Meeting on the Integration of Fisheries and Environment Issues. Bergen, 13–14 March 1997. Norwegian Ministry of the Environment, Oslo.

Jongman, R.H.G., ter Braak, C.J.F., and van Tongeren, O.F.R. 1987. Data analysis in community and landscape ecology. Pudoc-Wageningen, Den Haag. 299 pp.

Kabuta, S., and Enserinck, L. 2000. Development of indicators for the Dutch North Sea project GONZ. ICES CM 2000/ACME:WP7.

Kareiva, P., and Wennergren, U. 1995. Connecting landscape patterns to ecosystem and population processes. Nature, 373: 299–302.

Kempton, R.A., and Taylor, L.R. 1974. Log-series and log-normal parameters as diversity discriminants for the Lepidoptera. Journal of Animal Ecology, 43: 381-399.

Lambshead, P.J.D., Platt, H.M., and Shaw, K.M. 1983. The detection of differences among assemblages of marine benthic species based on the assessment of dominance and diversity. Journal of Natural History, 17: 859-874.

Lanters, R.L.P., Skjoldal, H.R., and Noji, T.T. 1999. Ecological Quality Objectives for the North Sea. Basic document for the Workshop on Ecological Quality Objectives for the North Sea, 1–3 September 1999, Scheveningen, The Netherlands.

Lindeboom, H., and de Groot, S.J. (Eds.). 1997. The effects of different types of fisheries on the North Sea and Irish Sea Benthic Ecosystems. European Commission, Brussels, IMPACT II Final Report. Contract Report AIR2-CT94-1664. 396 pp.

MacArthur, R.H., and MacArther, J.W. 1961. On bird species diversity. Ecology, 42: 594-598.

Magurran, A.E. 1988. Ecological Diversity and its Measurement. Chapman and Hall, London.

May, R.M. 1975. Patterns of species abundance and diversity. *In* Ecology and Evolution of Communities, pp. 81-120. Ed. by M.L. Cody and J.M. Diamond. Belknap Press, Harvard.

May, R.M. 1976. Patterns in multi-species communities. Theoretical Ecology; Principles and Applications. Blackwell, Oxford.

McKinnell, S.M., Brodeur, R.D., Hanawa, K., Hollowed, A.B., Polovina, J.J., and Zhang, C.-I. (Eds.). In press. Pacific climate variability and marine ecosystem impacts from the tropics to the Arctic. Progress in Oceanography.

McRae, G., Camp, D.K., Lyons, W.G., and Dix, T.L. 1998. Relating benthic infauna to environmental variables in estuaries using non-metric multidimensional scaling and similarity indices. Environmental Monitoring and Assessment, 51: 233–246.

Olsen, J.L. 1999. Earth is a marine habitat and marine biodiversity matters. Third European Marine Science and Technology Conference (MAST conference), Lisbon, 23–27 May 1998: Conference proceedings. European Commission DG XII Science, Research and Development, Luxembourg. Pp. 319–328.

OSPAR. 2000a. Report of the OSPAR Environmental Assessment and Monitoring Committee, Hamburg, 27–31 March 2000. OSPAR Commission, London.

OSPAR. 2000b. Quality Status Report 2000, Region II – Greater North Sea. OSPAR Commission, London.

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

Peet, A.K. 1974. The measurement of species diversity. Annual Review of Ecology and Systematics, 5: 285-307.

Piet, G.J. 2001. Development of ecological quality objectives for the North Sea fish community. Working Paper at the 2001 WGECO meeting.

Pimm, S.L. 1982. Food Webs. Chapman and Hall, New York.

Pimm, S.L., and Rice, J.C. 1987. The dynamics of multi-species, multi-life stage models of aquatic food webs. Theoretical Population Biology, 32: 303–325.

Policansky, D., and Magnussen, J.J. 1998. Genetics, metapopulations, and ecosystem management of fisheries. Ecological Applications, 8: S119–S123.

Preston, F.W. 1962. The canonical distribution of commonness and rarity. Ecology, 43: 185–215.

Reid, P.C., Borges, M.F., and Svensen, 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.

Rice, J.C. 1995. Food web theory, marine food webs, and what climate change may do to northern marine fish populations. *In* Climate Change and Northern Fish Populations, pp. 561–568. Ed. by R.J. Beamish. Can. Spec. Publ. Fish. Aquat. Sci. 121.

Rice, J.C. 2000. Evaluating fishery impacts using metrics of community structure. *ICES Journal of Marine Science*, 57: 682–688.

Rice, J.C., and Gislason, H. 1996. Patterns of change in the size spectra of numbers and diversity of the North Sea fish assemblage, as reflected in surveys and models. Proceedings of the Symposium: Changes in the North Sea ecosystem and their causes. Åarhus 1975 revisited. *ICES Journal of Marine Science*, 53: 1214–1225.

Ripley, B.D. 1988. Statistical inference for spatial processes. Cambridge University Press, Cambridge, UK. 148 pp.

Robinson, J.V., and Sandgren, C.D. 1984. An experimental evaluation of diversity indices as environmental discriminators. Hydrobiologia, 108: 187-196.

Rogers, S.I, Clarke, K.R., and Reynolds, J.D. 1999. The taxonomic distinctness of coastal bottom-dwelling fish communities of the North-east Atlantic. Journal of Animal Ecology, 68: 769–782.

Rogers, S.I., and Ellis, J.R. 2000. Changes in the demersal fish assemblages of British coastal waters during the 20th century. *ICES Journal of Marine Science*, 57: 881–881.

Rosenzweig, M.L. 1995. Species Diversity in Time and Space. Cambridge University Press, UK. 436 pp.

Rothschild, B.J., and Osborn, T.R. 1988. Small-scale turbulence and plankton contact rates. Journal of Plankton Research, 10: 465–474.

Serchuk, F., Griffith, D., Hoydal, K., Jakobsen, T., Kirkegaard, E., Maguire, J. J., Ulltang, O., and Vaske, B. 2000. The ICES Advisory Committee on Fishery Management (ACFM), 1978–2000: a quarter century agrowing. *In:*

Symposium on 100 Years of Science under ICES, 1–4 August 2000, Helsinki, Finland. *ICES Marine Science ymposia, Vol. 215* (2002).

Sheldon, R.W., Prakash, A. and Sutcliffe Jr, W.H. 1972. The size distribution of particles in the ocean. Limnol. and Oceanograph. 17: 327–339.

Skjoldal, H.R. 1999. Overview report on Ecological Quality (EcoQ) and Ecological Quality Objectives (EcoQOs). Institute of Marine Research, Bergen, Norway.

Soetaert, K., and Heip, C. 1990. Sample-size dependence of diversity indices and the determination of sufficient sample size in a high-diversity deep-sea environment. Marine Ecology Progress Series, 59: 305-307.

Southwood, T.R.E. 1978. Ecological Methods: with Particular Reference to the Study of Insect Populations. 2nd Edition, Chapman and Hall, London.

Sprules, W.G., and Goyke, A.P. 1994. Size-based structure and production in the pelagia of Lakes Ontario and Michigan. Canadian Journal of Fisheries and Aquatic Science, 51: 2603–2611.

Steele, J.H. 1998. From carbon flux to regime shift. Fisheries Oceanography, 7: 176–181.

Symes, D., and Pope, J.G. 2000. An ecosystem-based approach to the Common Fisheries Policy: Achieving the objectives. English Nature, Peterborough, UK. 52 pp.

Thiebaux, M.L., and Dickie, L.M. 1992. Models of aquatic biomass size spectra and the common structure of their solutions. Journal of Theoretical Biology, 159: 147–161.

Thiebaux, M.L., and Dickie, L.M. 1993. Structure of the biomass in aquatic ecosystems: a consequence of allometry in predator-prey interactions. Canadian Journal of Fisheries and Aquatic Science, 50: 1308–1317.

Thingstad, T.F. 1998. A theoretical approach to structuring mechanisms in the pelagic food web. Hydrobiologia, 363: 59–72.

Ulanowicz, R. 1997. Ecology: the Ascendant Perspective. Columbia University Press, New York.

van Beek, F.A., and Pastoors, M. 1999. Evaluating ICES catch forecasts: the relationships between implied and realised fishing mortalities. ICES CM 1999/R:04.

Warwick, R.M., and Clarke, K.R. 1995. New 'biodiversity' measures reveal a decrease in taxonomic distinctness with increasing stress. Marine Ecology Progress Series, 129: 301–305.

Warwick, R.M., and Clarke, K.R. 1998. Taxonomic distinctness and environmental assessment. Journal of Applied Ecology, 35: 532–543.

Washington, H.G. 1984. Diversity, biotic and similarity indices: a review with special relevance to aquatic ecosystems. Water Research, 18: 653-694.

4   Seabirds and Marine Mammals in an ECOQO-framework

WGECO addressed EcoQOs for marine mammals and seabirds at a larger scale than the individual species levels that the Working Group on Seabird Ecology (WFSE) and the Working Group on Marine Mammal Populations and Habitats (WGMMPH) considered (ICES, 2001d). Specifically, WGECO:

1) compared the framework developed for EcoQs and EcoQOs (Section 3) with the approach taken by WGMMPH and WGSE;
2) commented on the species metrics identified by WGSE and WGMMPH with regard to either their efficiency in detecting impacts or protecting the integrity of the community/ecosystem.

4.1   The Approaches taken by WGSE and WGMMPH

The approaches taken by both Working Groups are comparable (Table 4.1.1). In both cases the development of EcoQ starts with detecting general issues that are of concern for either seabirds or marine mammals.

Based on the OSPAR JAMP list, WGSE considered all possible classes of human activities that could affect seabird populations. This selection resulted in ten categories. For each category, potential EcoQ metrics were considered. WGSE used nine criteria to screen potentially suitable EcoQ metrics. Their criteria match closely with the ones used by WGECO (see Section 3.3.2). For each of the final EcoQ metrics selected, WGSE tried to identify a reference level (often "pristine" levels), described the current status, and identified a target level for the EcoQ metric, if possible (Table 4.1.1). The target level chosen was that which WGSE considered achievable by current management, based on available evidence. WGECO observes that this differs from its interpretation of OSPAR target level (EcoQO) and the target levels proposed by WGSE could be regarded as "manageable levels" (which would form another category into the lower-most box of the EcoQO framework in Figure 3.1.1).

WGMMPH took a similar approach to that of WGSE, but focused more on metrics that described marine mammal populations rather than searching for metrics as descriptors of the state of the wider environment. After selecting and reviewing potential EcoQ metrics, six were selected for further development. WGMMPH tried to identify reference levels, their current status *sensu* OSPAR, and target levels for the selected EcoQ metrics. Just for one EcoQ, the population size of bottlenose dolphins in the NW North Sea, a target *sensu* OSPAR was defined.

Both Working Groups interpreted "reference level" in most cases as the pristine state or the state where human impact is minimal, but for a few EcoQs other reference levels were used. Reference levels were suggested for most EcoQs. There was insufficient information on cetaceans to allow estimates of total population numbers to be used for EcoQOs (CVs too high). Monitoring data on seabird populations seems to be sufficient. The target levels set for the EcoQOs differ in nature within the sets of both groups. In ICES terminology, Limit Reference Points are suggested for several marine mammal and one seabird EcoQO.

Both groups defined single-species metrics at the population scale and applied them to as many species as possible. This approach increases the actual number of EcoQs to be further developed and ultimately used in management decisions (especially WGSE referred to quite a long list of bird species). However, these EcoQOs differ from the OSPAR framework in that both groups suggest that they should be used as triggers for further research on the causes of change, rather than as triggers for direct management action. In fact these EcoQOs do not really reflect management objectives or reference points but benchmarks for triggering further research.

Both groups recognised that EcoQOs have ultimately to be set by society through the political process. They respond to their terms of reference by interpreting the request to formulate provisional target levels by suggesting "manageable" levels or limit levels. The variety of levels that can be set on the EcoQ metric is potentially confusing; WGECO therefore advises that advice on EcoQs and levels needs to be carefully and precisely worded.

Limit Reference Points (LRP) may often be easier to develop from a scientific point of view. Some of them come directly from legislation and have a legal basis. Others may be developed from the dynamics of the populations concerned. The levels of LRPs used by both groups are based on international standards (IUCN standards used by the mammal group, the widely used BirdLife International standards for the bird group). It may be wise to choose one approach rather than use two standards. If two standards are to be used, this choice would need to be justified. Moreover, there is substantial debate within the marine science community regarding the appropriateness of the IUCN standards for marine species.

4.2 Evaluation of the Preliminary Results of WGSE and WGMMPH

From this overview, it is clear that all selected EcoQs refer to single-species metrics only (Table 4.1.1). This is an important observation because both groups did not rule out the possibility of developing community-based metrics. WGECO considered this issue and could not suggest alternative community or ecosystem scale properties that would be of any greater help in the management of human activities in the marine environment with reference to marine mammals and seabird populations than those suggested for single-species.

The WGMMPH report failed to report many EcoQ. WGECO was therefore unable to assess whether sufficient and reliable data are available to describe the current status of the EcoQs they proposed.

The match of some EcoQ metrics with the themes they covered raised some questions. The report of WGSE suggests the use of "breeding productivity of kittiwakes as an index for sandeel stocks in the North Sea". This would be a useful indicator within the foraging area of the kittiwakes, but not necessarily at the North Sea scale as the WGSE title suggests. Since the direct assessment of sandeel stocks is very difficult, it would not be straightforward to evaluate independently the accuracy or precision of seabird breeding productivity as an index for sandeel stocks at various spatial scales. Nevertheless, this EcoQ would be usable as a metric of availability of sandeels to predators, and recent decisions of fisheries managers in the EU are consistent with the information contained in this metric.

Table 4.1.1 Preliminary results of the Working Group on Seabird Ecology and the Working Group on Marine Mammal Population Dynamics and Habitats on the development of EcoQs and EcoQOs. Column headings are taken directly from both Working Group reports, although their use of the terminology may differ from the ones used within OSPAR or ICES (see Section 3.1.2).

Theme	Category	EcoQ/EcoQ metric	Current level	EcoQO Reference level	Target level
Pollution	Oil contaminants	Proportion of oiled guillemots among those found dead or dying on the beach	12–85%	0%	10%
	Mercury	Mercury concentrations in eggs of selected seabird species	Various	no	no
		Mercury concentrations in body feathers of selected seabird species	Various	Possibly for situation in 1900	Suggested reference level
	Organochlorines	Organochlorine concentrations in seabird eggs	Various	zero	zero
Eutrophication					
Litter	Plastic particles	Number of plastic particles in gizzards of North Sea fulmars	Various, not well-known	0%	10 particles within any fulmar of a sample of 40
Fisheries	Bycatch				
	Harvesting food and predators	Index of breeding productivity of black-legged kittiwake as index for sandeel stocks	0.97	not known	LRP=0.5
	Increase in food supply				
	Mariculture				
	Habitats and ecosystem health	Seabird population trends as an index of seabird community health	Various	not known	LRP more than 20% decrease within 20 years
Threatened and declining					
Hunting/harvesting					
Disturbance					
Introduced/conflicting species					
Climate change					
Community health	Harbour/grey seal	Population size	Increasing	0% increase	More than 10% decrease within 10 years
	Bottlenose dolphin	Population size in NW North Sea		Stable at a higher level than currently	>2% increase per annum over at least 10 years
	Harbour/grey seal	Abandonment of breeding sites	Needs research	zero	Loss of more than 10% of breeding sites within 10 years
	Harbour/grey seal	Number of births	Needs research	Current level	More than 10% decrease within 10 years
	Harbour porpoise and other small cetaceans	No appropriate EcoQ selected	Needs research		
Contaminants	Seals	Concentrations of PCB, DDT, OC in body fat	Available	zero	Limit Reference Points are given
Fisheries	Bycatch of harbour porpoise	Percentage of population killed (incidental bycatch)	Available	zero	<1.70%
	Bycatch of seals	Percentage of population killed (incidental bycatch)	Available	zero	<1%

5 ECOQOS for fish and benthic communities and threatened and declining species

5.1 Introduction

The evaluation process undertaken in Section 3.3 provided a meaningful short-list of metrics that were considered the most appropriate descriptors of EcoQ of fish and benthic communities. Not only were they relevant to life history characteristics of species and their ecological functionality, but they also fulfilled a range of other criteria related to their implementability as EcoQOs. Table 5.1.1 provides the final list of metrics identified.

Table 5.1.1 Metrics selected by the process described in Section 3.3, scoring well on the seven selection criteria applied.

> **Fish communities**
>   Length frequency (percentage composition by size class; slope of size spectrum)
>   Mean length/weight of fish within specified limits
>   Presence of indicator/charismatic/sensitive species
>   Species abundance (k-dominance curves; species composition)
>   Maximum length (weighted mean L_{max} of community)
>   Mean and distribution of "body condition"
> **Benthos communities**
>   Presence of indicator/charismatic species

The ten environmental issues identified by OSPAR as requiring EcoQOs are listed below:

1) Reference points for commercial fish species
2) Threatened and declining species
3) Sea mammals
4) Seabirds
5) Fish communities
6) Benthic communities
7) Plankton communities
8) Habitats
9) Nutrient budgets and production
10) Oxygen consumption

Issues 1 to 6, and 8 are considered to be directly influenced by fishing effects. WGECO included fish communities, benthic communities, and threatened and declining species in the discussion of this topic. The following sections review recent progress to provide further guidance on EcoQOs for these three environmental issues. Suggested metrics of EcoQ are then reviewed in the light of the framework suggested in Section 3.3. The final section describes a proposed approach to EcoQOs.

5.2 EcoQOs for North Sea Fish Communities

5.2.1 Introduction

Aspects of fish community structure may be described by a large number of metrics, and the effect of fishing activity on some of these metrics is currently under study. There is an extensive literature describing North Sea fish communities, and some large spatial and time-series databases. A pragmatic approach is to use these as the basis for evaluating fish communities, while recognising that this is only the most convenient method given the time available, and not necessarily the best. Initial discussion of this topic was greatly helped by a draft Netherlands working paper on potential EcoQOs for fish communities (Piet, 2001), which will be summarised in this section.

5.2.2 Summary of Preparatory Work (information from Piet 2001)

The study identified the following set of properties that covered both the structure and the functionality of the fish community (the list is similar to the one in Section 3):

Structure

- Biomass
- Size structure
- Species composition
- Species diversity
- Composition based on traits (e.g., life history, habitat preference, etc.).

Functionality

- Trophic structure
- Body well-being

For the development of EcoQ metrics, Piet (2001) used a pragmatic, largely data-driven approach. First, a suite of properties of ecological quality were established, and then the most suitable metrics were chosen based on a number of criteria:

- Representativity of a relevant aspect of the ecological quality;
- Quantifiability of the metric;
- Data availability through existing time-series or historic data of a sufficiently large proportion of the area to which the metric is to apply;
- Causality (partitioning among effects of human activity, other forcing factors and inherent variability);
- Comprehensibility and communicativity (also to non-scientists, e.g., policymakers);
- Sensitivity for detecting gradual change (the signal should not be concealed by noise; possibility to determine a meaningful trend or variations in an objective manner.

Data from several North Sea surveys were available that differed in gears used and the area or time period covered. However, for calculating potential metrics, first quarter IBTS data were used (1974–1999).

5.2.2.1 Biomass

According to Piet (2001), the total biomass of the fish community present in the North Sea may depend on several factors such as the availability of food, water temperature, or fishing effort. The total fish catch in weight per haul in a survey was suggested as a metric. The total catch per haul was calculated from the numbers caught per species and an appropriate length-weight relationship. This implies that the metric will reflect changes over time in species composition and size structure, but not in condition (Table 5.2.3.1). The total catch per haul for the whole North Sea showed considerable variation, with a relatively low biomass in the late 1970s/early 1980s followed by an increase towards a relatively high level in the 1990s. The time-series of total biomass in different roundfish areas showed that within the North Sea there was considerable spatial variability.

According to Piet (2001), the functioning of a community with high biomass is not necessarily better than one with relatively low biomass, and likewise a pristine fish community does not necessarily have a higher or lower biomass than one that is impacted by human activities such as fisheries, eutrophication, etc. The high spatial and temporal variability provided no clue as to what the total biomass of a fish community in "optimal" or "pristine" state would be like. Hence, although there is no scientific basis for setting an EcoQO, they consider biomass an important metric of the functioning of the fish community and as such would certainly be worth monitoring, but would regard long-term trends with caution.

5.2.2.2 Size-structure

The study identified three metrics to describe the size-structure of the fish community:

Slope of the biomass size spectra;
Number or biomass in a specific size-class;
Average size or weight.

All approaches to assess the change in size-structure of the North Sea fish community over time revealed the same pattern, which was a decline in abundance of large fish over time. This trend has been confirmed by studies that

analysed change in the size distributions of roundfish and flatfish species using historic catch data (Rijnsdorp *et al.*, 1996; Rogers and Ellis, 2000), which found that the relative contribution of the larger fish has decreased since the early years of the 20th century.

Considering that all metrics reveal the same trend, Piet (2001) felt that the choice of the most appropriate metric for the size-structure of the fish community could be based on other criteria. Both the slope of the biomass size spectra and the average weight of an individual fish showed significant trends over time, with similar variation around the trend. They based the choice of a reference level on the time-series available. Here the linear fit indicated a reference level of 230 grams average individual fish weight per individual in the early 1970s (Table 5.2.3.1). Although the average weight in a pristine environment should be higher, the authors felt that it was not possible based on available data and knowledge to come up with a reasonable estimate. Moreover, considering the measures necessary to at least reverse the current downward trend and realise a modest increase in average weight, it is hardly realistic to aim at this point for levels higher than this reference level.

5.2.2.3   Species diversity

In the study, three diversity indices were calculated per year for the North Sea fish community: Hill's N0, N1, and N2. All indices showed an increase over time. Hill's N0 showed a sudden step-wise increase in the late 1980s, and the other two indices showed a more gradual increase. The authors explain the difference as resulting from the increase in sampling effort in the late 1980s, because Hill's N0 as a metric of the total number of species is highly dependent on sampling effort. Moreover, interpreting trends in species richness may be flawed because inconsistencies in reported species by different countries participating in the IBTS indicate that species identification has been unreliable (Daan, 2001).

Hill's N1 and N2 are mainly dependent on the numbers of abundant and very abundant species, respectively. The explanation given for the increasing trend in these indices was that fishing mainly targets some of the most abundant species and that the additional mortality has resulted in an increased evenness and hence a higher index. Comparison with historic catches showed that in the early 1900s the fish community was slightly more diverse (Rijnsdorp *et al.*, 1996; Greenstreet and Hall, 1996), with Hill's N1 being markedly higher in the past and Hill's N2 within the range observed for present-day catches.

It was not always clear what changes in the fish community caused the observed changes in diversity, how this change affected the stability or productivity of the community, and to what extent it was induced by anthropogenic activities. Thus, the authors had difficulties in suggesting EcoQOs based on specific biodiversity indices.

5.2.2.4   Species composition based on traits

Piet (2001) considered the description of the fish community in terms of its biological traits an important ecological quality. The functional groups chosen were based on species characteristics pertaining to life history, habitat preference and biogeographic region. Habitat preference was captured in an index based on a distinction of two groups: (1) demersal (i.e., benthic and demersal), and (2) pelagic (i.e., pelagic, semipelagic, epipelagic, mesopelagic, and bathypelagic). The index was calculated as biomass pelagic/biomass demersal. The index showed considerable variation over time, with high values at the start (late 1970s) and end of the sampling period (late 1990s) and lower values in between.

The biogeographic origin of a species was used to distinguish southern (i.e., Lusitanian, Atlantic tropical, Mauretanian) and northern (i.e., Arctic, Arctic/Boreal, Atlantic polar, Atlantic temperate, Boreal, Boreal/Arctic) species. The authors developed an index by dividing the biomass of "southern" species by the biomass of "northern" species. This index showed a trend towards a community with a high proportion of southern species. Although linear regression on all years showed that the trend was not significant (p=0.16), elimination of the 1991 outlier rendered a significant (p=0.03) trend.

These indices showed that the composition of the fish community in terms of functional groups changes over time and in some cases displayed trends. Piet (2001) concluded that the index based on the ratio of southerly/northerly species was the most sensitive to water temperature and represented a metric of the effect of water temperature on species composition of the fish community (Table 5.2.3.1). However, the contribution of human activities to the temperature changes and to the metric was not fully understood. Although biological traits are important ecological qualities of fish populations, the setting of EcoQOs, and the identification of reference levels, was not straightforward.

5.2.2.5   Trophic structure

Current theories of food-web structure and community regulation are based on a model in which species are described as homogeneous units, while the dynamic interactions among them form a network of consumer-resource relations. For most fish species, diets shift during early development and this complicates trophic interactions because a species may

feed at different trophic levels during its ontogeny. Therefore information on both species and size is relevant when studying the trophic structure of the fish community.

Piet (2001) considered that the trophic structure of the fish community might be measured as the average trophic level of the fish community. Pauly *et al.* (1998) suggested that overfishing of stocks at a higher trophic level (i.e., piscivores) may result in refocusing of fishing effort on planktivores and lead to a corresponding decline in the average trophic level of the landings. "Fishing down the food chain" may significantly disrupt the food web and models suggest that it may have cascading implications for the stability of stocks and ecosystems (Christensen, 1996).

However, the authors conclude that quantification of the trophic level of each species- and size-specific trophic group is not straightforward. For many species the necessary information was lacking, or not even relevant given the shifts in trophic niche of many fish populations. Moreover, the determination of the trophic level of the fish community requires the intervention of some form of ecological model, representing hypotheses about the trophic interactions among species- and size-specific groups in the model. The question of how well the metric reflects the properties of the fish community cannot be dissociated from the question of how well the model represents the ecosystem (Rice, 2000).

5.2.3 Summary

The data in Table 5.2.3.1 summarise the properties and metrics that Piet (2001) considered relevant to EcoQs of the North Sea fish community. The table also presents reference levels for two of these metrics.

Table 5.2.3.1   Set of metrics to monitor the ecological quality of North Sea fish communities with current values and reference values. All values are based on the first quarter IBTS.

Metric	Value of the metric	
	Present	Reference
Average weight of individual fish (g)	60	230
Hill's N0	9.5	
Hill's N1	2.6	
Hill's N2	2.0	
Average maximum length (cm)	38	42
South/North ratio (x100)	2.5	
Pelagic/Demersal ratio (x100)	63	
Total biomass (kg/haul)	276	

The availability of survey data for the North Sea has obviously enabled a wide range of potential quality objectives to be evaluated. Most refer to community metrics involving the size structure of the populations and offer hope that meaningful objectives and reference levels can be reached. The way in which these ideas link to the WGECO framework is evaluated in Section 5.5.

5.3   EcoQOs for North Sea Benthic Communities

5.3.1   Introduction

Although much of the North Sea benthic environment is sedimentary in nature, a full consideration of EcoQOs for benthic communities must recognise that there are diverse habitats within this broad classification. Thus, a complete discussion of benthic EcoQOs in the North Sea must include habitats associated with the shallow and dynamic coastal waters, intertidal flats, offshore coarse environments and rocky reefs, and deepwater sediments such as those in the Norwegian Trough. Furthermore, a complete review of benthic communities must consider invertebrate infauna at a range of scales, as well as the macro-epibenthos and mega-epibenthos, which should include benthic and demersal fish populations. We provide here a summary of the study by de Boer *et al.* (2001), which provides a first attempt to develop EcoQOs for benthos, and thus serves as a useful starting point. Nonetheless, de Boer *et al.* (2001) develops another independent framework which must be modified further to correspond directly with the EcoQO framework.

5.3.2   Summary of preparatory work (de Boer *et al.* 2001) with comments

The study starts off with considering the benthos of the North Sea and the various schemes to classify the benthos. The authors settle on the scheme devised by Kunitzer *et al.* (1992), which identifies eight benthic community types. They acknowledge that this scheme ignores a large number of potentially important ecological habitats including kelp forests, sub-tidal rocky reefs, inshore and estuarine sediments, and intertidal areas. The classification also ignores epibenthos and is not consistent with either the MNCR Marine Biotope Classification (Connor *et al.*, 1997) or the EUNIS scheme.

The report then addresses which human pressures are most likely to influence benthic communities and proposes six measures that can be used to assess the status of benthic communities. Three measures (i to iii) address issues relating to species diversity and three (iv to vi) try to characterise community structure and function. The measures are: (i) species diversity as shown by Shannon-Weiner; (ii) abundance of fragile, vulnerable species; (iii) incidence of scar damage in *Arctica islandica* shells; (iv) the ratio of r- to K-strategists as shown by the W statistic derived from ABC curves; (v) abundance of opportunistic species; and (vi) the *Vas Deferens* Sequence Index in female *Nucella lapillus*. De Boer *et al*. (2001) noted that not all proposed indicator taxa (ii and v) will be present in all eight communities and suggested that EcoQOs and reference levels should be set differentially for each community type.

The report states that "EcoQO reference levels should represent the situation under minimal human impact" and advocates the use of values derived from the 1986 data series (Kunitzer *et al.*, 1992) as a basis for EcoQOs. In doing so, the authors ignore any of the other time-series data sets that are available and that may provide information on a situation more close to the "minimal human impact". If the assumption is that the eight community types are true reflections of real, distinct systems, then data from any station/study in one of the areas should be representative of the whole and therefore valuable in the consideration of reference levels. The other assumption made is that the 1986 values are lower limits for (i) and (ii) and higher limits for (v). It is by no means clear that human impacts have led to a decrease in diversity across large areas of the North Sea. It is widely recognised that natural disturbance is important for promoting diversity (Connell, 1978; Thrush, 1991; Hall *et al.*, 1994), through co-existence in a spatial mosaic. Anthropogenic disturbances can act in the same way, and a moderately impacted ecosystem can show either an increase or a decrease in diversity. The following metrics were considered:

Species diversity as indexed by Shannon-Weiner

De Boer *et al*. (2001) acknowledge the wide range of indices available that provide measures of "species diversity" and elect to use the Shannon-Weiner (H') function primarily because of its wide usage in the literature and the fact that its limitations are well known. They recognised that measures of diversity are method/sample-effort dependent and will vary between community types. In spite of the well-described non-normality of this metric, the authors proceed to present means, and even standard errors, of H'. While acknowledging that human impacts can increase diversity, the presumption is made that high diversity indicates low levels of anthropogenic effects.
.
Abundance of fragile, vulnerable species

This metric, and the subsequent one, focus on the physical impact of towed benthic fishing gears on the benthos. A number of short-term experimental studies (e.g., Lindeboom and de Groot, 1997) have shown dramatic levels of mortality on certain fragile taxa. The list proposed by the authors includes target species such as *Ostrea*, *Cancer*, *Hommarus*, and *Nephrops*. Many of the species listed are not recorded in the 1986 survey data, as they are epibenthic. This means that no reference levels can be set from this source.

Incidence of scar damage in Arctica islandica *shells*

Arctica is a long-lived species that is recorded from a number of the community types. It has the capacity to recover from some physical impacts, but this leaves a scar on the shell. A relationship is available (Witbaard and Klein, 1997), which could allow this incidence to be used to quantify the level of impact occurring from heavy fishing gears.

Ratio of r- to K-strategists as indexed by the W statistic derived from ABC curves

The underlying assumption of this metric is that an impacted community will have more r-strategists than an un-impacted one. The authors selected the W-statistic, derived from ABC analysis (Clarke and Warwick, 1994), as the most appropriate measure.

Abundance of opportunistic species

This measure was suggested to provide information that complements the information from the ABC analysis. In particular, the authors focus on "small, opportunistic species". The need to develop different suites of metrics for each community is recognised, as is the need to then establish community-based reference levels. There is no discussion of the circularity inherent in the selection of indicator taxa from the same data set that is then used to set their reference levels.

Vas Deferens Sequence index in female Nucella lapillus

This is the only metric proposed by de Boer *et al*. (2001) that addresses coastal and hard substratum communities. This measure is targeted at a single issue—of coatings containing TBT. It seems strange to target this issue using this index.

5.3.3 Summary

It is clear from this report that, in the short term, the most achievable measures of ecological quality are likely to be those which measure some aspect of species dominance in the community, and those which link the presence of fragile benthic species with direct impact from fishing gears. The way in which these ideas link to the WGECO framework is evaluated in Section 5.5.2.

5.4 EcoQOs for North Sea Threatened and Declining Species

5.4.1 Introduction

The availability of a working paper on threatened and declining species, and the recognition that fishing effects can impact individual fish and benthic species as well as communities, encouraged the WGECO to consider this topic. Threatened and declining marine mammals and seabirds were not considered.

To date there has been limited progress towards the development of ecological quality objectives for threatened and declining species in the North Sea (Lanters *et al.*, 1999). Attempts to list and categorise threatened and declining species are relatively well developed, especially by the IUCN and the Bern Convention, OSPAR and a number of national "red listing" programmes. IUCN uses a system of standardised, quantitative risk criteria, while work in OSPAR has tested the subjective selection criteria produced at meetings in Texel (1997), Horta (1999), and at the 1999 OSPAR IMPACT meeting. Within the OSPAR region, the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) has listed several fish species for which trade is prohibited or restricted (World Conservation Monitoring Centre, 1993), although only two of these (the coelacanth *Latimeria chalumnae* and totoaba *Cynoscion macdonaldi*) are exclusively marine fish. The applicability of the criteria used by CITES when evaluating the risk status of marine fish and invertebrates is currently under review (FAO, 2000). The criticisms of the CITES criteria would apply equally to those of IUCN, from which they were derived (FAO, 2000).

Within Europe, the Corine Biotopes Project, used as the basis for species and habitat selection for the Natura 2000 network, listed 43 species in a checklist of threatened fish, but all species were either freshwater, estuarine or diadromous (i.e., fish that migrate between freshwater and the sea). No exclusively marine fish was listed. Within Europe, the rare fish of the Wadden Sea have been described and listed (Berg *et al.,* 1995), and within the British Isles, eight species have been identified as threatened or suspected of being threatened (Swaby and Potts, 1990). Recent developments in endangered species management by the American Fisheries Society suggest the use of the intrinsic rate of population increase to allocate fish species to a range of productivity categories (Musick, 1998, 1999). This helps to distinguish declining species with high rates of production, such as those of commercial importance, from others that are less able to recover stock size.

5.4.2 Summary of preparatory work (Gubbay 2001)

The working paper by Gubbay (2001) has taken further the development of EcoQOs for threatened and declining species, and this section describes progress made, and suggests a way forward. The approach taken was a pragmatic one, and used the output of a range of existing selection criteria as the basis for producing a short-list of threatened and declining species. It is logical to assume that strictly protected or endangered/vulnerable species previously identified by IUCN, the Bern and Bonn Conventions, OSPAR and other national programmes should provide a comprehensive species list. The list was further refined by excluding commercially exploited species and those which have a brackish water phase or which do not occur in the North Sea. A clear link between one of six human activities (OSPAR, 2000a) and the decline of a species was also used as a final criterion to provide a possible explanation of the reasons for decline. Based on these criteria, Gubbay (2001) identified fifteen species, which were considered to be particularly suitable for the preparation of EcoQOs. The list comprised macrobenthic invertebrate species as well as teleosts and elasmobranchs (Table 5.4.2.1).

Table 5.4.2.1   Threatened and declining species identified on international "red lists" that could be used to inform EcoQOs (Gubbay, 2001).

Lithothamnion corallinoides	maerl
Lithothamnion calcareum	maerl
Zostera marina	eelgrass
Alcyonium digitatum	dead man's fingers
Lophelia pertusa	deepwater coral
Atrina fragilis	fan mussel
Cetorhinus maximus	basking shark
Dasyatis pastinaca	stingray
Galeorhinus galeus	tope
Mustelus mustelus	smooth hound
Myliobatis aquila	eagle ray
Prionace glauca	blue shark
Raja batis	common skate
Somniosus microcephalus	Greenland shark
Trachinus draco	greater weever

The list includes species that have already been identified for conservation and others that have not previously been identified as threatened or declining throughout the North Sea. Elasmobranchs are correctly identified as those known to be under the greatest threat, although the author acknowledges that some species, such as tope, skate, and basking shark, have been, or still are, subject to commercial fisheries and should therefore be included in OSPAR issue 1 (reference points for commercial fish species). The greater weever has also been identified as a species that has undergone recent decline in the southern North Sea, and, although occurring elsewhere in the Northeast Atlantic, is not an abundant species. The reasons for decline are unclear, and there is no obvious management measure that could be applied in order to restore the populations of greater weever.

While several of the invertebrate species are entirely appropriate for potential EcoQOs, others are less useful. For example, dead man's fingers, a relatively robust colonial Cnidarian, have been listed as declining in the Dutch and German Wadden Sea but are very abundant in the Channel, Western Approaches, and the Central and Northern North Sea. This illustrates the difficulties of using local metrics of decline to infer species status on a larger scale. Similarly, the fan mussel is a species that has the northern limits of its distribution in the North Sea and is more abundant to the southwest of the region. As species at the outer limits of their normal distribution generally show greater variability in population size than nearer the centre, they may not be the best choice of species to use as an ecological objective.

5.4.3   Summary

In accordance with the conclusions of the Scheveningen Workshop, the general EcoQOs suggested by Gubbay (2001) for threatened and declining species are related to the Reference Level, which is defined as the species abundance when the anthropogenic influence on the system is minimal. While it is certainly important that objectives for this group of species highlight the need for improvement in population status, it is not clear that it is necessary or even achievable to restore populations to their unperturbed size and/or extent. These are societal decisions that cannot be made here. However, the author makes a basically sound proposal for an overarching EcoQO, viz. "an absence of threatened and declining species in the North Sea where the principal causes of threat or decline are linked to human activities".

5.5   Application of the WGECO Framework

Our approach to setting EcoQOs is described in Section 3. The following sections describe the application of this framework to fish communities, benthic communities, and threatened and declining species. It contains WGECO recommendations for the further development of EcoQOs for these three issues.

5.5.1   Fish communities

The ecological quality of fish communities can be described by a broad array of metrics, including the relative abundance of individuals, their species membership, the biological traits of individuals, and their life history strategy. The number of realistic metrics, however, is restricted by the information that is available for these communities from existing surveys, our understanding of the processes involved, and our ability to communicate complex metrics effectively (Table 5.1.1). Life history characteristics that involve fecundity, for example, are available for only a few species, and body condition and growth are not known for all species. When data for a specific variable are known for only a subgroup of species, it should be judged whether the subgroup is representative of the fish community.

The two metrics, average weight of individual fish and average maximum length, are those which meet the scientific standards that were set and thus were considered by WGECO to be the most suitable metrics of community structure. They describe key features of the relative abundance and size distributions (Section 3.3).

Fishing is probably the human activity that affects the fish community most. Fishing is size-selective because the gear targets the larger individuals and allows the smaller ones to escape. As a result, fishing tends to change the size structure of a community resulting in a decreased average body size. The size-specific mortality caused by fishing also affects the species composition through differences in life history parameters. This occurs in spite of the relatively unselective nature of the fisheries with regard to target species relative to other species with similar general morphology and habitat usage patterns.

Several of the aspects of the fish community represented by different metrics appeared to be related and could be traced to one specific type of human activity: fishery induces size-specific mortality which changes the size-structure of the population. Therefore, the proposed metrics for the North Sea fish community are the average weight of individual fish and the average maximum length. From a conservation perspective, appropriate EcoQOs would move these metrics towards a larger proportion of large fish and would improve fisheries yields. Neither metric would discriminate between treatments that simply allowed individuals of exploited species to grow larger (and live longer, i.e., lower mortality) and treatments that changed the species composition towards a higher proportion of species with larger maximum possible weights and lengths (redistributing mortality across species, away from ones with greater maximum sizes).

5.5.2 Benthic communities

5.5.2.1 Introduction

In considering the broad aims of ecosystem management with reference to the benthos, the most important community metrics appear to be the species composition (including the presence of fragile, opportunistic, and keystone species), and its productivity and trophic structure (Section 3.3). It must also be emphasised that few, if any, studies provide a holistic picture of the benthic community. This is largely the result of the constraints of the sampling regimes required. Meio-infauna, macro-infauna, and epibenthos of soft sediments are rarely recorded in the same surveys, let alone in a way that would allow synthesis of the data into a "community picture". The situation is even more problematic when one considers hard grounds not amenable to grab/core sampling. There are two possible approaches to setting EcoQOs under these conditions. One alternative would be to focus on one aspect of the benthic community, and assume that if this component meets the EcoQ then other parts of the community will also conform. Alternatively, one could set EcoQOs for each component of an area: meiofauna, infauna and epifauna of sediments, sessile epibiota, and mobile epifauna for rocky areas. The latter approach would greatly increase the number of EcoQOs required and might involve problems of consistency among components in their response to management measures.

5.5.2.2 Metrics of EcoQ

In Section 5.3 only one metric, the presence of indicator or sensitive species, was identified as a good metric of ecological quality in benthic communities (Table 5.1.1). There are several indicator species, often consisting of structural biota such as corals (Fosså *et al*., 2000) and epifaunal organisms that are known to be sensitive to bottom fishing disturbance (Freese *et al*., 1999). These species are often apparent in bottom photographs and videos (Collie *et al*., 2000). The use of indicator species obviates the need to identify all species in benthic samples. However, in some benthic communities, there may be no obvious indicator species, suggesting that this EcoQ may not be comprehensive. Also, some epifaunal species that may make good indicators may have been removed by past fishing practices, yet present fishing practices may continue to impact the benthic ecosystem function.

5.5.2.3 Metrics that might be developed further

The presence of indicator or sensitive species cannot measure all the properties of benthic communities. Three other metrics measuring different properties of benthic communities scored quite highly using the framework in Section 3.3. These metrics were biomass, K-dominance curves, and the presence of non-indigenous species. Adoption of these as metrics of benthic EcoQ may address some of the shortcomings of the application of "the presence of indicator or sensitive taxa".

Biomass per m^2 is an aggregate measure of the benthic community that does not necessarily require all species to be identified. Biomass is also a component of benthic productivity. It is, however, difficult to measure benthic productivity directly, as often the benthic production is estimated by multiplying P/B ratios by biomass measurements. Hence, biomass is a more direct measure of benthic ecosystem quality. For example, significant decreases in benthic biomass have been measured in response to bottom fishing (Collie *et al*., 1997) and to sediment extraction in the eastern English Channel (Desprez, 2000). Disadvantages of using biomass as a metric are that environmental and anthropogenic

impacts on biomass variations may be confounded, and time-series of benthic biomass are also not available in most locations.

K-dominance curves may provide a useful measure of changes in species diversity in benthic communities. As this index was derived to measure impacts on benthic communities, it is clearly applicable as a measure of ecological quality of some parts of the benthic community. K-dominance curves are obtained by plotting cumulative ranked abundance against the log of species rank (Lambshead *et al.*, 1983) and the shape is a direct function of species relative abundance. Perturbations allow a subset of tolerant species to persist while the intolerant species disappear or become rare, hence the curve is expected to change in a predictable direction in response to disturbance. Shifts in K-dominance curves have been demonstrated in response to pollution (Warwick, 1986) and to experimental beam-trawling disturbance (Kaiser and Spencer, 1996). A potential disadvantage is that this graphical representation is somewhat difficult to comprehend and to communicate to policymakers and other non-specialists.

The presence and abundance of non-indigenous species may also be a useful metric of ecological quality. Non-indigenous species, both invertebrate and fish, have been widely spread by the discharge of ships' ballast water (ICES, 2000) and in some areas have markedly altered benthic food chains and community structure (ICES, 2000). As an example, the slipper limpet *Crepidula fornicata* has largely replaced the native oyster *Ostrea edulis* in Poole harbour (southern England) (ICES, 2000). The spread of non-indigenous species is clearly caused by human activity, but it can be very difficult to manage this activity and the invasion of indigenous species is likely to be impossible to reverse.

This suite of four metrics provides potentially useful measures of ecological quality in parts of the benthic community, but their practical application is limited by the history and intensity of benthic sampling across all parts of the meiofauna and macrofauna. While these metrics are most applicable to the benthic macrofauna and epibenthos, in principle they could also be applied to the meiobenthos, but there has been much less sampling to support their use in this part of the benthic community.

By not meeting our criteria fully, the inclusion of the additional three metrics is a weaker basis for ensuring that real conservation results from the management actions guided by advice developed within this approach. This does not mean that it is necessarily a bad approach, but WGECO expects it to be an approach with higher risk than would have occurred had more metrics met the required criteria.

5.5.2.4   Adding spatial dimensions

The purpose of EcoQOs is to assist in the development of an ecosystem approach to management (see Section 1.3.1). OSPAR has pragmatically adopted ten issues for which it believes that, if addressed by well-selected EcoQOs, ecosystem management will be assured. The problem is that this review has found that we are far from being able to identify scientifically sound and reliable metrics for enough of the benthic species and communities to address the ten issues effectively. It may, however, be possible to assure the quality of the ecosystem without necessarily having EcoQOs for all ten themes. Operationally, a restricted number of specific EcoQOs will ensure a more streamlined procedure. With regard to the benthos, we may therefore say that although it is critical that ecological qualities such as biomass, K-dominance curves, the presence of indicator or sensitive species, and the absence of non-indigenous species, are maintained, there is a more pragmatic approach that can be applied until science is able to support a full EcoQ and EcoQO framework.

Analyses of all the principal North Sea benthic data series which cover infauna and epibenthos (Buchanan, 1963; Basford *et al.*, 1990; Kingston, 1992; Kroncke and Rachor, 1992; Kunitzer *et al.*, 1992) highlight the critical role of the sediment in determining benthic community distribution. This is obviously superimposed on the broad framework of biogeographical and depth gradients (Pearson and Mannvik, 1998). The dynamics of these assemblages are also highly influenced by the quality of the overlying water, both in terms of pollution stress and also in response to variations in climate and pelagic dynamics (bentho-pelagic coupling) (Pearson and Rosenberg, 1986). A logical extension of this line of reasoning is that, if processes in the water column are managed and measures are taken to ensure that the benthic environment is not significantly altered (i.e., by sediment deposition, trawling impacts), then benthic ecological quality will be assured. We feel that to this should be added a clause recognising the need for a suitable larval supply and the fact that this supply may be spatially distant from the area under consideration. It must be emphasised that in this approach the benthic environment (habitat) comprises not just the physical features but also the biotic assemblage.

Adoption of this approach has a number of advantages:

- It reduces the total number of EcoQOs that need to be managed while still having a reasonable (if unquantified) likelihood of preserving benthic ecological quality;
- It is operationally easier to manage water quality and habitat destruction than directly managing the processes behind the more complex graphical presentations such as benthic K-dominance curves;
- It has an explicit spatial dimension and so is amenable to the spatial application of management techniques.

This approach is based on the relationship between benthic communities and their environment. It reaffirms that threats to benthic ecological qualities must be managed, but avoids the setting of EcoQOs based on any specific metric of the benthic community. The approach has obvious parallels with terrestrial conservation biology, where protection of a species' habitat is considered essential to protecting a species' population. The parallels between seafloor obligates and their structural habitat with terrestrial species is not a coincidence; "habitat" for both types of organisms has a potential stability over space and time that can be protected, in a way that "habitat" in the water column often lacks.

This approach does not obviate the need to monitor the properties of benthic systems which might provide an early warning of ecosystem disruption. High quality habitats may be a necessary, but may not be a sufficient, condition for stable and mature benthic communities. Such an approach also reinforces the need for effective Habitat and Water Quality EcoQOs. Habitat is used here in the sense defined in Section 3 and includes consideration of biotic associations (EUNIS levels 4–5). These EcoQOs must be developed and subjected to the rigorous selection process described here (Section 3). This approach does not lessen the need for effective management measures to protect threatened and declining species and to continue to monitor benthic communities. This approach may be seen as being pragmatic given our current level of knowledge about North Sea benthic communities but may also be seen as precautionary, as it will ensure protection of suitable habitats, provided that habitat EcoQOs are successfully identified and management measures are successful in protecting them.

5.5.2.5 Conclusion

There are at least three approaches to the setting of EcoQOs for benthic communities:

a) the use of a single metric, presence of indicator or sensitive species (Section 5.5.2.2);
b) the use of four metrics which provide a wider coverage of ecological qualities of benthic communities, but three of which fail to meet the strict criteria set by WGECO (Section 5.5.2.3); and
c) the use of EcoQOs for benthic habitats and water quality to provide the necessary protection to the ecological quality of the benthos (Section 5.5.2.4).

The first approach is more rigorous but very specific and suffers a number of limitations (Section 5.5.2.2). The second lacks the full rigour of the WGECO approach and so will provide a weaker basis for ensuring that conservation objectives are met. However, it covers a wider range of benthic community ecological qualities. The third approach avoids the need to identify specific measures of the community, and instead adopts a threats-based approach, recommending that protection of benthic community qualities is provided, at least initially, through the setting of rigorously developed and comprehensive habitat and water quality EcoQOs. These approaches are not incompatible, however, and WGECO favours the use of both the four metrics and the threat-based approach while further development of specific benthic EcoQOs is progressed, and the efficiency of each approach is tested in practice.

5.5.3 Threatened and declining species

Important ecological qualities of threatened and declining species are a measure of the population size, its trend and rate of change, and the assurance that current populations are sustainable. Invariably this requires information on population abundance collected either from catch statistics or independent surveys, with a sufficiently long time frame to assess the rate of change. The most important management objective for species that are under threat or are declining is to prevent further decline, and then restore population size and spatial extent. The reference levels for these metrics need to be debated but there are few alternatives to gradual improvement in population status.

Choosing the correct criteria for selecting species that are under threat is a crucial decision, and further elaboration of these criteria is required to select the most appropriate list of species for protection in the North Sea. Inappropriate criteria which do not take account of different species life histories can result in the selection of an incomplete list, ignoring some genuinely threatened species while including others in less need of action (Musick, 1998, 1999; IUCN, 1994; FAO, 2000). The list of species identified by Gubbay (2001), while identifying the elasmobranchs as requiring protection, also specify other species which are less appropriate, or less in need of protection. This largely resulted from the use of criteria that depended to some extent on the selection processes undertaken for different purposes by other bodies.

While recognising that there is a need to improve the population status of threatened and declining species, it is not at all clear to what level the populations should be restored. It is unlikely, however, that the reference level in an unperturbed environment will be a realistic or necessary objective. WGECO considers that there are insufficient data available for populations of threatened or declining species to identify levels that will ensure their long-term security, and inadequate guidance on what is the best reference level. The field of population viability analysis has been developed to address exactly these questions for tetrapod species. However, the analysis methods are highly sensitive to non-linearities and density dependence in population dynamics equations (Burgman *et al.,* 1993), and such attributes are

routine in fish population dynamics models, such as stock-recruit equations. There is no doubt, however, that the EcoQ of these populations, i.e., their absolute population size, trend and rate of change, must be recorded in detail.

Gubbay (2001) suggested an overarching EcoQO based upon the "absence of threatened and declining species in the North Sea where the principal causes of threat and decline are linked to human activities". This would seem a useful suggestion, because the single metric might be the number of such species and the objective to reduce the number to zero. WGECO did not discuss the potential at any depth, largely because threatened and declining species did not stand out in the terms of reference. However, this possibility should be evaluated further in the future, because it would overcome the problem of setting EcoQOs (and consequently collecting detailed data) for each individual species identified as belonging to this category. Only appropriate criteria for inclusion or exclusion need be developed. This will not necessarily be scientifically straightforward, nor without debate (FAO, 2000).

5.5.4 Concluding thoughts and the way forward

EcoQOs are a tool for integrating the management of human activities at the ecosystem level. The OSPAR definition of EcoQOs makes it clear that objectives need to be seen as components of an integrated set, which together comprise the overall expression of the structure and function of the marine ecosystem. Due to the complexity of the ecosystem, a pragmatic approach to setting EcoQOs is to target specific, separate issues as components of the ecosystem (OSPAR, 2000b). Most importantly, the metrics must be related to specific human activities, so that plausible and effective management solutions are advanced and the link between the human activity and the impact is clearly understood.

The preceding sections describe our recommendations on the development of EcoQOs for fish and benthic communities. We have also included in this section an analysis of the issue "threatened and declining species" for two reasons. First, we considered that many of the potential candidates in this category were either fish or benthos, and as such may anyway be discussed in separate sections, and secondly, a consultant's report on threatened and declining species was made available to WGECO for consideration.

Throughout this report, we have aimed to provide a limited number of metrics that can be made operational now. WGECO does not feel that it is necessary to select several EcoQs for all components of the system, especially where some characteristics of a particular metric, i.e., its degree of association with a human impact, and the complexity in deriving it, will largely prevent its use. There are also clear disadvantages in trying to apply some of the metrics that are described in Sections 5.2, 5.3, and 5.4, as EcoQOs. For example, if the metric is a good descriptor of some aspect of the community but its link with one or several human impacts is weak and poorly understood, there is little chance that the EcoQO will be effective. This failure will allow the approach to be undermined by those groups with an interest in maintaining the *status quo*.

We have identified several other metrics which may be useful in the medium term, but which we do not feel are sufficiently developed for immediate use. Given the time-scale for the current process of developing EcoQ, WGECO feels that there is a real need for additional development and testing of several metrics described here. There are some components of the ecosystem for which we do not have adequate measures (see Section 3.3.6). In the long term, there is a need for the development of new metrics to describe these key ecosystem qualities.

There is another reason why it is not thought necessary to set multiple objectives for all parts of the marine ecosystem. There are key environmental qualities such as habitat integrity and water quality which underpin the structuring of the marine ecosystem, particularly its benthic components, and influence communities at all trophic levels. By targeting these key features, and setting specific objectives to ensure that there is maintenance and improvement in quality, we can do a great deal to ensure the natural composition and function of the community.

References

Basford, D., Eleftheriou, A., and Raffaelli, D. 1990. The Infauna and Epifauna of the Northern North-Sea. Netherlands Journal of Sea Research, 25: 165–173.

Berg, S., Krog, C., Muus, B., Nielsen, J., Fricke, R., Berghahn, R., Neudecker, T., and Wolff, W.J. 1996. Red list of lampreys and marine fishes of the *Wadden Sea*. *In* Red lists of biotopes, flora and fauna of the trilateral W*adden Sea* area, 1995. Pp. 101–105. Helgolander Meeresuntersuchungen, Hamburg, vol. 50, Suppl.

Buchanan, J.B. 1963. The bottom fauna communities and their sediment relationships off the coast of Northumberland. Oikos. 14: 154–175.

Burgman, M.A., Ferson, S., and Akckaya, H. R. 1993. Risk assessment in conservation biology. Chapman and Hall, London, UK.

Christensen, V. 1996. Managing fisheries involving predator and prey species. Review in Fish Biology and Fisheries, 6: 1–26.

Clarke, K.R., and Warwick, R.M. 1994. Change in marine communities: An approach to statistical analysis and interpretation. Natural Environment Research Council, Plymouth UK. 144 pp.

Collie, J.S., Escanero, G.A., and Valentine P.C. 1997. Effects of bottom fishing gears on the benthic megafauna of Georges bank. Marine Ecology Progress Series, 155: 159–172.

Collie, J.S., Escanero, G.A., and Valentine, P.C. 2000. Photographic evaluation of the impacts of bottom fishing on benthic epifauna. *ICES Journal of Marine Science*, 57(4):987–1001.

Connell, J.H. 1978. Diversity in tropical rain forests and coral reefs. Science, 199: 1302–1310.

Connor, D.W., Brazier, D.P., Hill, T.O., and Northern, K.O. 1997. Marine Nature Conservation Review: Marine biotope classification for Britain and Ireland. Version 97.06. JNCC, Peterborough, UK.

Daan, N. 2001. A spatial and temporal diversity index taking into account species rarity, with an application to the North Sea fish community. ICES CM 2001/T:04.

de Boer, W.F., Daniels, P., and Essink, K. 2001. Towards ecological quality objectives for North Sea benthic communities. A report to RIKZ, Haren, NL, (contract RKZ 808) from Koeman en bijkerk bv, Haren, Nl, Report no. 2001-11. 62 pp.

Desprez, M. 2000. Physical and biological impact of marine aggregate extraction along the French coast of the eastern English Channel: short- and long-term post dredging restoration. *ICES Journal of Marine Science*, 57: 1428–1438.

FAO. 2000. An appraisal of the suitability of the CITES criteria for listing commercially-exploited aquatic species. FAO Fisheries Circular, No. 954. FAO, Rome. 66 pp.

Fosså, J.H., Mortensen, P.B., and Furevik, D.M. 2000. Lophelia-korallrev langs Norskekysten forekomst og tilstand. Fisken og Havet (journal), 2-2000.

Freese, L., Auster, P.J., Heifetz, J., and Wing, B.L. 1999. Effects of trawling on seafloor habitat and associated invertebrate taxa in the Gulf of Alaska. Marine Ecology Progress Series, 182: 119–126.

Greenstreet, S.P.R., and Hall, S.J. 1996. Fishing and the groundfish assemblage structure in the North-western North Sea: an analysis of long-term and spatial trends. Journal of Animal Ecology, 65: 577–598.

Gubbay, S. 2001. Development of ecological quality objectives for the North Sea: Threatened and declining species. A report to Birdlife International, March 2001.

Hall, S., Raffaelli, D., and Thrush, S. 1994. Patchiness and disturbance in shallow water benthic assemblages. In Aquatic ecology- scale, pattern and process, pp. 333-375. Ed. by P. Giller, A. Hildrew, and D. Raffaelli. Blackwell Science, Oxford.

ICES. 2000. Report of the Working Group on Introductions and Transfers of Marine Organisms. ICES CM 2000/ACME:07. 65 pp.

International Union for Conservation of Nature. 1994. IUCN red list categories. IUCN Species Survival Commission, Gland, Switzerland.

Kaiser, M.J., and Spencer, B.E. 1996. The effects of beam trawl disturbance on infaunal communities in different habitats. Journal of Animal Ecology, 65: 348–358.

Kingston, P.F. 1992. Impact of Offshore Oil Production Installations on the Benthos of the North-Sea. *ICES Journal of Marine Science*, 49: 45–53.

Kroncke, I., and Rachor, E. 1992. Macrofauna Investigations Along a Transect From the Inner German Bight Towards the Dogger Bank. Marine Ecology-Progress Series, 91: 269–276.

Kunitzer, 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.

Lambshead, P.J.D., Platt, H.M., and Shaw, K.M. 1983. The detection of difference among assemblages of marine benthic species based on an assessment of dominance and diversity. Journal of Natural History, 17: 859–874.

Lanters, R.L.P, Skjoldal, H.R., and Noji, T.T. (Eds) 1999. Ecological Quality Objectives for the North Sea. Basic Document for Workshop on Ecological Quality Objectives for the North Sea. 1–3 September 1999. Scheveningen, the Netherlands. RIZK Report 99.015. Fisken og Havet 10-1999.

Lindeboom, H., and de Groot, S.J. (Eds.) 1997. The effects of different types of fisheries on the North Sea and Irish Sea Benthic Ecosystems. European Commission, Brussels, IMPACT II Final Report. Contract Report AIR2-CT94-1664. 396 pp.

Musick, J. 1998. Endangered marine fishes: Criteria and identification of North American stocks at risk. Fisheries, 23(2): 28–30.

Musick, J. 1999. Criteria to define extinction risk in marine fishes. Fisheries 24(12): 6–14.

OSPAR 2000a. Quality Status Report 2000. Region II. Greater North Sea. OSPAR Commission, London.

OSPAR. 2000b. The further development of Ecological Quality Objectives within OSPAR. Issues and work plan agreed by ASMO 2000. OSPAR 00/8/1, Annex 1.

Pauly, D., Christensen, V., Dalsgaard, J., Froese, R., and Torres, F.C. Jr. 1998. Fishing down marine food webs. Science, 279: 860–863.

Pearson, T.H., and Mannvik, H.P. 1998. Long-term changes in the diversity and faunal structure of benthic communities in the northern North Sea: natural variability or induced instability? Hydrobiologia, 376: 317–329.

Pearson, T.H., and Rosenberg, R. 1986. Feast and Famine: Structuring factors in marine benthic communities. In Organization of communities: Past and present, pp.373–398. Ed. by J. H. R. Gee and P. S. Giller. Blackwell Scientific.

Piet, G.J. 2001. Development of ecological quality objectives for the North Sea fish community. Working Paper at the 2001 WGECO meeting.

Rice, J.C. 2000. Evaluating fishery impacts using metrics of community structure. *ICES Journal of Marine Science*, 57: 682–688.

Rijnsdorp, A.D., van Leeuwen, P.I., Daan, N., and Heessen, H.J.L. 1996. Changes in abundance of demersal fish species in the southern North Sea between 1990–1995 and 1906–1909. *ICES Journal of Marine Science*, 53: 1054–1062.

Rogers, S.I., and Ellis, J.E. 2000. Changes in demersal fish assemblages of British coastal waters during the 20th Century. *ICES Journal of Marine Science*, 57: 866–881.

Swaby, S.E., and Potts, G.W. 1990. Rare British marine fishes: identification and conservation. Journal of Fish Biology, 37 (Supplement A): 133–143.

Thrush, S.F. 1991. Spatial Patterns in Soft-Bottom Communities. Trends in Ecology and Evolution, 6: 75–79.

Warwick, R.M. 1986. A new method for detecting pollution effects on marine macrobenthic communities. Marine Biology, 92: 557–562.

World Conservation Monitoring Centre 1993. Checklist of fish and invertebrates listed in the CITES appendices. Joint Nature Conservation Committee, Peterborough, UK.

Witbaard, R. and R. Klein 1997. The effects of southern North Sea beam trawl fishery on the bivalve molluscs *Arctica islandica* L. (Mollusca, Bivalvia). *In* Tree of the sea: the use of internal growth lines in the shell of *Arctica islandica* L. (Bivalvia, Mollusca) for retrospective assessment of marine environmental change. R. Whitbaard, PhD Thesis, University of Groningen, pp. 84–101.

6 RAPID SCREENING OF METRIC FOR USE AS EcoQs

6.1 Introduction

In the process of conducting the evaluation of candidate metrics reported in Sections 3 and 5, we realized that a sequence of steps are followed in the screening process. Laying out the sequence of steps will facilitate further testing and selection of EcoQs in diverse application contexts. The series of trials for metrics that are to be considered for EcoQ status are not meant to replace or contradict the seven (or eight) criteria listed in Sections 3.2 and 3.3. Rather serve as an operational structure to ensure efficient screening of the candidate metrics..

The comparisons are set up as a cascading series of tests. A trial metric, after passing a series of increasing restrictive tests, would be given provisional EcoQ status, as an informative metric. This would indicate that it is useful and informative. EcoQs would be further tested to ascertain if they would be suitable for management and the formulation of accompanying EcoQOs. We call this full EcoQ status. It would require the ability to define limits or desirable states and the ability to be manipulated. This sequence brings together ten days of discussions and accumulating experience during the meeting, but is proposed without having been tested and may well be amended with experience. The order is suggested as being increasingly restrictive, analogous to a series of sieves. We will use EcoQ(I) to denote one which is informative but unlikely to be useful in a management context and EcoQ(F) for a full EcoQ.

Table 6.1.1.1 Cascading tests for indices; if a test is passed go down to the next.

	Status	Test	Comment
	Metric		
↓		Preliminary screening	expert scientific opinion
↓		Detection screen	does the index detect a known event
↓		Signal quality screen	stability, error rates, S/N…
↓		Uniqueness/contribution	How does it act in concert with others
↓	**EcoQ(I)**		
↓		Manageability	Can objectives be defined?
↓		Interference/contribution	How does it act in concert with others
	EcoQ(F)		

Most of the following comments are based on case studies and real data as opposed to simulations. When a specific topic is amenable to simulation, it will be so indicated. However, a simulation would have to be fairly sophisticated to serve as a basis for index comparisons. Top-down and bottom-up controls, prey switching, susceptibility to exogenous factors (temperature, pollution…) and perhaps spatial structure might all be essential components of a simulation, if the simulation is to be informative about the performance of the metric. However, real-world experience tends to lack replication and controls, so simulations, although complex, will often be essential in full evaluation of candidate metrics.

Preliminary screening of metrics

Metrics may be classified as single-species sole (sss), single-species indicative (ssi), emergent, or aggregate. The single-species (sss) indicator is only applicable at the species level and is not asserted to have significant ecological implications – for example, the abundance of a non-rare, not highly coupled, non-charismatic species. The indicator single-species is of broader interest. The emergent indices are only applicable at the community or higher level; an example is the r/K ratio. The aggregate metrics are aggregates of single-species metrics (mean length across species).

Detection screen

Case studies, particularly for the North Sea, require an agreement that an event has happened for which management action was justified (whether action was taken or not). Events in this context are things like (excess) fishing and pollution events. The event must be expressed at the community level to test "emergent" property candidates. Does the reduction (say to 10% of historical levels) of a single keystone predator signal an important change to the ecosystem? Once there is agreement on what "events" should have been picked up by useful metrics, which metrics showed trends

or changes in value that corresponded to the "events"? The first screening includes those indices that experts feel have ecological relevance and are likely to perform well. These can be compared (statistically) to the timing of the onset of the event. If time-series are not available, spatial data may be used for comparison: areas where the event took place versus areas where it did not.

Signal quality screen

This screening consists of estimation of signal/noise sensitivity, false positive rates (hysteria), false negatives (misses). Replicate trials and controls are difficult to find in ecosystems. Areas and seasons tend to be fished for decades. Closed areas and fishery closures offer some controls (in the experimental sense) that could possibly be used to assess the responsiveness of the signal to a perturbation. Signal-to-noise ratios need to consider a number of aspects of the noise. The noise may be estimated from bootstrapping or parametric analyses. The mean level of noise (or some index of a central tendency like the standard deviation) can be compared to the dynamic response of the metric, a traditional signal-to-noise ratio. This ratio gives an insight into the risk of the metric to miss real events. Also of importance are the tails of the noise distribution—what could happen on an unusually bad year. Is the noise correlated—will a number of bad years happen in a row? Insights are gained from meta-analysis of noise in related systems in an attempt to anticipate events that have not been seen in the analysed system. The more extreme events, especially correlated ones, may trigger false positives.

Uniqueness/contribution screen

If they pass the detection of event test, the next series of questions needs to be considered. Are the metrics independent? From one perspective managers do not need six indices changing at the same time, so one should pick the easiest to measure, or the most charismatic. On the other hand, if they are not perfectly redundant, it could be argued that a suite of related metrics might be more informative than any single one, just as meta-analysis is thought to add strength to analyses of numerous, individually noisy, data sets. However, whenever there are multiple indices, it is necessary to ask if they are contradictory, and if so, how to use models for resolving them or methods of integration, perhaps Boolean operators or weighted means, or fuzzy logic. Some metrics are model-based derivatives of others, for example, production may be composed of survivorship, growth and reproduction metrics. They will be correlated with others but the linkage is explicit. The signal detection attributes, or cost, could be used to discriminate among correlated metrics.

Integration of metrics will require some structure. The object would be to improve the reliability of detection with a combination of metrics. One model could be Boolean, any alarm will be carried through. This means that the most sensitive metrics will dominate. Another could be some sort of averaging weighted by importance or reliability. Fuzzy logic can also be of use in integrations of this sort.

Manageability screen

The scientific contribution to screening EcoQ(I)s as potential EcoQ(F)s may not be as important as social, economic, and political considerations. One role for scientists is in the expected ecological effectiveness of a measure. How fast would the action prompted by the metric have an effect and how tightly coupled is the EcoQ to the control (fishing effort reduction, cessation of oil exploration). Ecosystems can behave in unexpected ways. Irrespective of effort reduction, populations may be in a new (stable) regime; for example pink salmon in Peterman (197x) or the failure of some Northwest Atlantic cod stocks to recover after eight years of closure.

A related issue in elevating an EcoQ to full EcoQ(F) status is the requirement of the ability to define relevant management objectives. As discussed with metrics of species diversity, it is very often hard to know what measure(s) is the right one to "correct" an anomaly in a diversity index, harder still to convince managers that the measure is essential, and perhaps hardest of all to convince stakeholders restricted by the measure to comply with it, if the debate centers on the value of a diversity index.

Interference/contribution screen

How potential EcoQ(F)s act in concert would be amenable to simulation. A simulation would have to be fairly sophisticated to serve as a basis for EcoQ(F) with applied EcoQO comparisons. Top-down and bottom-up controls, prey switching, susceptibility to exogenous factors (temperature, pollution, etc.) would be required and if possible spatial heterogeneity.

6.1.1   Concluding remarks

These proposals are suggested as a complement to the work carried out in Section 5. The steps of the screening do not match the properties used in Section 3.3 closely. Most of these properties (for example, "Comprehensive and communicable", or "Easily and accurately measured") are dealt with in the first step of the screen. On the other hand,

several of the steps in the screen are collapsed in the Section 3.3 property "High response to signal from human activity...". Two of the remaining screens are focused on how groups of metrics work in concert, a characteristic not explicitly addressed in Section 3.3, but reserved for "later" consideration.

CHAPTER 2

ECOLOGICAL THEORY AND HOW TO MEASURE IMPACTS OF FISHING ON ECOSYSTEMS

7 The Need for Linking Ecological Theory to Evaluating Ecosystem Effects of Fishing

7.1 Linking the Theoretical Frameworks for Studying Fishing Effects and Ecosystem Structure, Function, and Dynamics

Even though there is great diversity of opinion in the complex fields of community ecology and ecosystem dynamics, fisheries theory clearly needs to forge clear links to these other bodies of theory. Fortunately, there are many points of potentially direct contact between the theoretical foundations of fisheries, and the diverse theoretical views of community structure (and particularly community change, which is the issue with ecosystem effects of fishing) and ecosystem dynamics. Hence it should be possible to build bridges with sound, recognizable foundations in each field.

For example, yield per recruit and stock-recruit functions are fundamental to fisheries theory and practice. These functions have parameters which are directly interpretable in terms of concepts from community ecology and general population dynamics. For example; the slope of the stock-recruit function is interpretable as the r parameter of classic population models (e.g., logistic), whereas the asymptotic parameter of a Beverton Holt model (or peak of a Ricker model) can be linked directly to K (Shepherd and Cushing, 1990). Recent work to link growth models to size spectra (Gislason and Lassen, 1997) provide another illustration of taking a model with traditional fisheries parameters and forming quantitative links to community level attributes.

7.2 How to Focus on Theoretical Frameworks with Greatest Promise

We can distinguish two types of ecosystem effects of fishing, direct and indirect. Amongst the direct effects are the changes in the target stocks, effects on non-target fish populations, direct mortality of benthos and other non-target organisms (wildlife impacts). Indirect effects include changes in levels of predation of fish, benthos and plankton resulting from changes in the size, and size structure, of the fish assemblage, changes in benthic productivity arising from changes in species composition, changes in the carbon mass balance of the system due to transfer of marine production to terrestrial systems (man) and avian scavengers, and alterations in the fluxes of nutrients due to changes in the food web and direct effects on the benthic–water column flux rate.

At present there exist a multiplicity of models of ecosystem function which have been applied to various marine communities. The majority of these are likely to apply in some circumstances, but most can be seen as being on a continuum between stochastic dynamics, biotically structured with strong stabilizing feedbacks and systems with strong non-linear dynamics (chaotic) (DeAngelis and Waterhouse, 1984). To date, no one has assessed the most widely accepted models to explicitly derive the predictions they make with regard to ecosystem effects of fishing. A critical comparison of such predictions should provide guidance on which parameters could form useful measures of ecosystem effects of fishing.

We are now in a position where considerable information has been assembled on the direct effects of fishing on target species, non-target fish and wildlife and on benthos (IMPACT II; Lindeboom and de Groot, 1998). These can be fed into a review of models of ecosystem function in order to test the predictions derived from the models.

The Working Group has made some progress in looking at the more tractable indirect effects—predation rate and ecosystem mass balance models. Further progress requires guidance from model constructors as to the likely indirect effects of fishing. The time would therefore seem ripe to review the direct effects studies in a holistic manner and against a background of ecosystem function models. This would then allow us to specifically address our levels of understanding and gaps in knowledge of the indirect effects.

References

DeAngelis, D.L., and Waterhouse, J.C. 1987. Equilibrium and nonequilibrium concepts in ecological models. Ecological Monographs, vol. 57, no. 1. Pp. 1–21.

Gislason, H., and Lassen, H. 1997. On the linear relationship between fishing effort and the slope of the size spectrum. ICES CM 1997/DD:05.

Lindeboom, H.J., and de Groot, S.J. (Eds.) 1998. IMPACT II. The effects of different types of fisheries on the North Sea and Irish Sea benthic ecosystem. NIOZ-Rapport 1998-1, RIVO-DLO Report C003/98. 404 pp.

Shepherd, J.G., and Cushing, D.H. 1990. Regulation in fish populations: Myth or mirage? Philos. Trans. R. Soc. Lond., B., vol. 330, no. 1257. Pp. 151–164.

8 Theory and the Performance of Data-Based Community Metrics for Evaluating Ecosystem Effects of Fishing

8.1 Integrating Information on North Sea Assemblages from Different Surveys

Long time-series of marine fisheries surveys are one of the most important datasets that we have available, and these can provide invaluable insights into the temporal changes in fish populations. In general, regular surveys become more valuable the longer they are continued, and it is therefore important that established surveys are continued. Recently it has become possible to extend some of these series backwards in time using archived data from earlier generations of scientists who did not have access to electronic data storage. Some of these datasets extend back almost to the start of the 20th century, and relate to a period which experienced less extensive fishing impact than the present day.

Before we can commence analyses of the longest time-series possible, we must identify all such time-series of data which are collected within the North Sea ecosystem, and which relate specifically to fisheries assemblages. These datasets fall into two distinct categories, those collected during research vessel surveys undertaken by European Research Laboratories, and those collected by other methods, unrelated to fishing surveys or routine market sampling of landings at ports.

8.1.1 Research vessel surveys

The earliest Research Vessel (RV) survey data from the North Sea that have been computerised relate to a series by Dutch and English vessels from 1906 to 1909. These are described by Rijnsdorp *et al.* (1996). For each haul the numbers of the larger fish species caught are available for 10-cm groups, and some information on the bottom fauna was also recorded. Smaller fish species not considered 'food fish' were not recorded systematically but were sometimes recorded under the heading 'bottom fauna'. The distribution of the fishing stations in the North Sea was fairly uneven, but the southeastern North Sea was well covered by most surveys.

The Scottish August Groundfish Survey (AGFS) has taken place every year since 1980. This survey was undertaken by the RV Explorer until 1982 and then by the RV Scotia. The Scotia is approximately twice as powerful as Explorer and its trawl speed was approximately 1.21 times that of the older vessel (Greenstreet *et al.*, 1999), consequently the distance covered in hours' trawling was greater. The 48-foot Aberdeen trawl has been used throughout this survey. This gear is identical to that used by the Marine Laboratory Aberdeen in groundfish survey work extending back to the early 1920s. The data for the entire AGFS, as well as for the months of July to September, are available in electronic format back to 1925.

One of the longest North Sea time-series for demersal species is provided by the first quarter International Bottom Trawl Survey (IBTS), which began in 1960/1961 and has been carried out annually in February since 1965. Initially the target species was herring (hence the initial name of the International Young Herring Survey), and the survey coverage was restricted to the southern and central North Sea, but the coverage was extended when it was realised that the surveys could also provide recruitment indices for cod, haddock, whiting, and Norway pout (hence the change to the International Young Fish Survey). Since 1969 the Skagerrak and Kattegat have been sampled and from 1974 the entire North Sea has been included in the survey area. The survey has evolved into a highly standardised, internationally co-ordinated trawl survey, in which nine countries have been participating (ICES, 1992). Although commercially important species have been the principle target, length data of all the bycatch species have been collected by most participants. The otter trawl gears employed during the series have varied, and over a period of several years up to 1982 have now become standardised on the French designed GOV trawl which has a high vertical net opening. Before this time there was also some inconsistency in the survey area covered annually, which has also been resolved. Data are stored on the ICES IBTS database, but only data collected since 1983 are completely computerised. For the period 1970−1982 the records are incomplete and many data are still in paper format, stored in different laboratories. The contributions of the different countries to the first quarter IBTS database is reported in ICES (1999).

The need to monitor flatfish stocks in the heavy beam trawl fishery in the shallow coastal waters of the southern North Sea led to the introduction of fisheries-independent surveys using beam trawls. By 1988 a number of countries which border the North Sea had developed these surveys, and these targeted different age ranges of flatfish and used beam trawls and vessels of different size and specification. Collation and analysis of some of the data derived from these surveys was initially focused on the North Sea and eastern Channel, but during the early 1990s all surveys in Subareas IV and VII were included (ICES, 1991). During the 1980s, five countries which border the North Sea and

western waters of the UK had developed a range of beam trawl surveys (ICES, 2001). Some of these surveys were designed to sample pre-recruit (0- and 1-group) sole and plaice on nursery grounds with light gears, while others used beam trawls of commercial design to catch juveniles and adults. Six of these surveys were modified following recommendations of the Beam Trawl Study Group to develop a more standardised sampling protocol (ICES, 1994).

In addition to these surveys already included in the activities of the Beam Trawl Study Group, there are others which have only recently been transferred to electronic format. The Sole Net Survey (SNS) was initiated in 1969 to obtain pre-recruit indices for 1- and 2-group plaice and sole. The survey consists of 10 transects parallel or perpendicular to the continental North Sea coast between the Dutch/Belgian border and Esbjerg in Denmark, and a number of fixed stations is fished on each transect. The DFS was initiated in 1970 to obtain pre-recruit indices of brown shrimps and 0- and 1-group plaice and sole. For this survey two types of gear are used, a single 3 m beam trawl in the Wadden Sea and Scheldt estuaries, and a pair of 6-m beam trawls along the Dutch coast. Three areas were distinguished: DFS1 the Scheldt estuary, DFS2 along the Dutch coast, and DFS3 the Wadden Sea estuary.

A coastal nursery ground survey has also been operating in the coastal waters (<20 m) on the east and south coast of England since the 1970s, and this survey uses the 2-m beam trawl and 1.5-m push net (Rogers and Millner, 1996). These two gears were specifically designed to have similar efficiency and selectivity so that the catches could be directly compared.

8.1.2 Other sampling methods

A variety of other sampling techniques have been used to collect time-series data on the abundance of North Sea fish. Examples in this category mainly include the use of fixed stations to collect fish on a regular basis. One example of this technique has been described by Phillipart *et al*. (1996), in which the fish of the Dutch Wadden Sea have been collected from a tidal inlet using a kom-fyke trap. This gear, operating since 1960, is emptied every day from Monday to Friday, and operates throughout the summer period. A similar series of data have been provided by the catches of fish which impinge on the filter screens of power station cooling water intakes (Henderson, 1989). For both these examples of fixed station recording, there is no information on the relationship of the catches with the total population abundance, but for power stations at least, the fish catch is thought to be highly effective at sampling a wide range of demersal and pelagic species.

8.1.3 Problems with combining gear catches

For a number of reasons, different fishing gears vary in catch efficiency for different sizes of fish, and this is the main problem encountered when comparing catch data collected between one survey and another. In those cases where catches have been combined, the swept area of each gear has often been used (Rijnsdorp *et al*., 1996). For beam trawls this is the fixed width of the trawl opening, but for otter trawls this parameter varies with the water depth and speed of towing. As all fishing gears are selective and the catchabilities of fish at size vary, this standardisation to the swept area of the gear does not resolve all the problems, and relative catchabilities can only be obtained when all gears are fished simultaneously on the same ground.

The selectivity and catchability of a demersal trawl is influenced by the way that the net is rigged, the type of ground gear, the length of the towing warp and otter trawl sweeps, the mesh size in the cod end and the speed at which the gear is towed. In addition, the ground over which the gear is towed and the tidal conditions during towing will also influence catch rates of fish. Gear parameters are most variable for otter trawls, where for the same gear, headline height varies with tow speed and depth of fishing, and catch rates are influenced by the length of the sweeps (ICES, 1996). The best way to ensure that fish catch rates from different surveys can be combined is to use identical gears operated in precisely the same manner. This situation rarely occurs, however, and a number of studies in the North Sea have attempted to get conversion factors between vessels and gear by undertaking comparative fishing trials. The gear used during the IBTS first quarter surveys, the GOV otter trawl, is recommended to have sweep lengths of 60 m for fishing in shallow areas and 110 m at stations deeper than 70 m to avoid possible changes in gear parameters due to depth and to the length of the warp. Comparative gear trials conducted in 1994 using warps of two different lengths concluded that catch rates of cod, haddock, whiting, and herring were different, and that for some species, particularly herring, the catch rate at size also varied.

It is possible that beam trawls of the same design but of different widths may not show a linear relationship in their catch rates of all demersal species, and that the use of different attachments (chain mat, flip-up ropes, etc.) will also affect the gear efficiency. Comparative fishing exercises (Groeneveld and Rijnsdorp, 1990) compared the 4-m beam trawl with chain mat and flip-up ropes, and the 8-m beam trawl with tickler chains and flip-up ropes. During surveys in 1990 and 1991, catch ratios of dab, sole, and plaice between the two gears were consistently different (ICES, 1993), suggesting that it was not possible to derive raising factors to convert the catch numbers of one gear into that of another gear.

When considering long time-series, particularly those covering periods of 20 years or more, it is important that the fishing characteristics of the different RVs that may have been used in the collection of the data are taken account of. Analyses that depend on absolute numbers of each species sampled may well be affected by differences in the areas swept by the fishing gear as a result of vessels of differing horse power towing the gear at different speeds. Some species diversity indices, for example, are particularly sensitive to variation in sampling effort. In the 70-year time-series (Greenstreet and Hall, 1996), four different RVs were involved. The area swept by the standard fishing gear used varied by a factor of approximately 1.89 from the most to the least powerful vessel.

A further related issue involves trawl tow duration. During the 1970s and 1980s many groundfish surveys used standard one-hour trawl durations. Recently some institutes have reduced this to half-hour tows, causing potential problems with the analysis of long-term trends in species diversity. Furthermore, other institutes have continued to trawl for one hour, making contemporary comparisons difficult. The issue of sample size dependence of some community metrics is particularly relevant when it comes to considering the effect of variation in trawl duration.

A final consideration in comparing different datasets, again related to the problems of sampling effort dependence, are the possible consequences of the protocols used for handing catches once they are brought aboard the vessel. It is frequently impossible to sort and handle every single fish in a large catch. Subsampling is necessary. Straight forward proportional division of the catch, sorting one fraction and discarding the rest effectively reduces sampling effort at that station, and it reduces the probability of finding rare fish. In biodiversity studies it is important that not only is the haul duration standardised, but that the entire catch is sorted in such a way as to obtain a reasonably accurate estimate of even the rarest species.

These examples of comparative gear trials suggest that the levels of standardisation currently used in the IBTS Database are important to ensure that catch data are collected in a similar way, and that catch comparisons between gears are important. They also illustrate how difficult it is to combine catches from similar gears. The relative catchability of different species by different gears is an important consideration in deciding which species to include in the species-suite in a particular analysis. To combine the catch rates of fish between, for example, the otter trawl catches of the IBTS, and the beam trawl catches of the beam trawl surveys in the North Sea, will require extensive species by species knowledge of the selectivity of each gear. These data are not yet available.

Catch efficiencies of species caught by the International Beam Trawl Surveys, for example, have been assumed to be in direct proportion to the width of the trawl. It has been necessary to make this assumption for these gears, in order to prepare a spatially extensive dataset and develop our understanding of the spatial dynamics of demersal species in the North east Atlantic. In this example, corrections between surveys are possible for tow duration and gear width, but more complex corrections for area swept are required for otter trawl gears. These decisions, however, are only partly based on scientific evidence and also include an element of judgement which is based on the experience of scientists who are familiar with the operation of the surveys and the properties of the gears. The selectivity of the gears to target species is also an important consideration. It is clear that more research is needed in order to ensure that these judgements are based on more sound scientific arguments.

8.1.4   References

Greenstreet, S.P.R., and Hall, S.J. 1996. Fishing and the ground-fish assemblage structure in the north-western North Sea: an analysis of long-term and spatial trends. Journal of Animal Ecology, 65: 577–598.

Groeneveld, K., and Rijnsdorp, A.D. 1990. The effect of a flip-up rope on the catch efficiency of an 8-m beam trawl. ICES CM 1990/B:16, p. 13.

Henderson, P. 1989. On the structure of the inshore fish community of England and Wales. Journal of the Marine Biological Association, UK, 69: 145−163.

ICES. 1991. Report of the Study Group on Beam Trawl Surveys in 1990. ICES CM 1991/G:81, p. 57.

ICES. 1992. Manual for International Bottom Trawl Surveys. Revision IV. ICES CM 1992/H:3/Addendum.

ICES. 1993. Report of the Study Group on Beam Trawl Surveys. ICES CM 1991/G:5, p. 92.

ICES. 1994. Report of the Study Group on beam trawl surveys in the North Sea and Eastern Channel. ICES CM 1990/G:59.

ICES. 1996. Report of the International Bottom Trawl Survey Working Group. ICES CM 1996/H:1.

Philippart, C.J.M., Lindeboom, H.J., Meer, J. van der, Veer, H. W. van der., Witte, J.I. 1996. Long-term fluctuations in fish recruit abundance in the western Wadden Sea in relation to variation in the marine environment. *ICES Journal of Marine Science*, 53: 1120−1129.

Rijnsdorp, A.D., Leeuwen, P.I., Daan, N., and Heessen, H.J.L. 1996. Changes in abundance of demersal fish species in the North Sea between 1906−1909 and 1990−1995. *ICES Journal of Marine Science*, 53: 1054−1062.

Rogers, S.I., and Millner, R.S. 1996. Factors affecting the annual abundance and regional distribution of English inshore demersal fish populaitons: 1973 to 1995. *ICES Journal of Marine Science*, 53: 1094−1112.

8.2   Theory of Community Metrics – Multivariate Indices and Analyses of Communities

In this section we review our current knowledge regarding changes in marine fish assemblages which could be attributed to variation in fishing practice. We first consider studies carried out within the North Sea, then take account of studies carried out in other ecosystems, and finally consider how examination of spatial differences in the behaviour of different community metrics might help to establish fishing pressure as the cause of observed changes. We therefore present our information in three subsections, each section dealing with one of these themes.

Only summaries of the materials and methods used in various Working Papers, and the key points from the results and discussion sections of the papers, have been included here. The analytical techniques applied in the various studies differ considerably. In some instances this makes comparisons between studies somewhat problematic, however we try to compare the implications of the various sets of results. We go on to present analysis of data available to the Working Group using a standard suite of community metrics, in an effort to make direct comparisons more simple. Lessons from these analyses lay the foundation for more integrated analyses of a considerable amount of groundfish survey data, collected by different institutes, using different fishing vessels and fishing gear, and covering a wide range of geographic locations and marine habitats.

8.2.1   North Sea region

8.2.1.1   Review of current information

8.2.1.1.1   Monitoring of changes in small-scale fish assemblages in the North Sea (S. Ehrich and C. Stransky, Working Paper)

Bottom trawl surveys were carried out on RV Walter Herwig III (using the standard GOV trawl) and RV Solea (with a similar, but smaller otter trawl) in eight boxes in the North Sea during summer-time from 1987–1996 (Figure 8.2.1.1.1.1). The towing time was 30 minutes with a speed of about 4 knots. Trawling positions and directions within the boxes were randomly selected. In general, at least 20 trawl hauls were made in 3 days within each box.

The catch data were standardised for one hour. The standardised abundance data were used to calculate a diversity index for each box and year. An MDS analysis was performed on the species composition similarity matrix (using a Bray-Curtis index on root-root transformed data) to compare species composition within and between the boxes. The boxes sampled with different vessels were analysed separately.

The boxes with the greatest geographical distance (e.g., A and D, F and H) show the least similarity in fish fauna. The boxes situated more closely to each other (e.g., B and C, E and K) were more similar (Figures 2.2.1.1.1.2 and 2.2.1.1.1.3). No obvious trend in species composition similarity over the years within the boxes was apparent, suggesting little in the way of long-term change in species composition in each box. The number of species caught in each box was relatively constant over the study period; however, variability in the Hill's N1 was high but showed no trend.

8.2.1.1.2   Long-term changes in North Sea fish assemblages based on different beam trawl surveys

Five datasets describing demersal fish abundance, sampled using beam trawl surveys (BTS), were used. These surveys covered the Dutch coastal zone, the Wadden Sea estuary, the Scheldt estuary, the coastal Southern North Sea (SNS) and BTS survey. The surveys differ in the gear type used and the geographical area covered. Because of these changes the Dutch Fisheries Survey (DFS) is more suited to studying changes in smaller fish, whereas the SNS and the BTS are relatively more affected by changes in larger fish.

Analysis of the surveys shows considerable changes in both size structure and species composition. All surveys showed, to a greater or lesser degree, a general trend in the size structure together with year-to-year fluctuations. The general trend was a shift in the relative distribution of biomass towards the smaller size classes probably caused by the effect of fisheries exploitation in the region. The year-to-year fluctuations in size structure were to a large extent caused by differences in year-class strength of the most abundant species, such as dab and plaice.

The species composition was also affected by changes observed in size structure in combination with life history characteristics of the different fish species. For the SNS survey it was shown that the abundance of species with a large size at maturity decreased while that of species with a small size at maturity increased.

8.2.1.2   Analyses carried out by WGECO

8.2.1.2.1   Dutch Beam Trawl Survey data

The SNS was initiated in 1969 and is a national survey carried out only by The Netherlands. It is aimed at obtaining pre-recruit indices for 1- and 2-group plaice and sole. The survey is carried out using a 6-m beam trawl, rigged with 4 tickler chains and a sole net with a mesh size of 40 mm in the codend. The standard station grid of the survey consists of 10 transects parallel or perpendicular to the continental North Sea coast between the Dutch/Belgian border and Esbjerg, Denmark. On each transect a number of fixed stations is fished. There is no further stratification. In total, 55 hauls are made each year, with at least 4 hauls in a transect. The gear is fished with a fishing speed of 3.5 knots and the haul duration is 15 minutes. Three areas were distinguished: SNS1 south of Texel, SNS2 between Texel up to the German Bight, and SNS3 north of the German Bight.

The DFS was initiated in 1970. It aims at obtaining pre-recruit indices of brown shrimps and 0- and 1-group plaice and sole. For this survey two types of gear are used. In the Wadden Sea and Scheldt estuaries a single 3-m beam trawl is used. The gear is rigged with a shrimp net of mesh size 20 mm in the codend and one light tickler chain. A ground rope with wooden or rubber bobbins is used. Along the Dutch coast, fishing is done with a pair of 6-m beam trawls. The gear is rigged with a shrimp net in a similar way as the 3-m beam trawl. Fishing speed is 2–3 knots, depending on the strength of the current. Three areas were distinguished: DFS1 the Scheldt estuary, DFS2 along the Dutch coast, and DFS3 the Wadden Sea estuary.

Temporal trends in Hill's N1 and N2 are shown for both data sets in Figures 2.2.1.2.1.1 and 2.2.1.2.1.2. Some interesting trends are apparent. Both indices show species diversity to be consistently higher in SNS1 compared with SNS3. Hill's N1 suggests that diversity in SNS2 tracks diversity in SNS3 at the start of the time-series, flips in 1976 to track SNS1 until 1989, then tracks SNS3 for the next two years, before reverting to tracking SNS1. Species diversity, as sampled by the DFS, appears to fluctuate widely. No consistent temporal trend is suggested by either index. However, between-area variation is high from 1969 to 1977, whereupon species diversity in the different areas appears to converge for 10 years or so, before once again diverging towards the end of the time-series.

8.2.2   Other oceans and seas

8.2.2.1   Review of current information

8.2.2.1.1   Comparing diversity of coastal demersal fish faunas in the North-East Atlantic

International Beam Trawl Survey data were used to analyse the assemblage structure of commercially important and non-target demersal fish species collected from coastal waters of the Northeast Atlantic (Figure 8.2.2.1.1.1). Catches were dominated by a small number of species, which occurred in large numbers and at high biomass. The most abundant species (plaice, dab) were typical of shallow, uniform sandy and muddy seabed which occurs extensively throughout the southern North Sea and to a limited extent in western UK waters. Renyi's diversity index family was used to rank the diversity of coastal sectors throughout the region. The limited access of the southern North Sea to species-rich southern faunas and the uniform nature of the seabed were largely responsible for the lower diversity of North Sea coastal faunas compared to those in the Channel and west of the UK. West of the Dover Strait, the more heterogeneous substrate supported a more diverse fauna of smaller-sized fish with the occurrence of southern species such as red gurnard and thickback sole and an increasing abundance of elasmobranches. Patterns in community structure over such a wide spatial scale and without historical perspective can be explained by biogeographic factors, seabed structure, and the influence of regional hydrography. Inferring from these patterns an impact by anthropogenic factors (towed gears) is unlikely to be achieved.

8.2.2.1.2   Spatial patterns of groundfish assemblages on the continental shelf of Portugal

Five groundfish surveys (four in autumn and one in spring) were conducted off Portugal from 1985–1989 by RV Norugea using the Norwegian Campellen Trawl (horizontal opening 14 m, vertical opening 4 m, groundrope with rollers, codend mesh size 40 mm). Sampling was performed at randomly selected stations on longitudinally and latitudinally determined 36 depth strata (at least two stations per depth strata) along the coast of Portugal (depth range 20–500 m) during the day. Trawl duration was 30 min and the tow speed about 3 knots. Fish were identified, in general, to the species level. Any species that comprised at least 1% of the total biomass in at least one of the surveys was included in the analysis. The catch data were log-transformed before further analysis. The stations were compared by pairs using the Bray-Curtis dissimilarity index and clustered using group average hierarchical agglomerative cluster analysis.

Based on cluster analysis, the following depth groups of stations were separated (Figure 8.2.2.1.2.1) as follows:

Shallow-Northern Group (20–100 m). The following species dominated: sardine (*Sardina pilchardus*), mackerel (*Scomber scombrus*), horse mackerel (*Trachurus trachurus*), and European squid (*Loligo vulgaris*). Other commonly found species were: *Merluccius merluccius, Trisopterus* spp., and *Polybius henslwoi.*

Shallow-Southern Group (20–100 m). Horse mackerel and axillary seabream *Pagellus acarne* made usually over 50% of the total biomass. Other species usually occurring in catches were *M. merluccius, S. pilchardus, L. vulgaris*, and seabreams.

Intermediate Group (80–180 m). *M. merluccius, S. pilchardus, L. vulgaris*, and *T. trachurus* occurred most often whereas *M. merluccius* dominated in the catches.

Deep-Northern Group. Blue whiting (*Micromesistus poutassou*) made up the majority of the catches with *M. merluccius* composing the remainder of catches.

Deep-Southern Group. The biomass was dominated by blue whiting (*M. poutassou*) with the following fishes occurring in significant amounts: boarfish (*Capros aper*), *M. merluccius*, and some crustaceans.

These five clusters of stations allowed the mapping of groundfish assemblage areas on the Portuguese Shelf (Figure 8.2.2.1.2.2), i.e., the areas characterized by a relatively homogenous groundfish composition. Major changes in the composition of the demersal community off the Portuguese Shelf are associated with depth. The change is especially sharp at depths between 100–200 m, where separation of Deep and Shallow groups takes place. *M. poutassou* dominated in the catches from the deep region (150–400 m) whereas *S. pilchardus, T. trachurus*, and *S. scombrus* were the majority in shallow areas (20–120 m). The second major biogeographic transition occurs in near-shore waters (< 120 m depth). The relative proportion of *S. scombrus* and *Trisopterus* spp. decreases and that of *M. merluccius, L. vulgaris*, and sparids in catches increases.

8.2.2.1.3   Spatial distribution of species assemblages in the Celtic Sea and the Bay of Biscay

Two bottom trawl surveys were carried out in the Bay of Biscay and the Celtic Sea shelves and upper slopes in autumn 1990 and spring 1991 by using the GOV 36/47 trawl with 20 mm codend mesh size and of estimated headline height and distance between wings of 4 and 18–20 m, respectively. Trawling was carried out during day-time with a speed of 4 knots and duration of 30 minutes. In the Bay of Biscay, the survey area was divided, according to latitude, into 4 blocks and stratified sampling was performed in the following depth ranges: 15–30, 31–80, 81–120, 121–160, 161–200, 201–400 and 401–600). 137 and 142 hauls were made in 1990 and 1991, respectively. In the Celtic Sea, sampling was performed at fixed stations (grid length 25 nautical miles). 56 and 57 hauls were made in 1990 and 1991, respectively.

The total weight of the catch and abundance of fish species in a catch was recorded. In 1990, only selected fish species were measured whereas in 1991, this was performed for all species caught. Log-transformed catch data of fish species were classified by applying a hierarchical ascending classification procedure to their first PCA coordinates and the groups obtained were then clustered by using a moving centres procedure.

Six types of fish assemblages could be identified within the study area (Figures 2.2.2.1.3.1 and 2.2.2.1.3.2), as described below:

Fish assemblage of the central shelf of the Bay of Biscay (mean depth: 100–112 m, muddy bottoms prevail). Characteristic fish species: *Lesueriogobius friesii, Merluccius merluccius, Cepola rubescens, Nephrops norvegicus, Arnoglossus laterna*, and cephalopods from genus *Alloteuthis*.

Fish assemblage of the western shelf (mean depth: 151–152 m, mostly sandy bottoms). Characteristic species are: *Lepidorhombus whiffiagonis, Capros aper, Todaropsis eblanae*, and *M. merluccius*. Seasonally characteristic species are: *Illex coindeti, Argentina sphyraena,* and *Callionymus maculatus* in autumn and *Pollachius virens* in spring.

Fish assemblage of the Continental slope (mean depth: 310–351 m, hard bottoms dominate in some areas). Characteristic species: *Malacocephalus laevis, Chimaera monstrosa, Galeus melastomus, Helicolenus dactylopterus,* and *Lepidorhombus boscii*.

Fish assemblage of the southern Celtic shelf (mean depth: 143–148 m, mostly coarse sand bottom). Characteristic species: *Aspitrigla cuculus, Arnoglossus imperialis,* and *Raja naevus*.

Fish assemblage of the northeastern Celtic shelf (mean depth: 115 m, soft bottom type). Characteristic species: *Trisopterus esmarki, Gadus morhua, Merlangius merlangus, Melanogrammus aeglefinus, Hippoglossoides platessoides limandoides, Eutrigla gurnardus, Glyptocephalus cynoglossus,* and *Pleuronectes platessa*. In addition, *Squalus acanthias* is typical in autumn and *Clupea harengus* in spring.

Shallow water fish assemblage of the Bay Biscay (mean depth: 39–47 m). The highest number of species has been recorded in this assemblage. Characteristic species: *Sardina pilchardus, Trachurus mediterraneus, Scomber scombrus, Engraulis encrasicolus, Sprattus sprattus, Merlangius merlangus, Ammodytes tobianus, Hyperolpus lanceolatus, Spondyliosoma cantharus, Dicentrarchus labrax, Callionymus lyra, Dicologlossa cuneata, Solea vulgaris, Echiichthys vipera, Trachinus draco, Loligo vulgaris, Sephia officinalis,* and *Crangon crangon.*

8.2.2.1.4 Analysis of the spatial and temporal variability of the size spectrum of the fish community in the Bay of Biscay, 1987–1995

The data used in this analysis were gathered from seven bottom trawl surveys with RV Evehoe using GOV 36/47 trawl from 1987–1995. Until 1989, sampling was performed by the following scheme: 100 hauls were made at fixed stations and 35 hauls at changeable stations. From 1989 onwards, all hauls were performed at fixed locations (ICES, 1991, 1997).

Table 2.2.2.1.4.1   The number of trawlings by depth ranges and years in the Bay of Biscay.

Stratum	1	2	3	4	5	6	7	
Depth (m)	< 31 m	31–80	81–120	121–160	161–200	201–400	> 400	Total
1987	14	21	30	33	14	9	9	130
1988	14	23	26	38	15	11	7	134
1989	15	24	27	40	17	13	6	142
1990	16	21	28	38	18	10	6	137
1992	14	22	26	25	11	5	4	107
1994	11	18	22	25	12	8	5	101
1995	14	18	22	25	18	10	7	114

Fish size spectra were constructed for each depth range and year by summing catch numbers over species within 5-cm size classes ranging from 20–24 cm to 75–79 cm. The size spectrum for the whole Bay of Biscay is the mean of depth strata spectra weighted by the numbers of hauls in each respective depth strata. For the long-term data analysis, a set of fish species was selected for analysis over all years. All treatments were performed separately for all species measured during the study period and for the subset of demersal species.

For data analysis, fish abundance data and size-class categories were log-transformed. Analysis on long-term trends of slopes and intercepts of the size spectra were performed as outlined by ICES (1996). The survey data were also disaggregated by depth strata. To study the covariance of fish abundance by size spectra, the following models were applied:

$y = \mu + \beta x + \alpha_i + \beta_i + \gamma_j + \beta_j x + \delta_{ij} + \varepsilon$ (1) and $y = \mu + \beta x + \alpha_i + \gamma_j + \delta_{ij} + \beta_{ij} x + \varepsilon$ (2), where
y - fish abundance
x - fish size-class
β - slope of the size spectra
α_i - depth effect
γ_j - year effect
δ_{ij} - interaction term of the year and the depth effect
β_{ij} - slope of the size spectra by depth strata and years
μ - the general mean term
ε - the error term

When both demersal and pelagic species were treated together, regressions between fish abundance and size-class were significant for each year. Compared to those in the North Sea, the slopes in the Bay of Biscay were lower (ranging from −6.2 to −3.4 and −7.4 to −6.1, respectively) and the intercepts were higher (13.4–19.0 and 25.1–29.8, respectively). No significant long-term trend in slopes or intercepts was found.

For demersal species only, slopes and intercepts were better determined than for all species, mainly due to better fitting of the linear model. The slopes were considerably higher and intercepts lower with significant long-term trends in both parameters (Figure 8.2.2.1.4.1). However, the conclusions of the analysis of disaggregated data did not change when the pelagic species were removed from the analysis.

The analysis showed that all the effects incorporated into the models were highly significant. The size-class term accounted for the largest amount of variance of the fish abundance whereas the class variables (depth and year) accounted for a smaller, but still significant part of the variation (Table 2.2.2.1.4.1).

8.2.2.1.5 Application of experimental trawl data for estimation of fish stock dynamics in the Gulf of Riga

Species richness and fish abundance dynamics, including currently non-assessed and non-target species, were monitored during 1974–1986 and 1994–1996 by using catch per unit effort data from monthly experimental bottom trawl surveys (Figure 8.2.2.1.5.1). The trawls were carried out in daylight with a mean trawling speed of 2.5 knots, the towing duration was 30 minutes, estimated trawl opening area was 40 m and mesh size in the codend was 8 mm (from May to July 1994; 20 mm). In 1981–1986, surveys were conducted only in autumn (1–3 surveys in September–

November). For further analysis, the basin was divided into two regions: shallow coastal area (Pärnu Bay, strongly dominated by freshwater fish) and deeper parts of the basin (> 20 m, dominated by marine species and glacial relicts). Abundance of all species in a catch was determined through direct counts or through sub-sampling if the catch was too large. Except for sticklebacks (*Gasterosteus aculeatus* and *Pungitius pungitius*) and gobies (*Pomatoschistus* spp.), the fish were identified to the species level.

The abundance data were analysed by the following GLM model:

log (mean catch+1) = Year + Month + ε,

where mean catch is the monthly average catch by number of a species; Year and Month - the year-effect and month-effect, respectively; ε - the error term. The year-effect in the model was used as an index describing dynamics of fish stock abundance. For estimation of species richness, the following GLM model was applied:

log (mean number of fish species) = Year + Month + ε,

where mean number of fish species is monthly average number of fish species in a catch per area.

Abundance dynamics of fish species inhabiting mostly the shallow region

This category includes all the freshwater species living in the basin and also certain euryhaline species. With a single exception (gobies), fish of this category have shown an increase in stock size (e.g., sticklebacks) or no clear tendency in the abundance estimates is evident (Figure 8.2.2.1.5.2).

Abundance dynamics of fish species living in deeper areas

This category includes marine boreal species and glacial relicts. Two general tendencies in the stock abundance of these species could be pointed out:

• Generally higher abundance in the late 1970s, lower values during the 1980s, and recent increase in stock size (e.g., sprat *Sprattus sprattus*, smelt *Osmerus eperlanus*, eelpout, *Zoarces viviparus*) (Figure 8.2.2.1.5.2).
• Obvious decrease in the abundance or extinction of some species, which were rather abundant during the late 1970s, from the community of the Gulf of Riga from the mid-1980s (e.g., cod *Gadus morhua callarias*, common sandeel *Ammodytes tobianus*, fourhorned sculpin *Triglopsis quadricornis*).

Species richness

In the shallow region, the model estimates indicate, with certain exceptions from 1983–1985, a slight increase in the mean number of fish species. Whereas an obvious decline in this characteristic was evident in deeper areas during 1978–1985, an increase followed in 1994–1996 (Figure 8.2.2.1.5.3). The increase was caused by an elevated frequency in the occurrence of pelagic euryhaline species (sprat *Sprattus sprattus* and sticklebacks) and some cold-water species (smelt *Osmerus eperlanus* and eelpout *Zoarces viviparus*) while other demersal and cold-water fishes, (e.g., sea snail *Liparis liparis* and fourhorned sculpin *Triglopsis quadricornis*), found relatively frequently in the 1970s, were absent or only rarely present in the hauls.

These changes probably reflect different responses of fish species from those different groups (freshwater species, marine fish, and glacial relicts) to alterations in the main abiotic and biotic parameters of the basin (due to natural causes and anthropogenic activities), but also affected by stock (over) exploitation and the presence or absence of the only large marine predator in the ecosystem, cod.

8.2.2.2 Analysis carried out by the Working Group

8.2.2.2.1 Gulf of Riga

Dynamics of three community metrics indices (species richness, Hill's N1, and Hill's N2) were investigated in the Gulf of Riga (Baltic Sea) during 1974–1986 and 1994–1996. The survey data and its collection methods are described above. For the estimation of species richness, the following GLM model was applied:

log (mean number of fish species) = Year + Month + ε,

where mean number of fish species is monthly average number of fish species in a catch per area; Year and Month - the year-effect and month-effect, respectively; and ε - the error term. As only the autumn period was sampled during all the study years, for Hill's N1 and Hill's N2 calculations, fish abundance data from this period was used.

In the shallow region the model estimates indicate, with certain exceptions in 1983–1985, a slight increase in the mean number of fish species. This is not the case in the deeper areas 1978–1985 which showed an increase in 1994–1996 (Figure 8.2.2.2.1.1). As mentioned above the last increase was caused by elevated frequency in the occurrence of pelagic euryhaline (sprat *Sprattus sprattus* and sticklebacks) and some cold-water species (smelt *Osmerus eperlanus* and eelpout *Zoarces viviparus*) while other demersal and cold-water fish (e.g., sea snail *Liparis liparis* and fourhorned sculpin *Triglopsis quadricornis*), found relatively frequently in the 1970s, were absent or only rarely present in the hauls. It seems likely that most of these changes were mainly governed by alterations in environmental conditions and predation by cod rather than the direct effect of fishing.

Long-term dynamics of Hill's N1 and Hill's N2 indices suggest the following patterns of the two spatially separated fish communities in the Gulf of Riga:

1)   The fish community in the shallow area is, in general, more heterogeneous than that in deeper areas with no clear trend in the indices calculated;

2)   The fish community in the deep area exhibits slight increasing tendency in the heterogeneity measures over the years studied.

8.2.2.2.2   Barents Sea bottom trawl survey

The bottom trawl survey data used for the Barents Sea analysis comes from a combined acoustic and bottom trawl survey for demersal fish in the Barents Sea which has been conducted annually since 1981. Only data from 1985–1996 are used. The survey methodology is described in Dalen *et al*. (1982), Hylen *et al*. (1986), and Jakobsen *et al*. (1997). The sampling trawl used is the Campelen 1800 shrimp trawl with 80-mm mesh size in the front. Until 1989 the trawl was equipped with a rubber bobbin, but in 1989 a rockhopper ground gear was introduced. This improved the catch efficiency of the trawl (especially the smaller gadoids). This change in ground gear is likely to show up in several of the analyses. The survey area was increased in 1993, but only data from the central regions covered in all years are used in the analysis. The survey area with subareas and strata system, together with the trawl stations taken in 1996, are shown in Figure 8.2.2.2.2.1. Note that only data from subareas A, B, C, and D are used in the analysis.

Both diversity indices fluctuated throughout the time-series, each index tracking the other. A slight negative trend in species diversity was apparent (Figure 8.2.2.2.2.2). This was corroborated by variation in k-dominance curves calculated over three four-year periods, combining data over years in each period. Dominance was greatest in the period 1993–1996, and least in 1985–1988 (Figure 8.2.2.2.2.3). Species evenness showed a decline over the course of the time-series (Figure 8.2.2.2.2.4), but little trend in species richness was apparent (Figure 8.2.2.2.2.5). Changes in the relative abundance of the most abundant species accounted for these trends, rather than any change in the number of species in the assemblage.

Size spectra were examined over two size ranges, 20–50 cm (Figure 8.2.2.2.2.6) and 50–100 cm (Figure 8.2.2.2.2.7). Trends in the slope are shown in Figure 8.2.2.2.2.8. The slope for 20–50 cm fish went through two ossilatory cycles, being least negative in 1988 and 1995. The trend in slope for fish 50–100 cm in length followed an opposite cyclical trend, being most negative in these years and least negative in 1992. This analysis was repeated, this time excluding cod and haddock (Figures 2.2.2.2.2.9 and 2.2.2.2.2.10, respectively). This had the effect of damping the second cycle (Figure 8.2.2.2.2.11).

Variations in N1, N2, and species richness were examined in area D separately and compared with trends for areas A, B, and C combined (Figures 2.2.2.2.2.12 to 2.2.2.2.2.14). Species diversity was generally lowest in area D and this could be explained by the presence of fewer species in this area.

8.2.3   Spatial patterns and the relationship with fishing

8.2.3.1   Review of current information

8.2.3.1.1   Changes in the groundfish species assemblage of the northwestern North Sea between 1925 and 1996

This study examined long-term changes in the structure and composition of the groundfish species assemblage in four regions of the northwestern North Sea (Figure 8.2.3.1.1.1). Scottish fisheries research vessel data primarily collected during from July to September during 1925 to 1996 were analysed. Trends in the whole groundfish assemblage and in a

subset of the assemblage, which is not targeted by commercial fisheries, were described. These trends are then related to variations in the patterns of fishing activity in each of the areas.

Species diversity in the whole groundfish assemblage had declined in the three areas where fishing pressure had been greatest; in the area where fishing pressure had been least historically, no trend in species diversity was detected (Figure 8.2.3.1.1.2). Only in the area where fishing pressure had been the highest, and at high levels for the longest period of time (Figure 8.2.3.1.1.3), was a negative trend in species diversity observed among the non-target species assemblage. Spatial variation in species diversity was clearly defined. Within the whole groundfish assemblage, diversity was greatest in the inshore and southern regions and least in the offshore northern area, while among the non-target species assemblage, the spatial diversity gradient was reversed.

Multivariate analyses indicated long-term changes and between-area differences in the species composition of both the whole groundfish assemblage and the non-target species subset (Figure 8.2.3.1.1.4). However, these changes consisted for the most part of subtle variations in the relative and absolute abundance of a few key species rather than involving major species replacement events. Only one species showed any marked increase in abundance and this was a case of a dominant species becoming even more abundant.

Examination of species-aggregated length-frequency distributions suggested a shift over time towards assemblages more dominated by smaller fish. This was mainly apparent, however, in the whole groundfish species assemblage; the length-frequency distributions of non-targeted species were much more stable.

8.2.3.2 Analyses carried out by the Working Group

8.2.3.2.1 Monitoring fish assemblages in small defined areas in the North Sea

The survey design, the position of the boxes (Figure 8.2.1.1.1.1) and some results are already described in Section 2.2.1.1.1. The 8 boxes are distributed over the entire North Sea and cover a depth range from 110 m (Box D) to less than 40 m (Box A) in the German Bight. The boxes are situated in areas where the main fishing gear used and the degree of fishing effort (hours fished) differ considerably. Fishing effort distribution in 1991 was used to calculate the mean annual effort within the ICES rectangles which are touched by the boxes. Effort data for the German fishing fleet and of the STCF data set (international effort without German data) were combined to estimate total international effort. The boxes can be separated into 5 categories: Box A represents an area of high fishing effort, mainly by beam trawl; boxes B and D belong to areas of medium fishing effort using otter trawls; boxes E and F represent areas of moderate fishing effort by beam trawlers; box C belongs to an area of low fishing effort conducted mainly by beam trawlers; and box H represents an area of low fishing effort mainly by otter trawlers (Figure 8.2.3.2.1.1).

Pelagic species such as herring, sprat, mackerel, and horse mackerel, can dominate the species composition since, when they occur, they can occur in very high densities. Under such circumstances diversity indices, such as Hill's N1 and N2, decrease to very low values. To make the results comparable with other papers investigating changes in fish assemblages in the North Sea (Greenstreet and Hall, 1996) the calculations were done including and excluding these 4 pelagic species.

The annual changes in the diversity (Hill's N1 and N2 indices) within the boxes are shown in Figures 2.2.3.2.1.2 to 2.2.3.2.1.9. As expected, there was a general shift to a lower range when the four pelagic species were excluded, but no trends in any of the 8 boxes were indicated by either index whether the pelagic species were excluded or not. Even in Box A, situated just outside the plaice box, and where beam and otter trawl effort has increased by a factor of two from 1982 to 1993 (de Groot and Lindenboom, 1994), no trend in species diversity was noted.

Changes in mean body weight were investigated in the heavily fished Box A and the less fished Box C to look for evidence that the size spectrum of the exploited fish assemblage had shifted towards the smaller-sized end of the size spectrum (e.g., Rice and Gislason, 1996). The pelagic species were excluded from the calculation. Length data were not available at the meeting so variation in the mean individual weight was examined instead (Figure 8.2.3.2.1.10). Apart from the high value in 1992 in Box A related to unusually high immigration of one-year old cod into the German Bight, mean individual weight has declined in both boxes with little difference between boxes, suggesting little effect of fishing effort.

Investigations of the fish assemblage in a small box of 100 nm² in an area of high fishing pressure over the last decade revealed no changes in the abundance of species like dab and grey gurnard, related to changes in fishing effort (Draft IMPACT-II Report). The data presented here also indicate the difficulties in detecting fishing-related changes in fish species assemblages. Neither the diversity metrics nor the mean individual weight revealed any temporal trends associated with increasing fishing effort. Nor did comparisons between boxes differing in the intensity to which they were fished show differences.

8.2.4 Concluding comments and discussion on multivariate metrics

There are two main features of the spatial analyses described here which are worthy of further comment. The first is the extensive amount of survey data that are available to research institutes in Europe collected using a wide range of vessels, gears, stratification methods, sampling accuracy, and seasonal and spatial resolution. Within a study these various factors are adjusted and modified to optimise the sampling programme within a region, often to conform with the national requirements for obtaining stock abundance estimates. This degree of national variation does not, however, help us to combine datasets on a larger spatial scale and in a general international perspective on assemblage structure. The varied and serious issues associated with combining datasets have already been described in detail in Section 2.1. The second important feature of the studies described here is the wide range of statistical analyses that have been applied to these datasets. Among the list of routine statistics that have been described here are Principal Components Analysis, k-dominance statistics, similarity and dissimilarity coefficients, and agglomerative clustering. More novel techniques such as size spectrum analysis (Section 2.3.2), phylogentic relatedness analysis (Section 1.2.2.1), and the use of diversity profiling (see Section 2.3.2) has also been applied.

In summary, while the analyses of each regional assemblage is informative and valuable, difficulties in combining these datasets, and the large number of different analyses applied to them, suggests that finding and describing a common theme or themes will be a challenge. For this reason, comparative analyses were carried out on a selection of datasets which were considered to be the most comparable, using a range of simple and widely used community metrics. The results of some preliminary analyses have been described. The covariation in both the N1 and N2 metrics was evident in many of the analyses. This is largely because the fish assemblages studied are all strongly dominance oriented, a fact underlined where k-dominance curves were applied. These were often the most informative metric revealing changes in species relative abundance and they are amenable to statistical comparison. In fact, ranked abundance biomass comparison curves (ABC) where both the cumulative biomass and abundance of ranked species are plotted may well be even more informative. This conveys information about shifts in size and/or weight of the dominant species as well as changes in their numerical contribution to the total assemblage. This may be particularly pertinent where fish assemblages are concerned, since we have a sound theoretical basis to expect fishing to shift assemblages towards the lower end of the size spectrum.

As demonstrated here, examination of trends in the value of particular metrics over time are frequently inconclusive. A major problem here is the short time scale of most datasets (one or two decades) compared with the time over which the North Sea and other marine ecosystems have been fished (one or two centuries). It should be remembered that changes in the populations of exploited species (abundance, age structure) are most apparent when fishing mortality is actually increasing (or decreasing). In periods of relatively constant fishing mortality (albeit high) a stable population structure develops. Over most of the last two decades many exploited species have undergone more or less constant high fishing mortality, giving rise to relatively stable (even if overfished) populations during the period when most groundfish assemblages have been sampled. The most obvious changes will have occurred when populations of the exploited species themselves were undergoing the greatest change, that is, when fishing mortality was increasing relatively fast from zero, well before the start of most surveys.

The spatial analyses presented here also suggest a second confounding influence, the powerful effects of abiotic and biotic characteristics of the environment in determining the species composition and structure of fish assemblages. This suggests that temporal variation in environmental parameters may have marked effects on the assemblages under investigation, and these may mask any trends arising from variations in fishing practices. However, careful analysis of spatially referenced fish species abundance data, which includes information regarding spatial variation in abiotic and biotic features of the environment as well as quantitative measures of fishing activity, may well help to unravel the different effects of these interacting influences.

8.2.5 References

Dalen, J., Hylen, A., Jakobsen, T., Niacin, O., Randa, K. and Smedstad, O. 1982. Norwegian investigations on young cod and haddock in the Barents Sea during the winter 1982. ICES CM 1982/G:41. 20 pp.

de Groot, S.J., and Lindeboom, H.J. 1994. Environmental impact of bottom gears on benthic fauna in relation to natural resources management and protection of the North Sea. NIOZ Rapport 1994/11 RIVO-CLO report CO 26/94.

Greenstreet, S.P.R., and Hall, S.J. 1996. Fishing and the ground-fish assemblage structure in the north-western North Sea: an analysis of long-term and spatial trends. Journal of Animal Ecology, 65: 577–598.

Greenstreet, S.P.R.; Spence, F.E.; and 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 Fishing effects in northeast Atlantic shelf seas: patterns in fishing effort, diversity and community structure. V. Changes in structure of the North Sea groundfish species assemblage between 1925 and 1996. Fisheries Research (Amsterdam) Vol. 40, no. 2, pp. 153–183.

Hylen, A., Nakken, O., and Sunnanå, K. 1986. The use of acoustic and bottom trawl surveys in the assessment of North-East Arctic cod and haddock stocks. *In* A Workshop on Comparative Biology, Assessment and Management of

Gadoids from the North Pacific and Atlantic Oceans, pp. 473–498. Ed. by M. Alton. Northwest and Alaska Fisheries Centre, Seattle, WA.

ICES. 1991. Report of the Study Group on the Coordination of Bottom Trawl Surveys in Sub-areas VI, VII and VIII and Division IXa. ICES CM 1991/G:13.

ICES. 1996. Report of the Working Group on Ecosystem Affects of Fishing Activities. ICES CM 1996/Assess/Env:1.

ICES. 1997. Report of the International Bottom Trawl Survey Working Group. ICES CM 1997/H:6.

ICES. 1999. Report of the International Bottom Trawl Survey Working Group. ICES CM 1999/D:2.

ICES. 2001. Report of the Working Group on Beam Trawl Surveys. ICES CM 2001/G:06.

Jakobsen, T., Korsbrekke, K., Mehl, S., and Nakken, O. 1997. Norwegian combined acoustic and bottom trawl surveys for demersal fish in the Barents Sea during winter. ICES CM 1997/Y:17. 26 pp.

Rice, J., and Gislason, H. 1996. Patterns of change in the size spectra of numbers and diversity of the North Sea fish assemblage, as reflected in surveys and models. *ICES Journal of Marine Science,* Vol. 53, no. 6. Pp. 1214–1225.
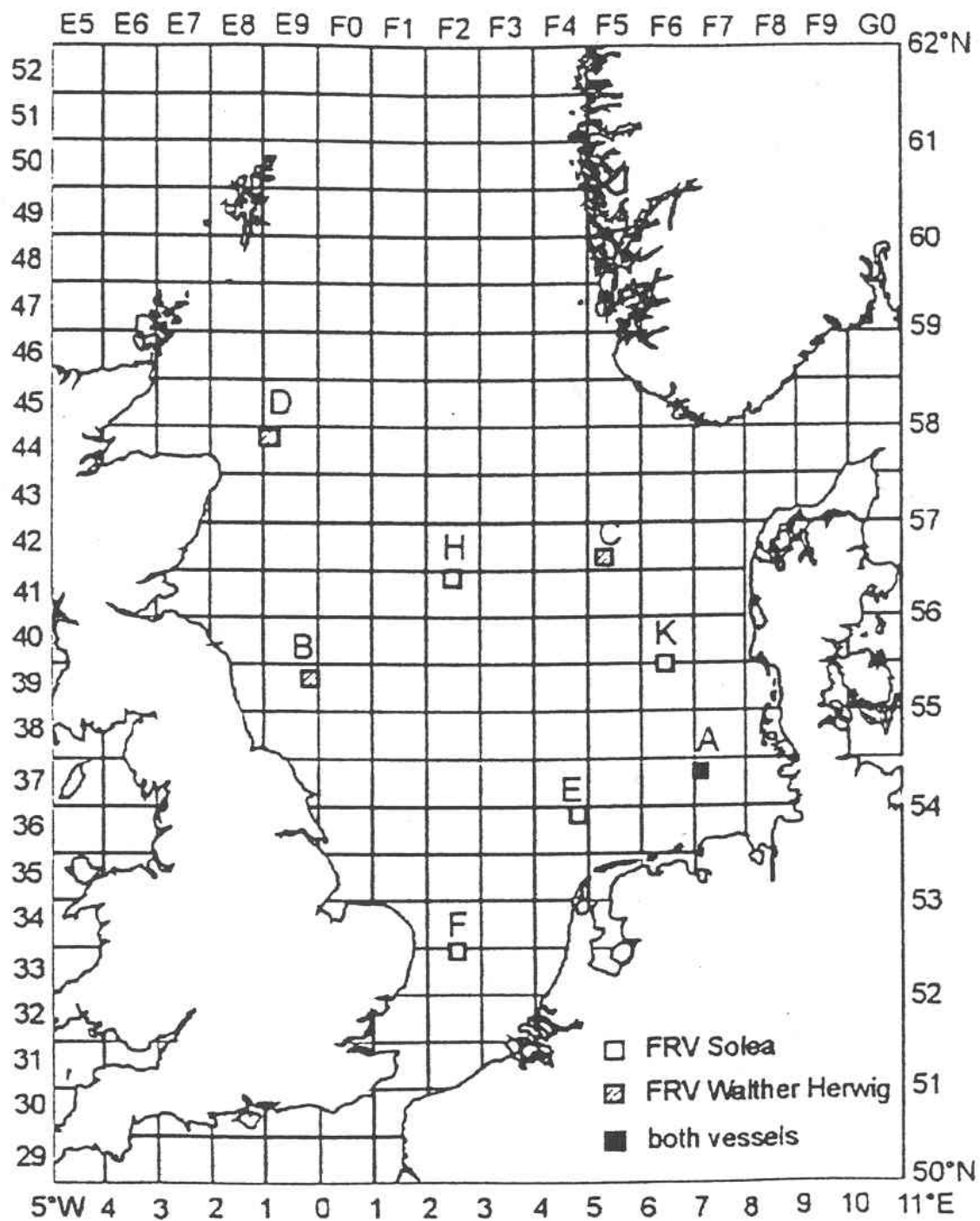
Figure 8.2.1.1.1.1. Position of the 8 boxes in the North Sea.
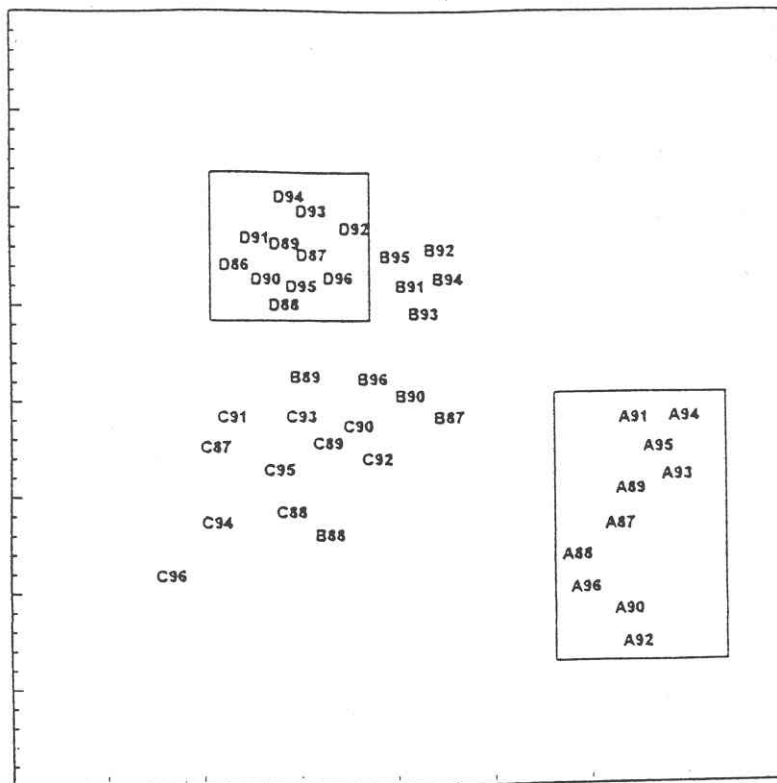
Figure 8.2.1.1.1.2 MDS plot of similarities (Bray–Curtis index) within and between the boxes A, B, C, and D.
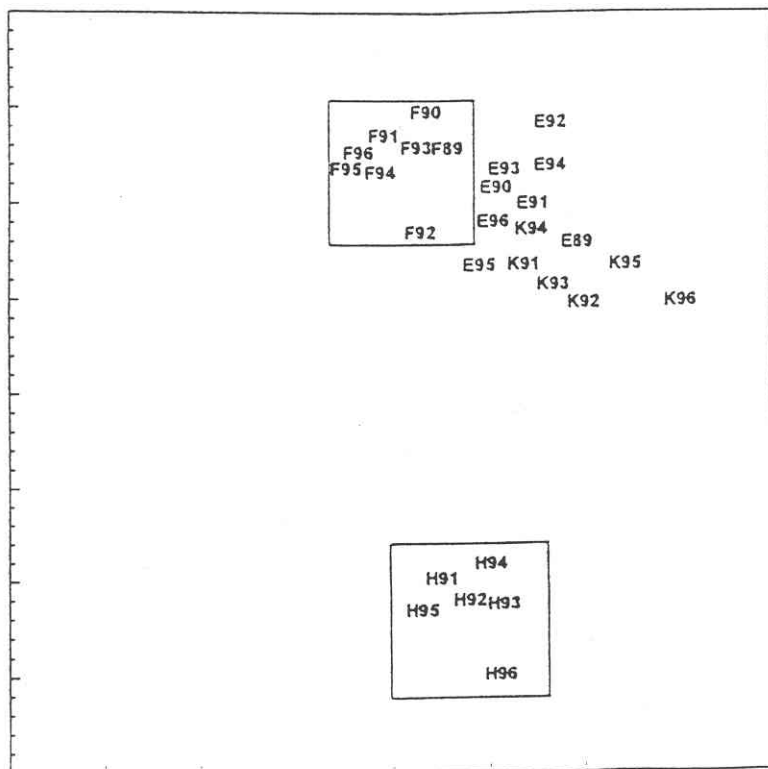
Figure 8.2.1.1.1.3 MDS plot of similarities (Bray–Curtis index) within and between the boxes E, F, H, and K.
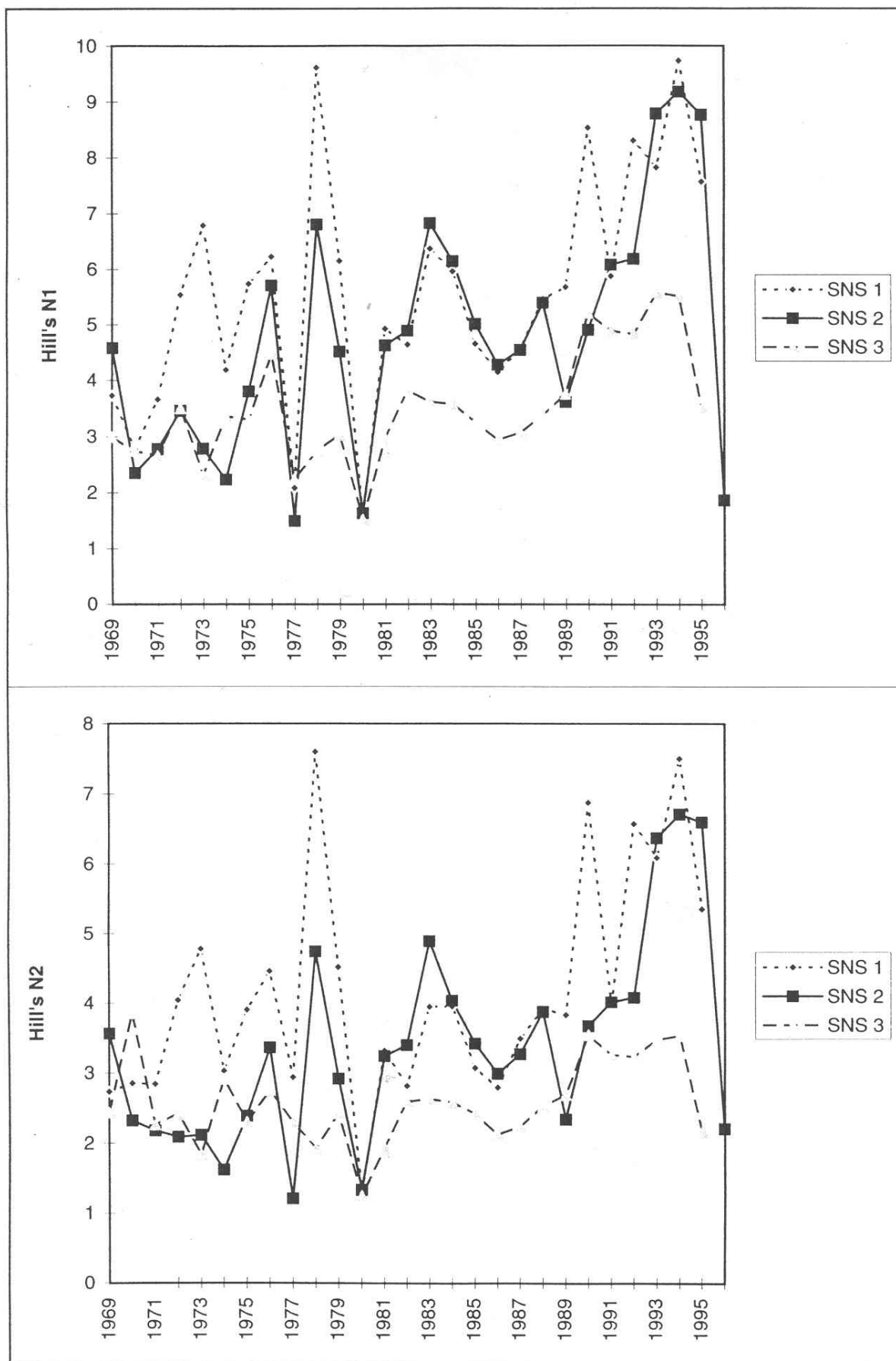
Figure 8.2.1.2.1.1
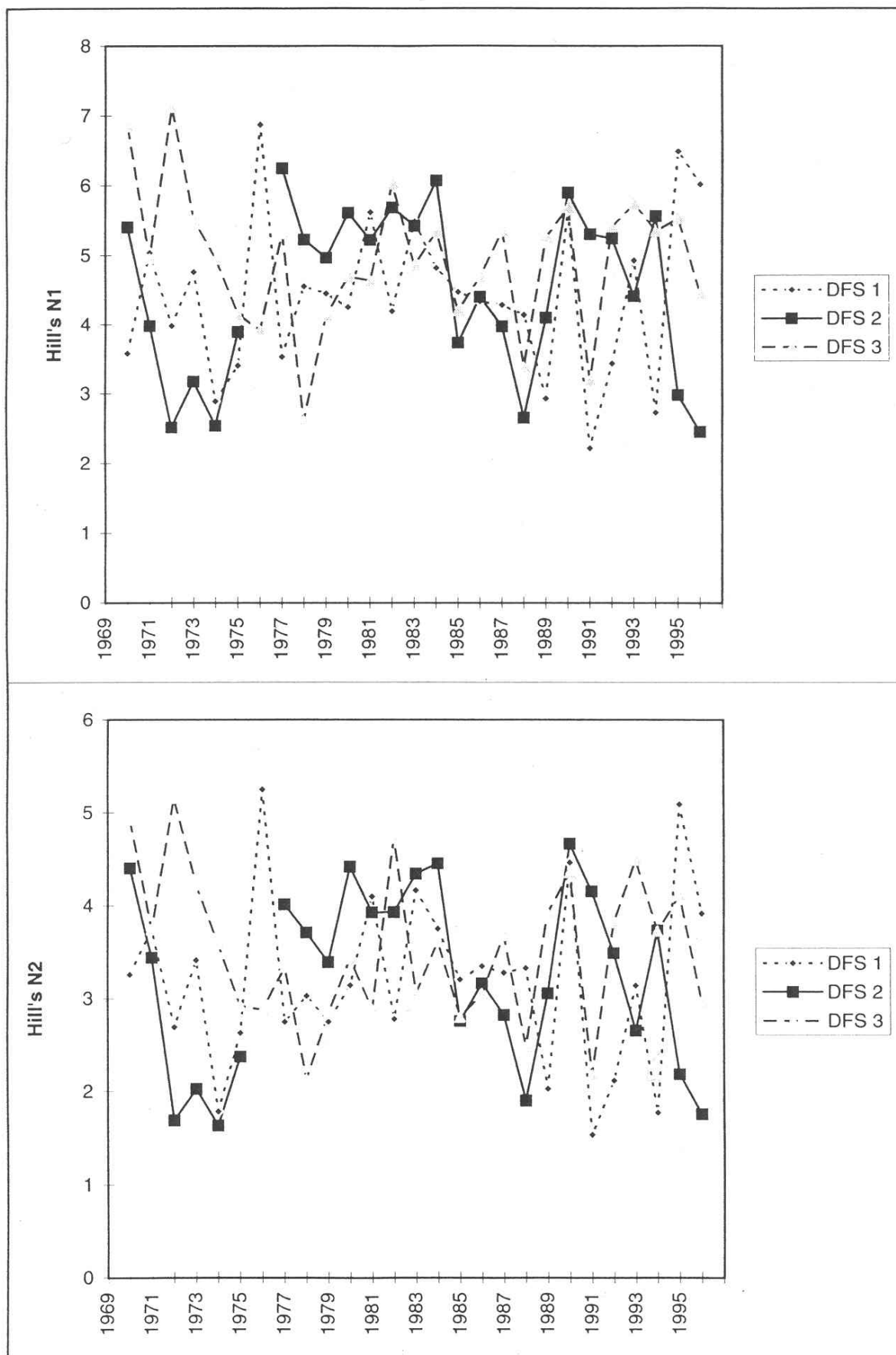
Figure 8.2.1.2.1.2.
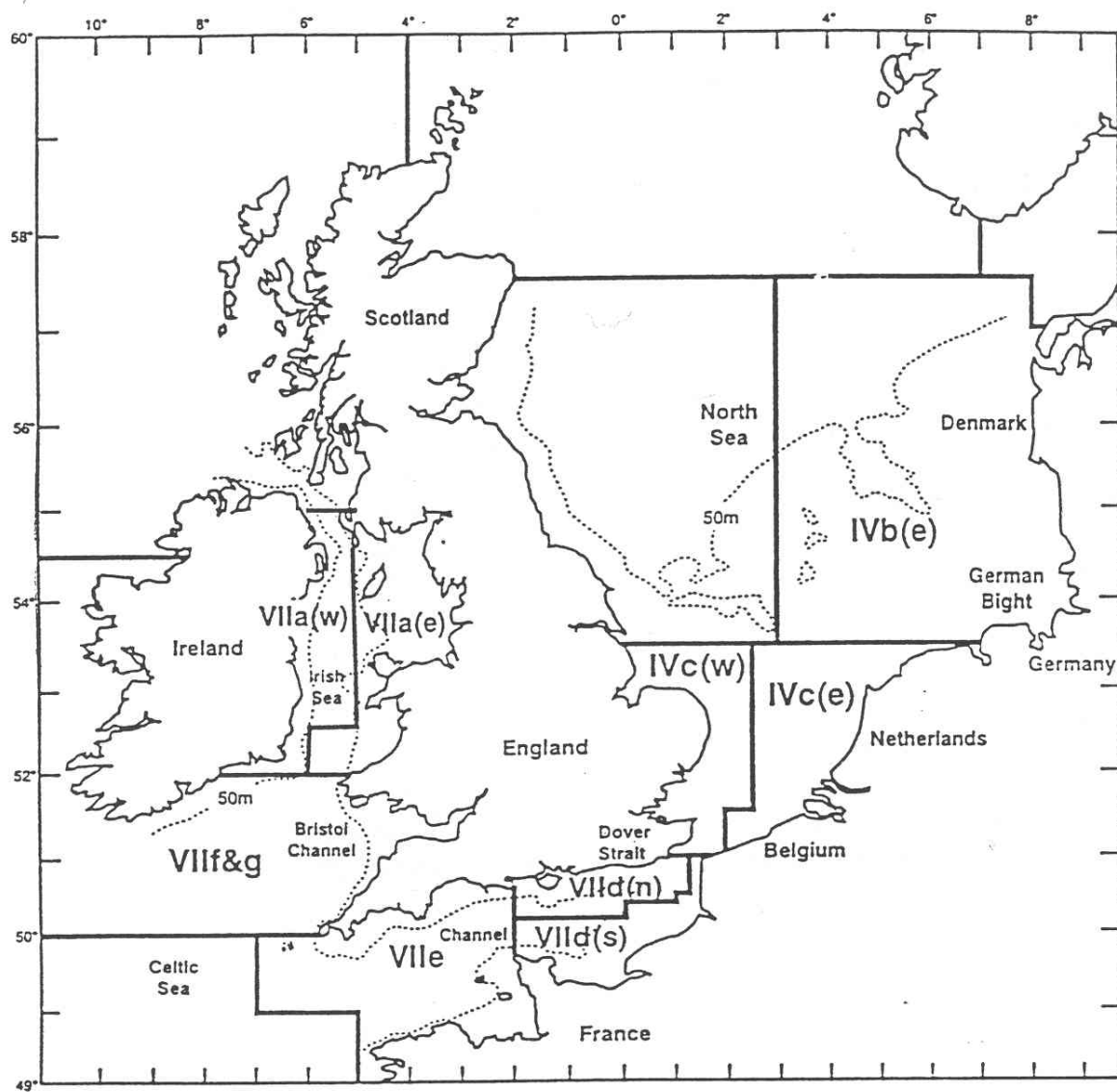
Figure 8.2.2.1.1.1.

Figure 8.2.2.1.2.1.

Oct 1985

41

39

37

Portugal

10   9   8

▦ Northern Shallow   ▦ Intermediate   ▦ Southern Shallow

▦ Northern Deep   ▦ Southern Deep   ■ Non-trawlable area
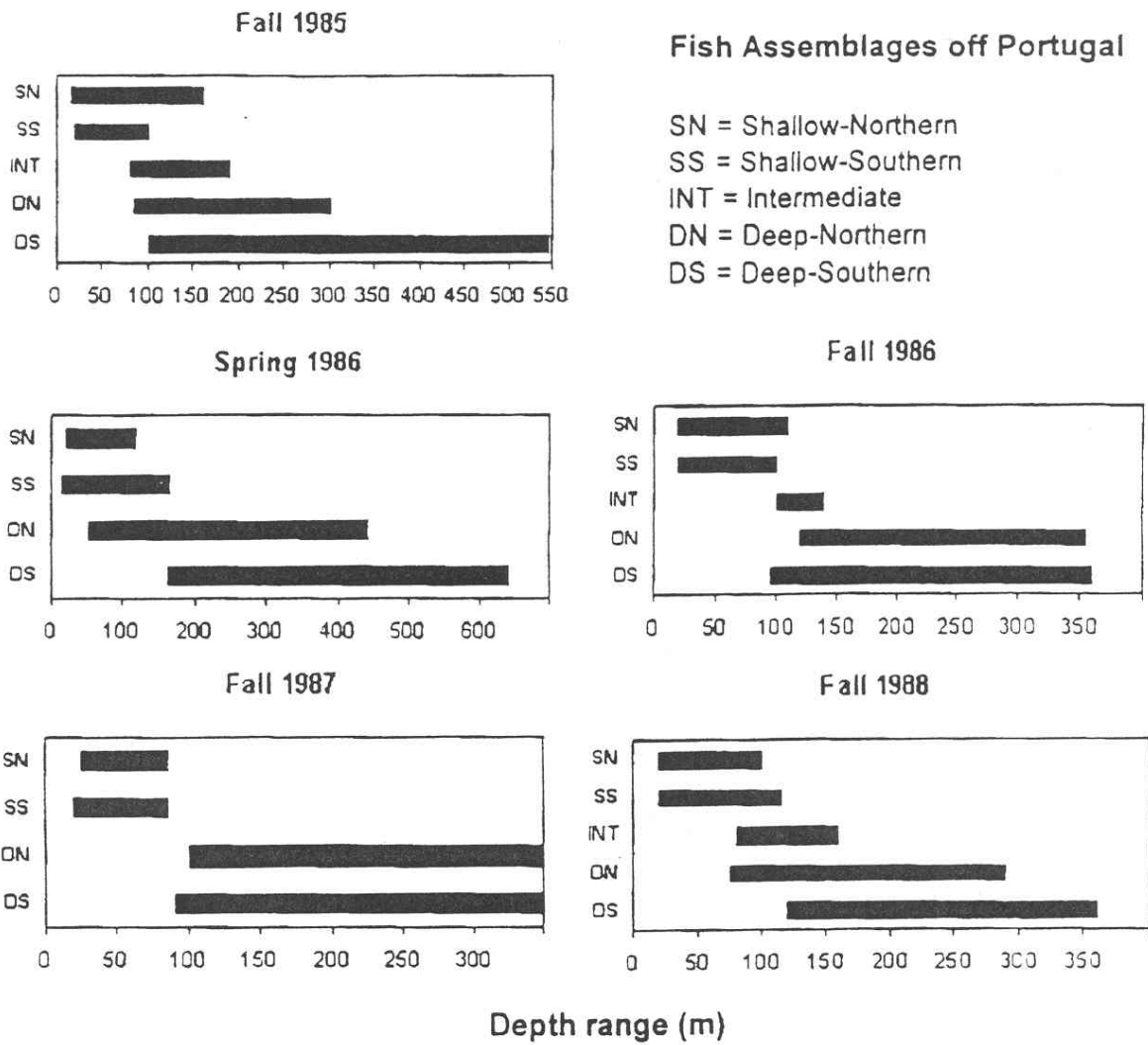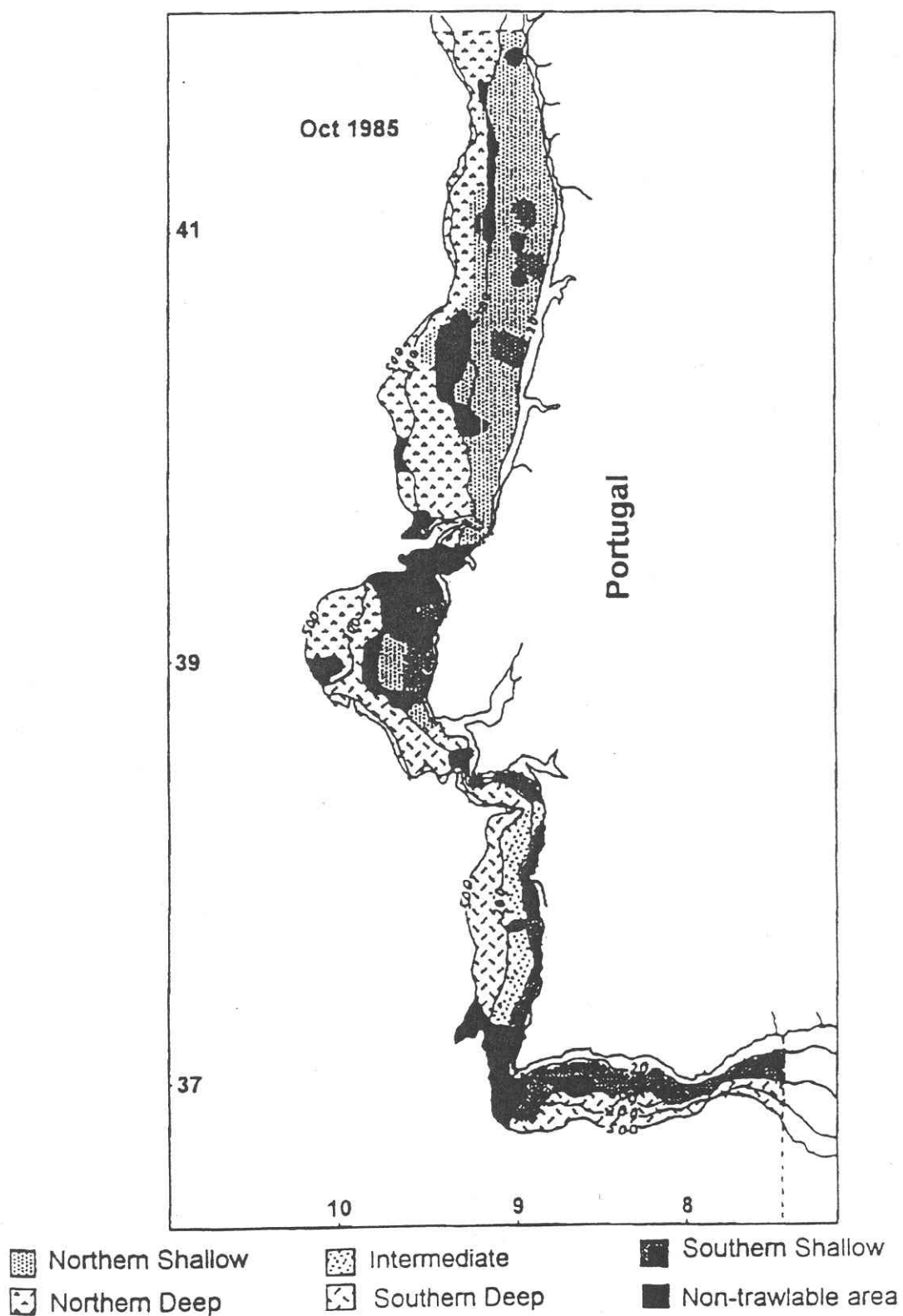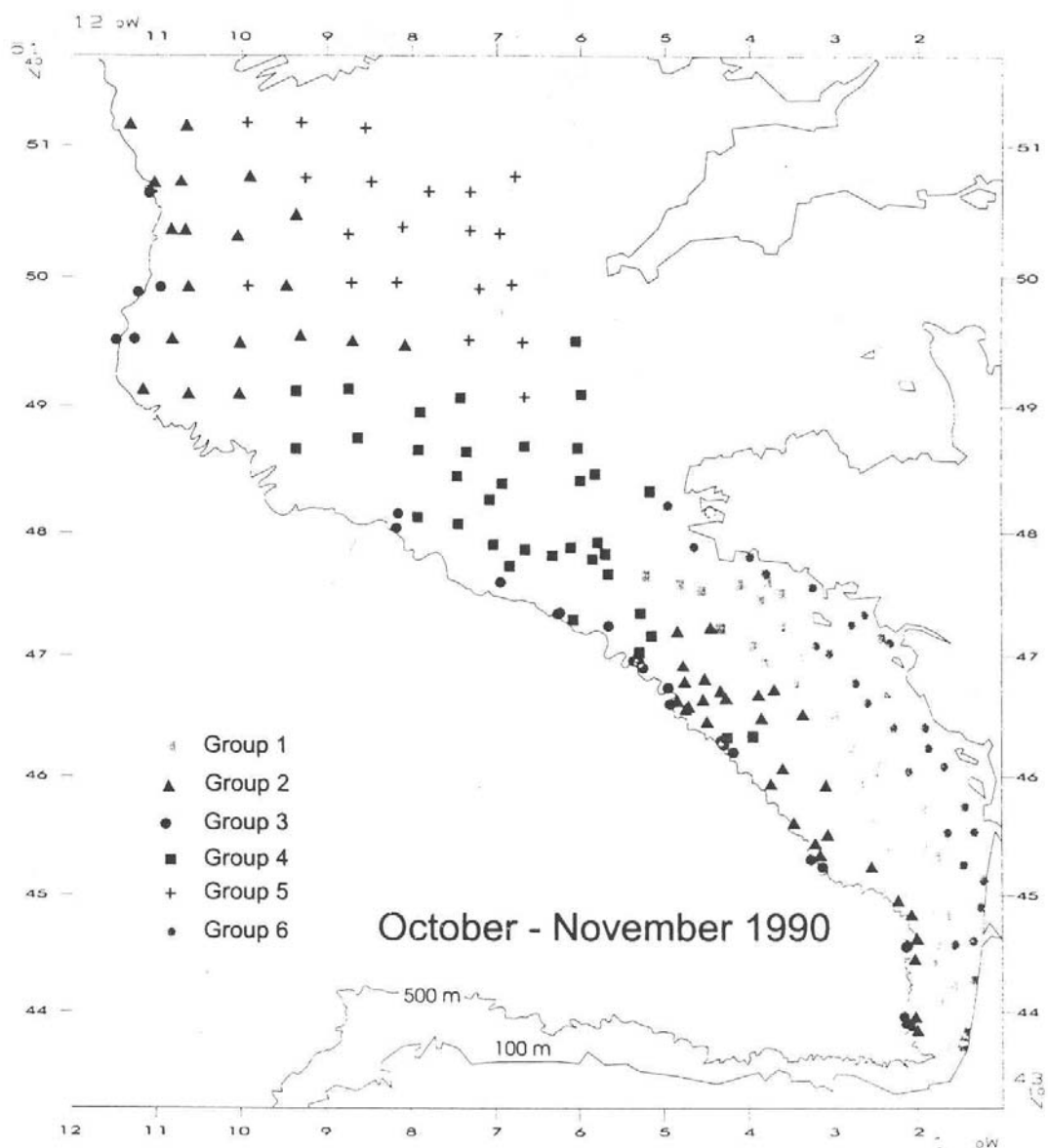
Figure 8.2.2.1.2.2.

Figure 8.2.2.1.3.1.
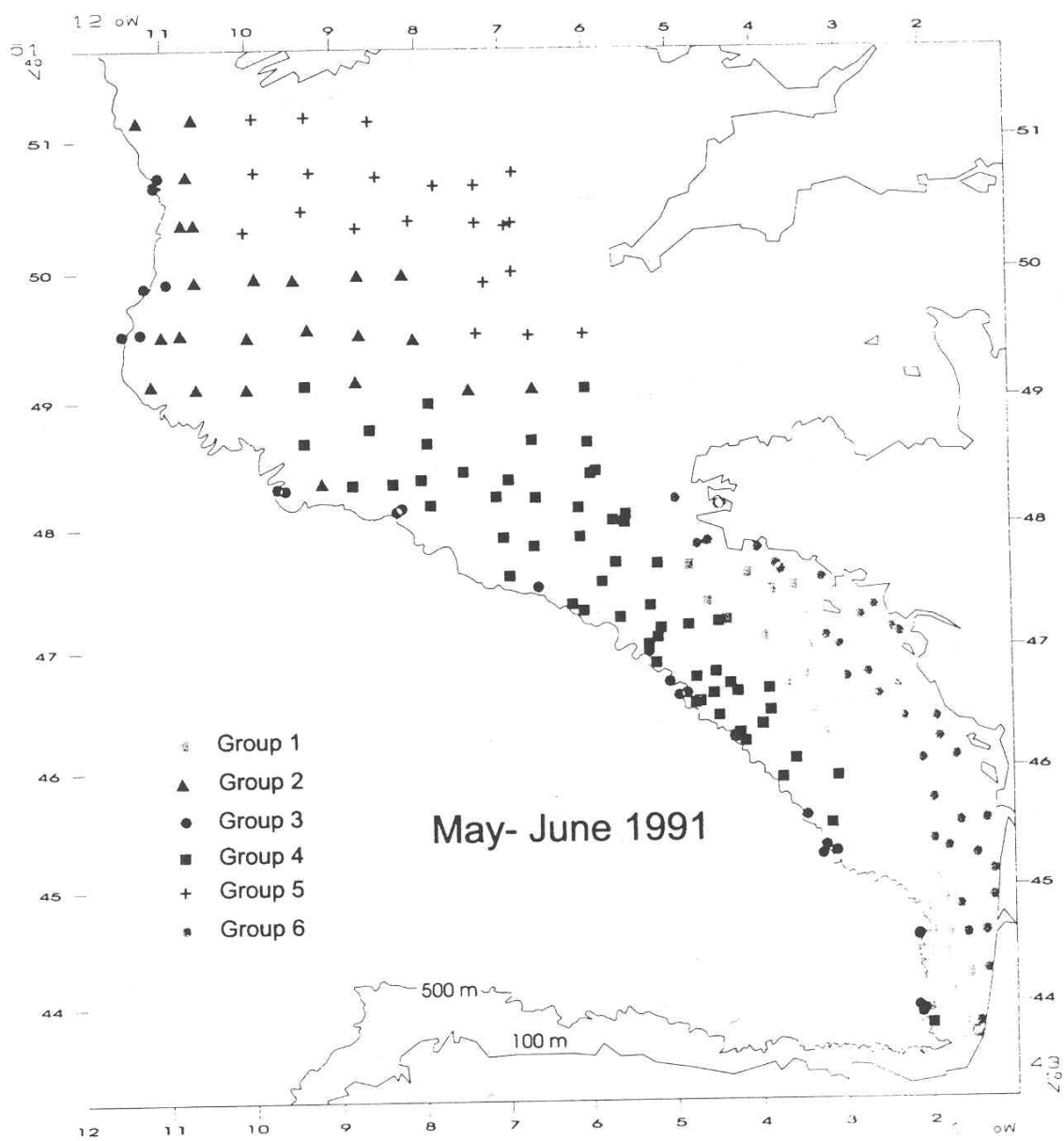
Figure 8.2.2.1.3.2.

Demersal species only
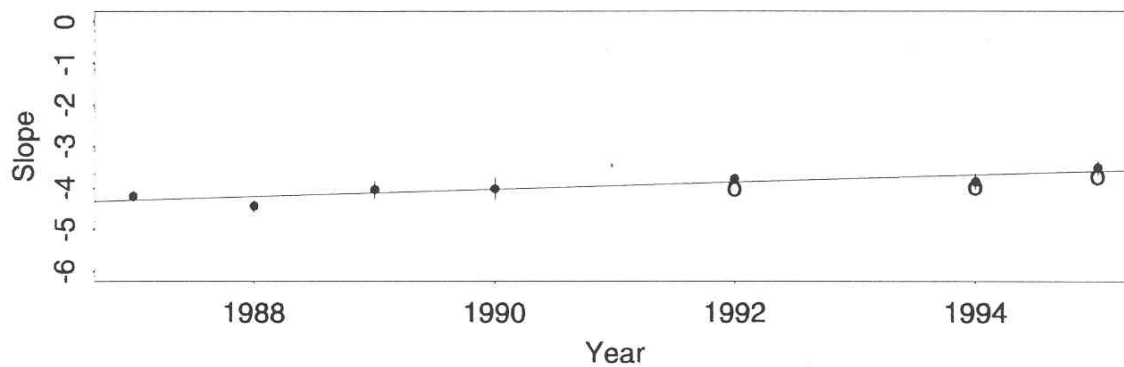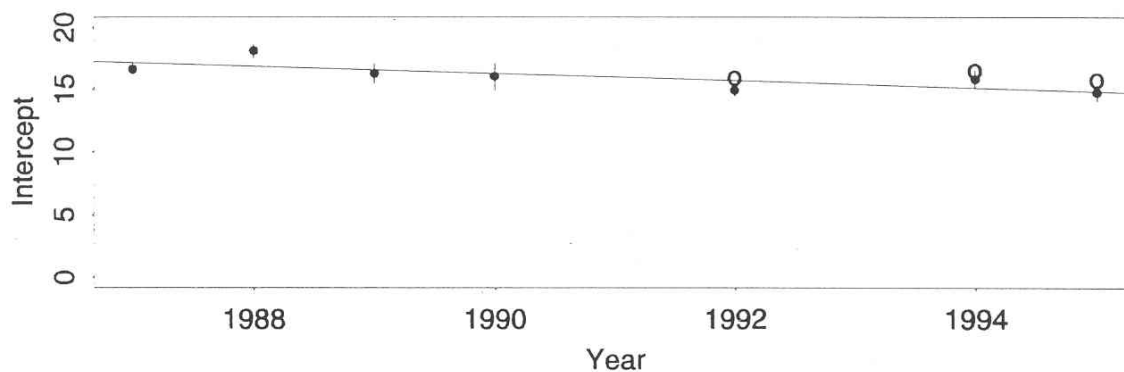
Figure 8.2.2.1.4.1 Patterns of slopes (upper) and intercepts (lower) ± standard error over years from annual regressions of ln(numbers) on ln(length) for demersal species only. Open circles: all species; dots: set of selected species.
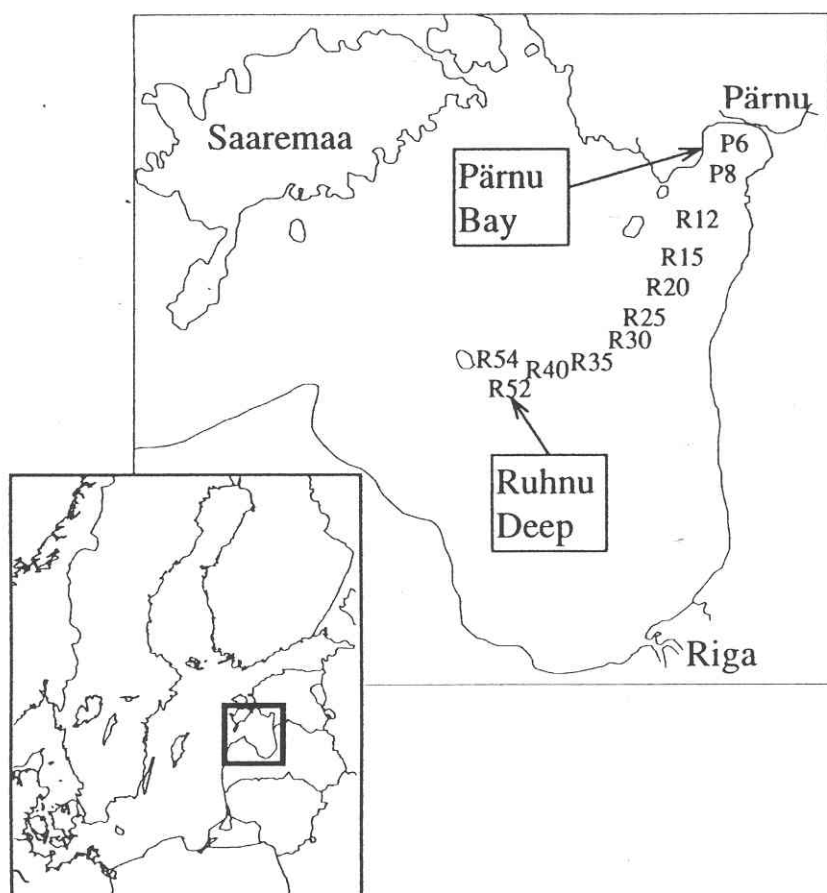
Figure 8.2.2.1.5.1 Location of trawling stations in the Gulf of Riga. The number indicate depth of a given station.
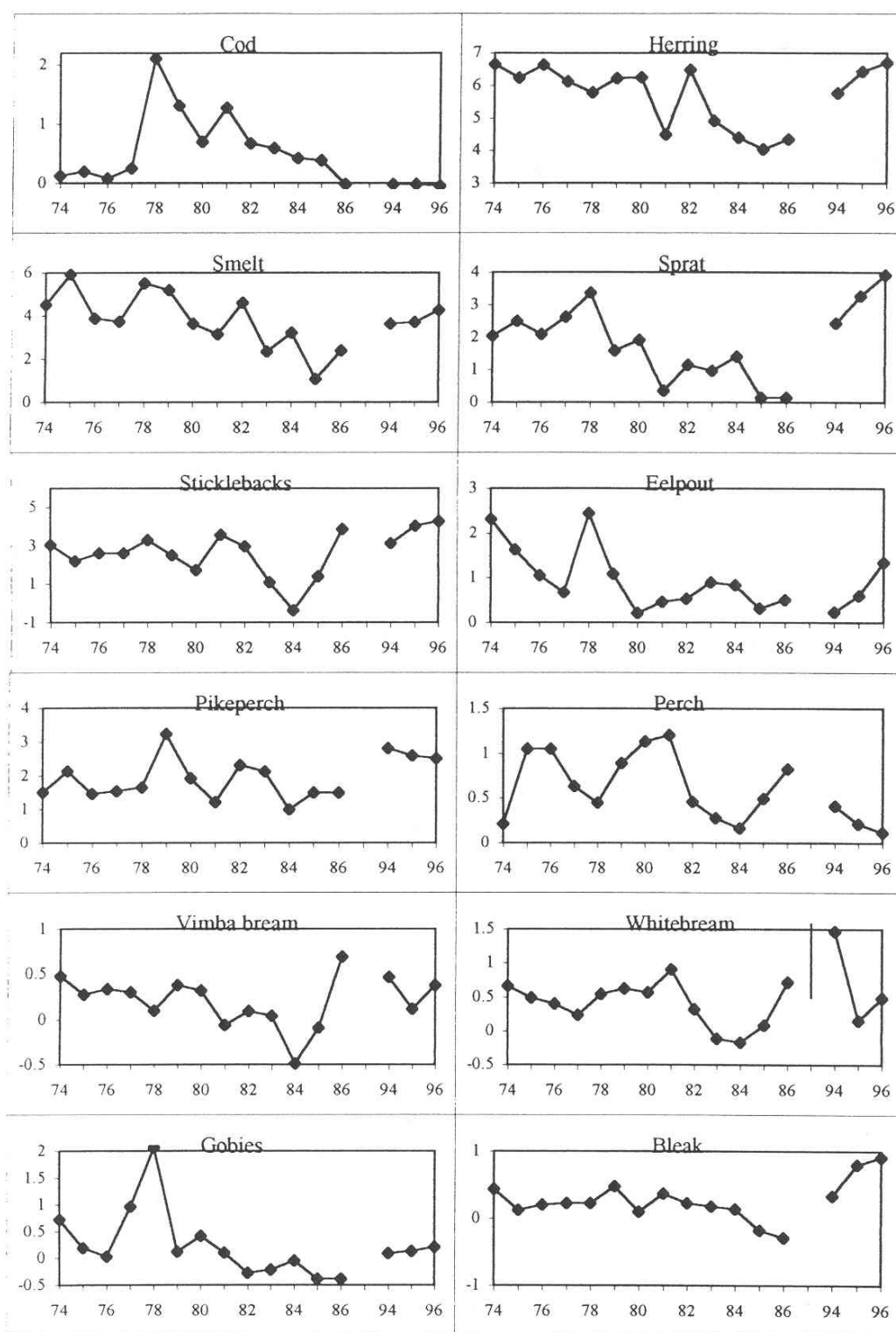
Figure 8.2.2.1.5.2   Dynamics of the abundance-based year-effect with the least significant difference (LSD) bar for the most abundant fish species in the NE Gulf of Riga over the years 1974–1986 and 1994–1996.
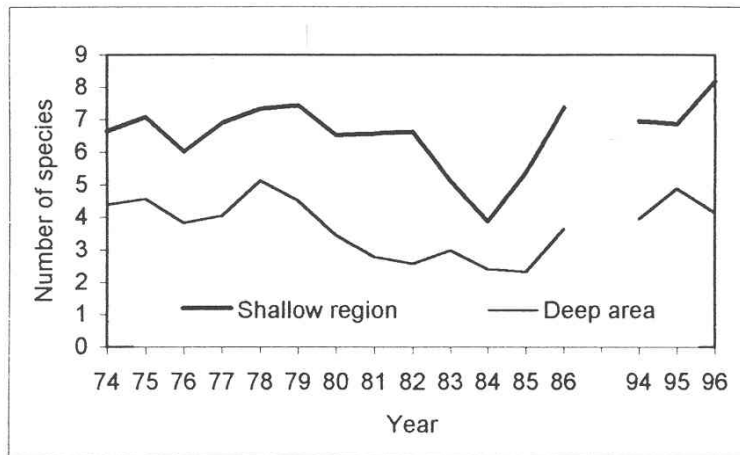
Figure 8.2.2.1.5.3 Dynamics of the mean number of fish species present in experimental bottom trawl catches in the shallow and deep areas during 1974–1986 and 1994–1996.
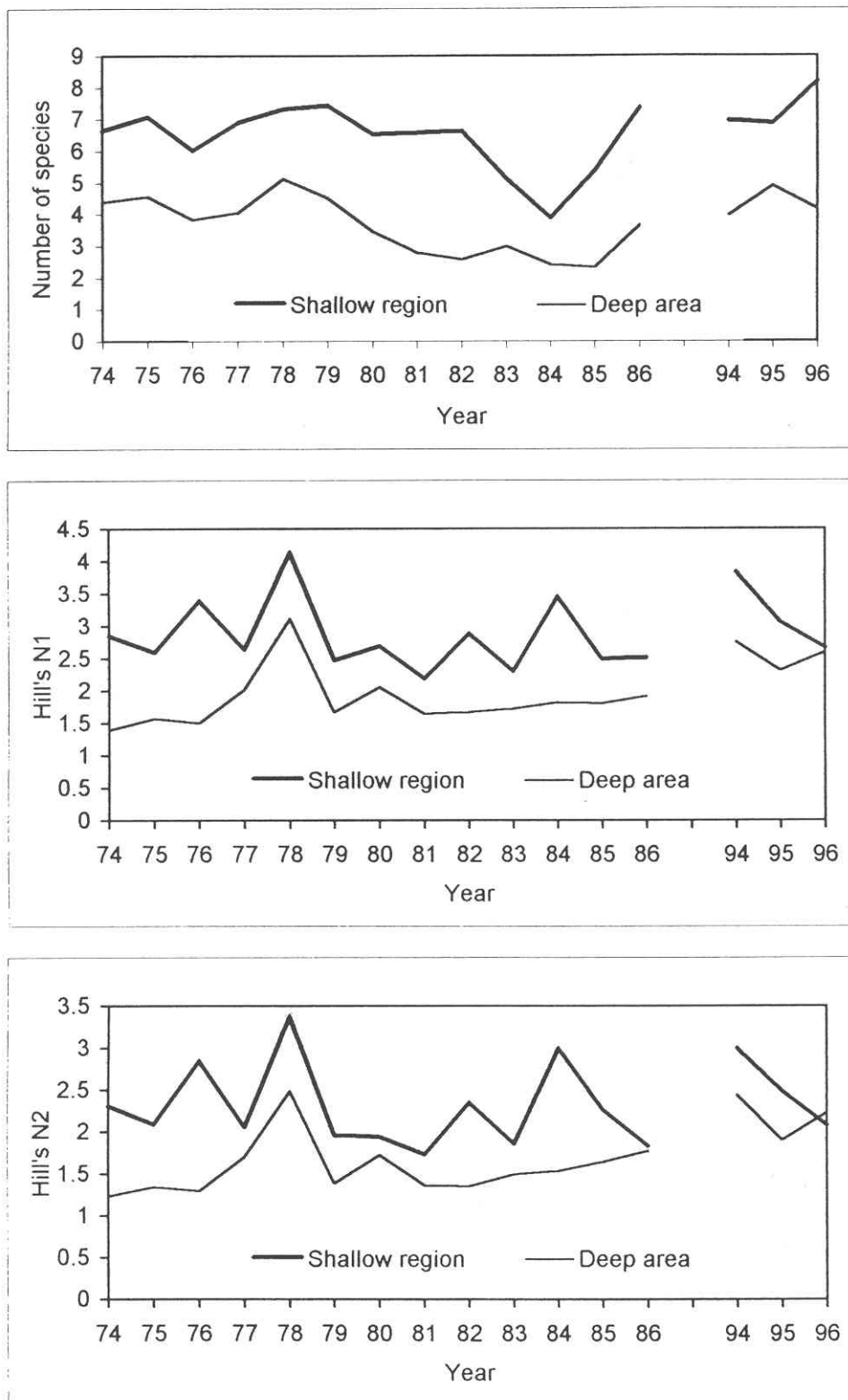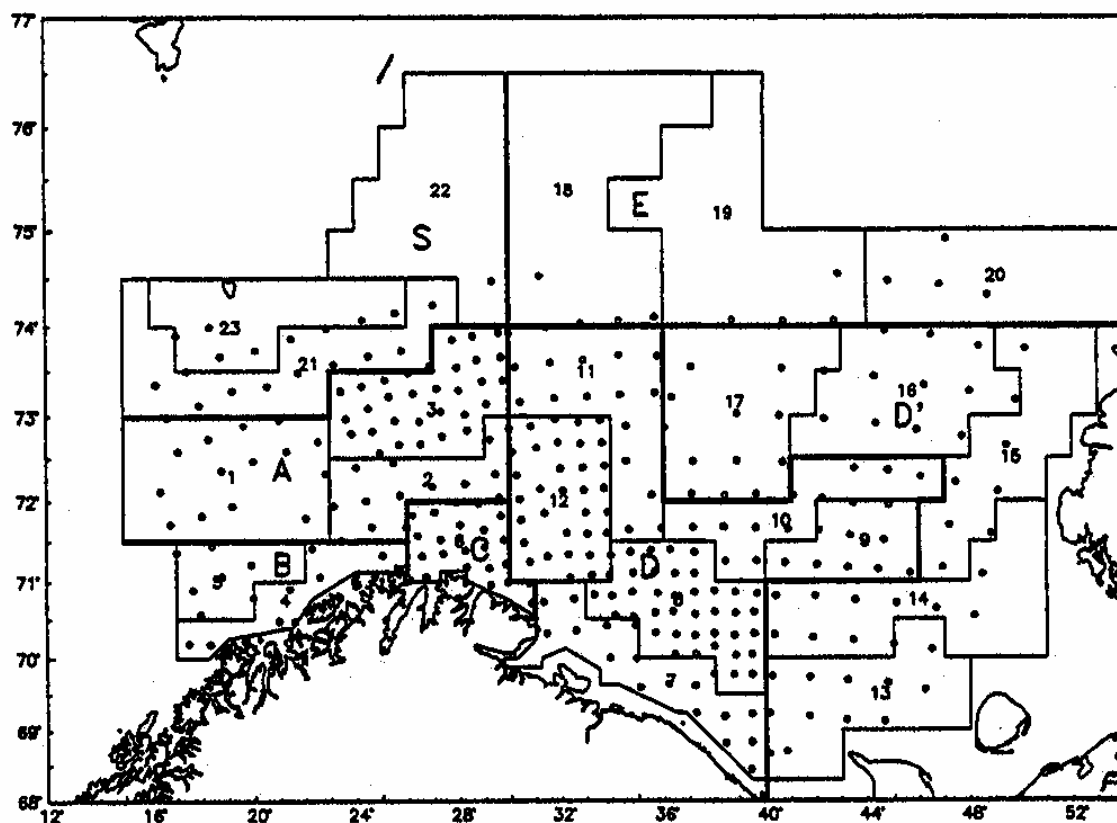
Figure 8.2.2.2.1.1

Figure 8.2.2.2.2.1 (Barents Sea case study). The survey area with subareas (A,B,C,D,D[1],E, and S) and strata used in the bottom trawl survey.
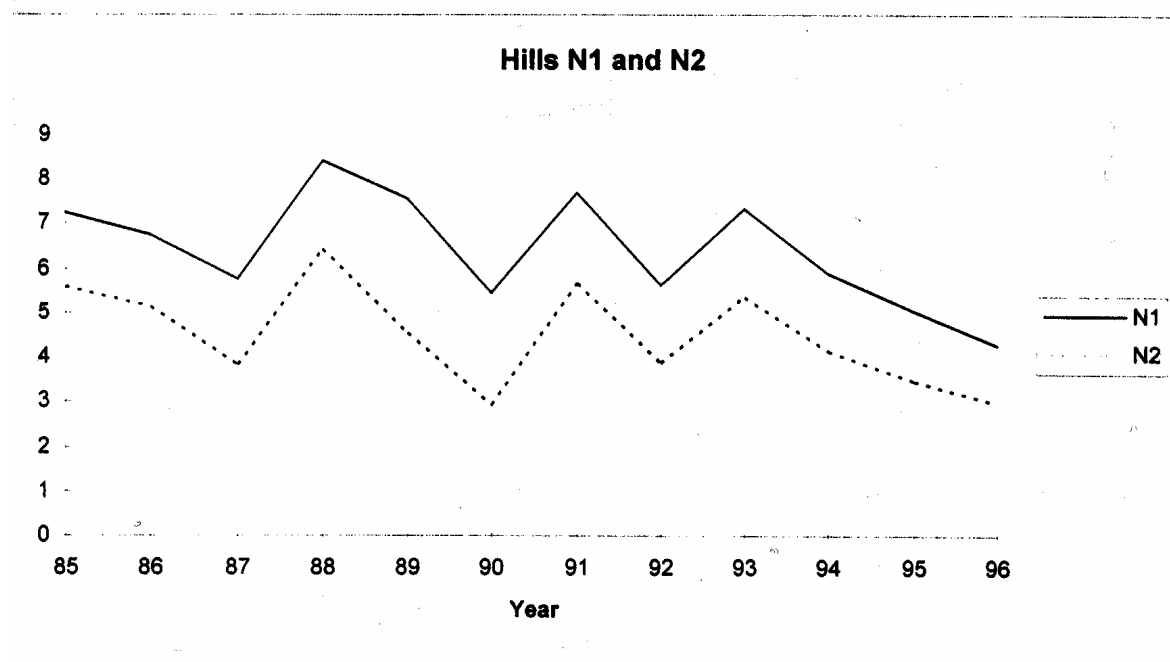
Figure 8.2.2.2.2.2 Hills N1 and N2 calculated from a series of 12 years of the Norwegian bottom trawl survey in the Barents Sea.
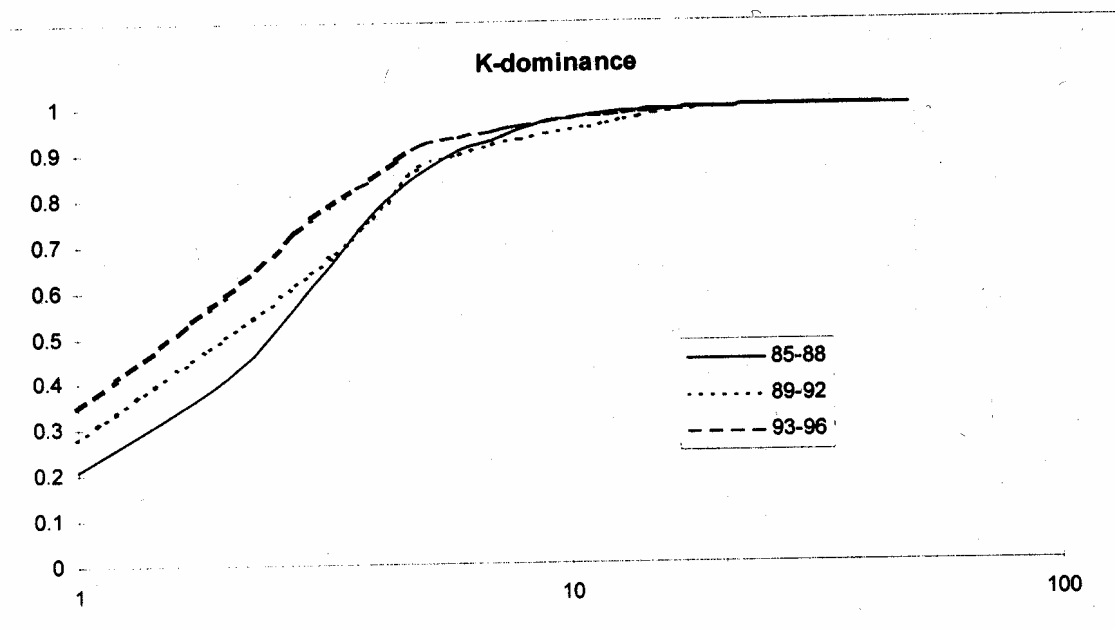
K-dominance

Figure 8.2.2.2.2.3   K-dominance curves. Barents Sea bottom trawl survey data.
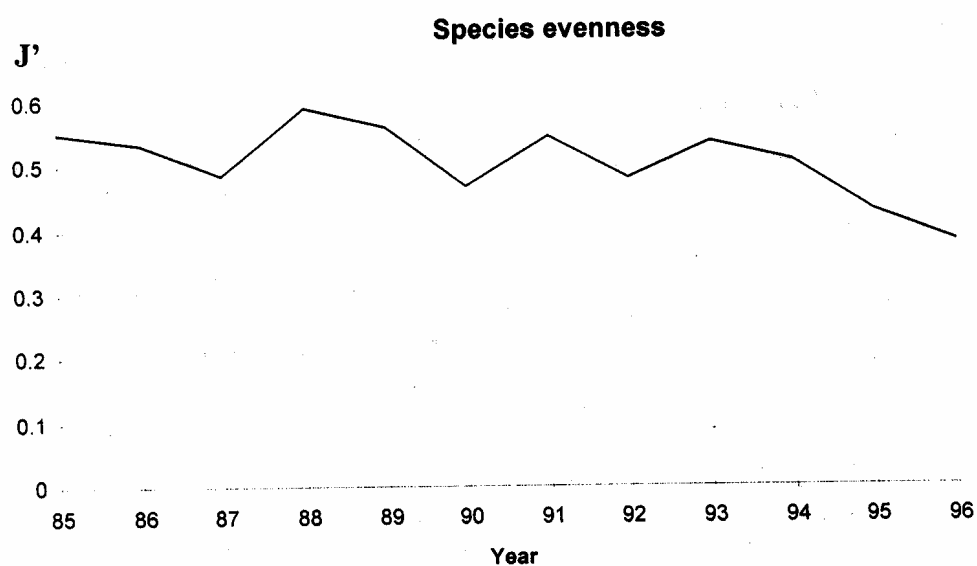
Species evenness

Figure 8.2.2.2.2.4   Trend in species evenness for Barents Sea bottom trawl survey.
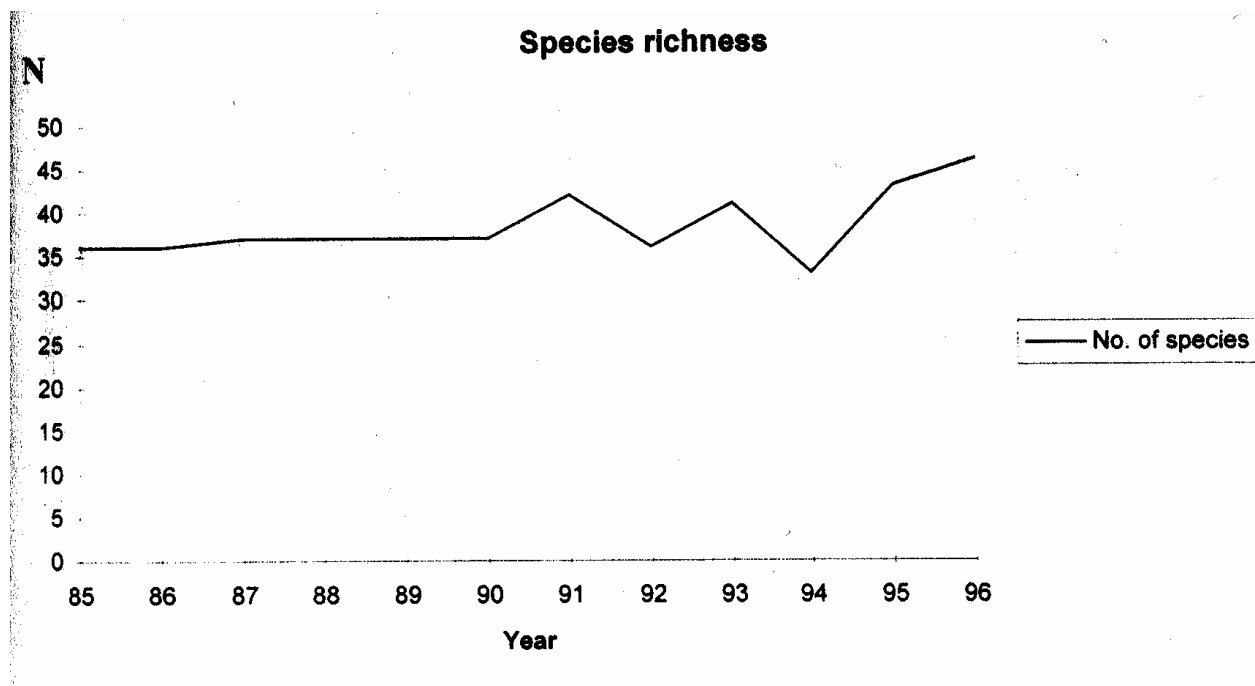
Figure 8.2.2.2.2.5   Trend in species richness for Barents Sea bottom trawl survey.
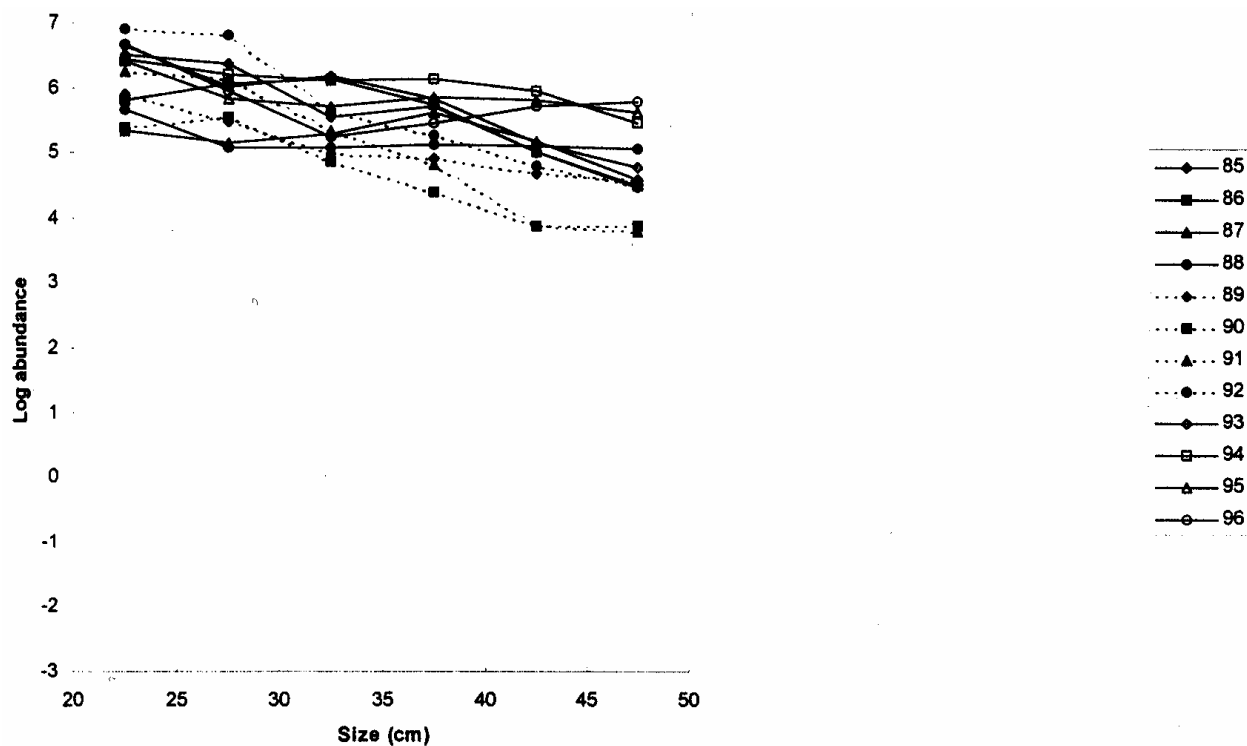
Figure 8.2.2.2.2.6   Size spectra for 12 surveys for lengths between 20 cm and 50 cm.
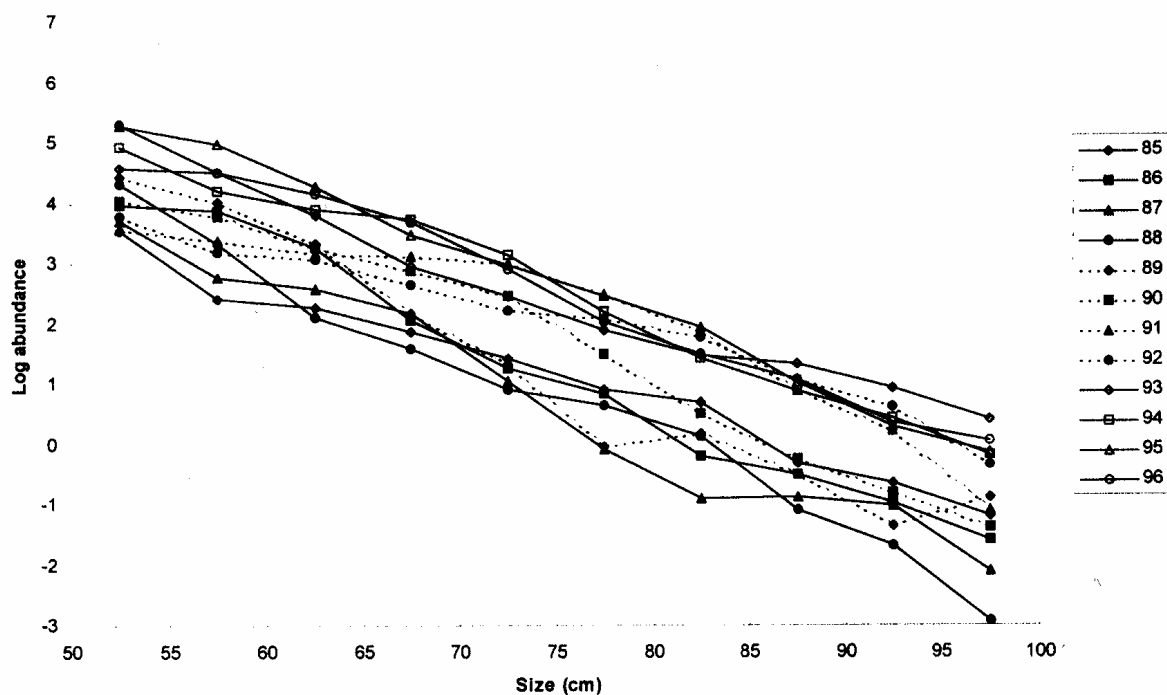
Figure 8.2.2.2.2.7 Size spectra from 12 surveys for lengths between 50 cm and 100 cm.
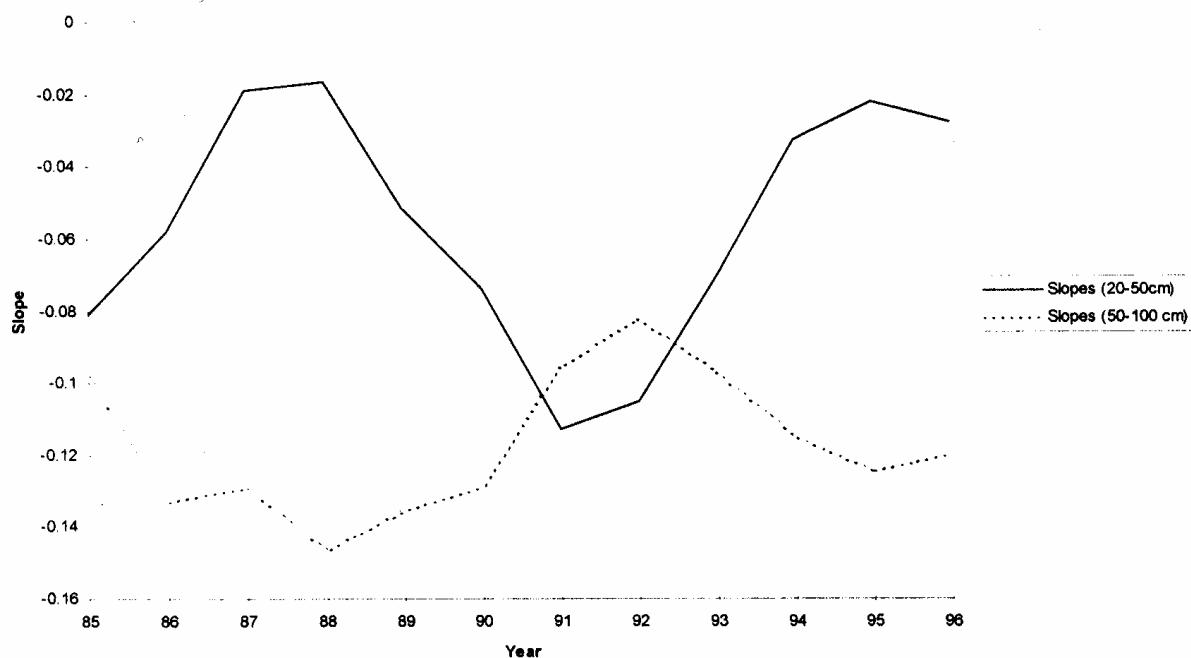
Figure 8.2.2.2.2.8 Comparison of trends in the yearly slopes of the size spectra for the length groups 20 cm to 50 cm and 50 cm to 100 cm.
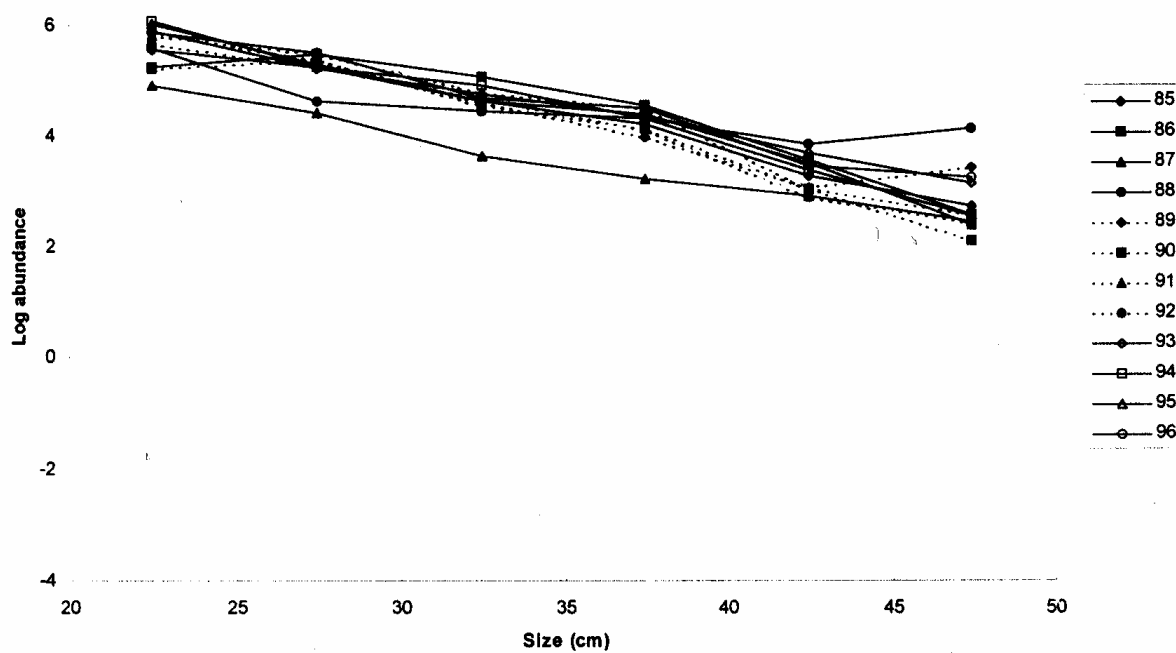
Figure 8.2.2.2.2.9 Yearly size spectra from 12 surveys for lengths between 20 cm and 50 cm excluding data for cod and haddock.
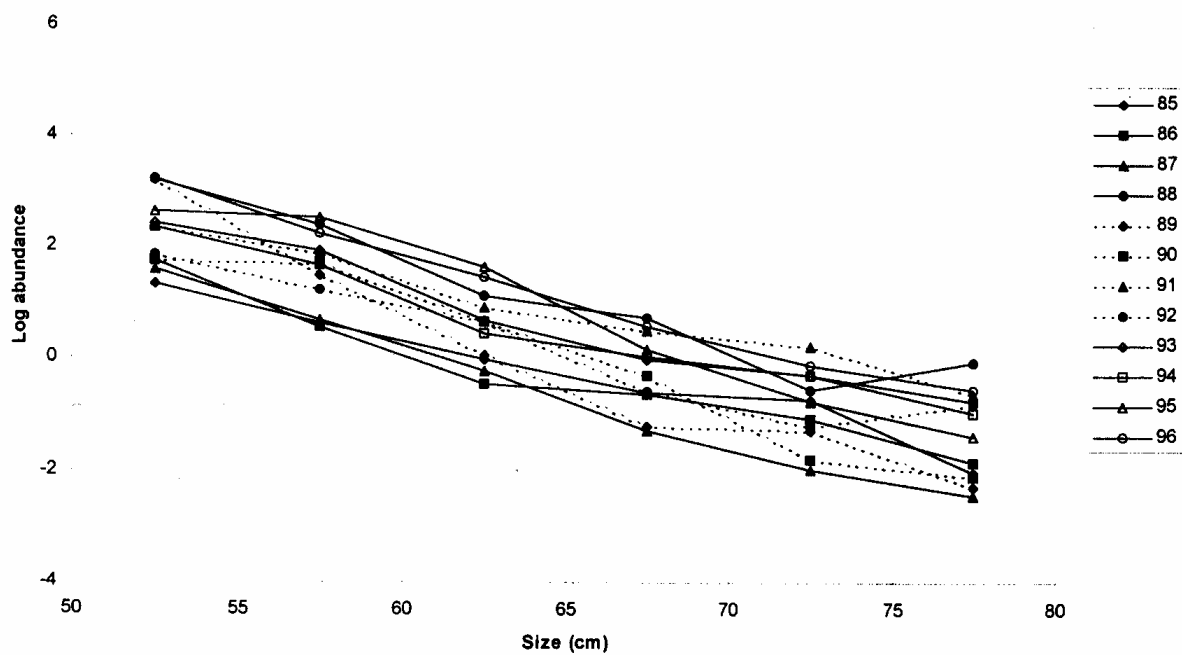
Figure 8.2.2.2.2.10 Yearly size spectra from 12 surveys for lengths between 50 cm and 80 cm excluding data for cod and haddock.
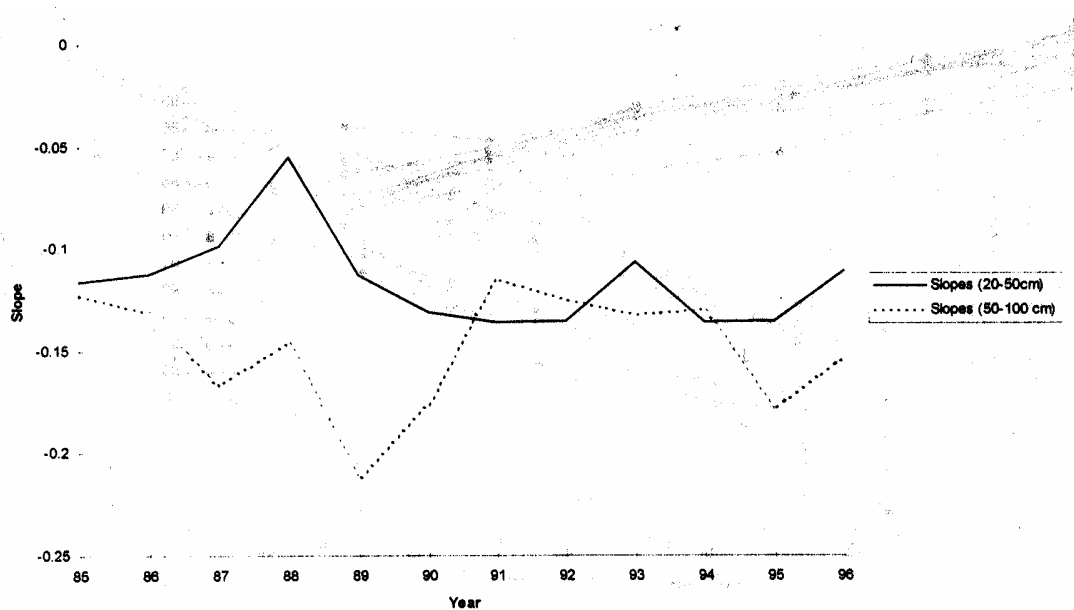
Figure 8.2.2.2.2.11 Comparison of trends in the yearly slopes of the size spectra for the length groups 20 cm to 50 cm and 50 cm to 80 cm with data for cod and haddock excluded.
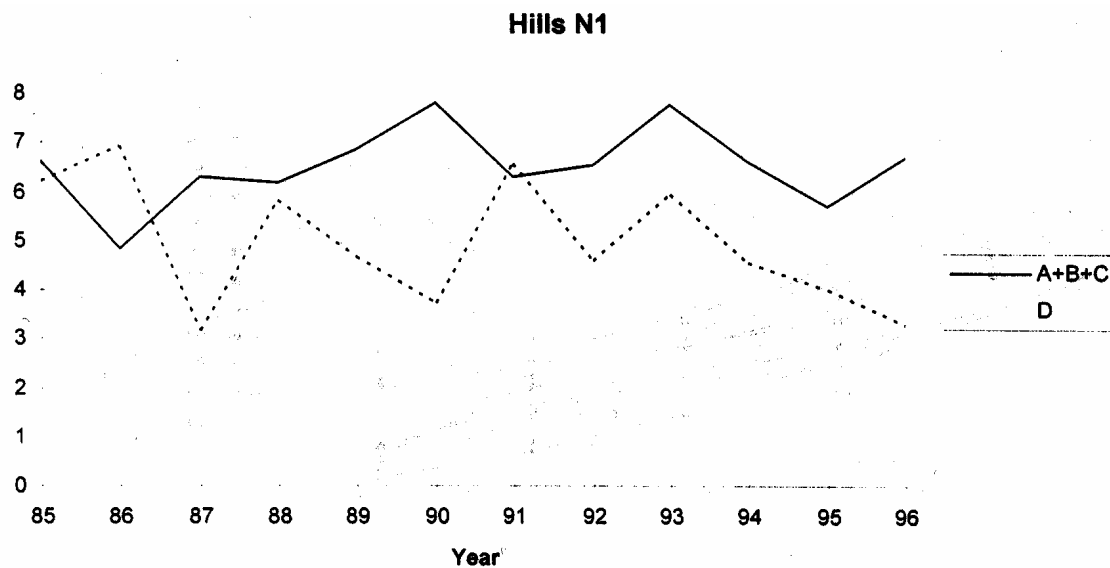
Figure 8.2.2.2.2.12 (The Barents Sea case study). Hills N1 calculated for 2 different areas in the Barents Sea. A+B+C is the western subarea while D is the eastern subarea.
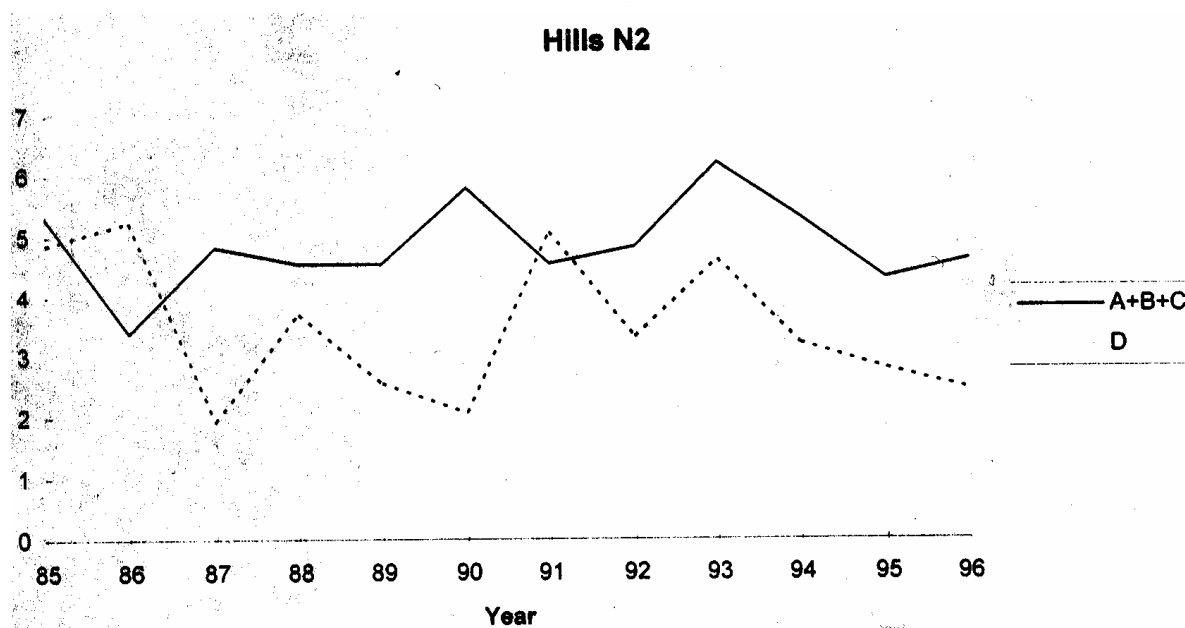
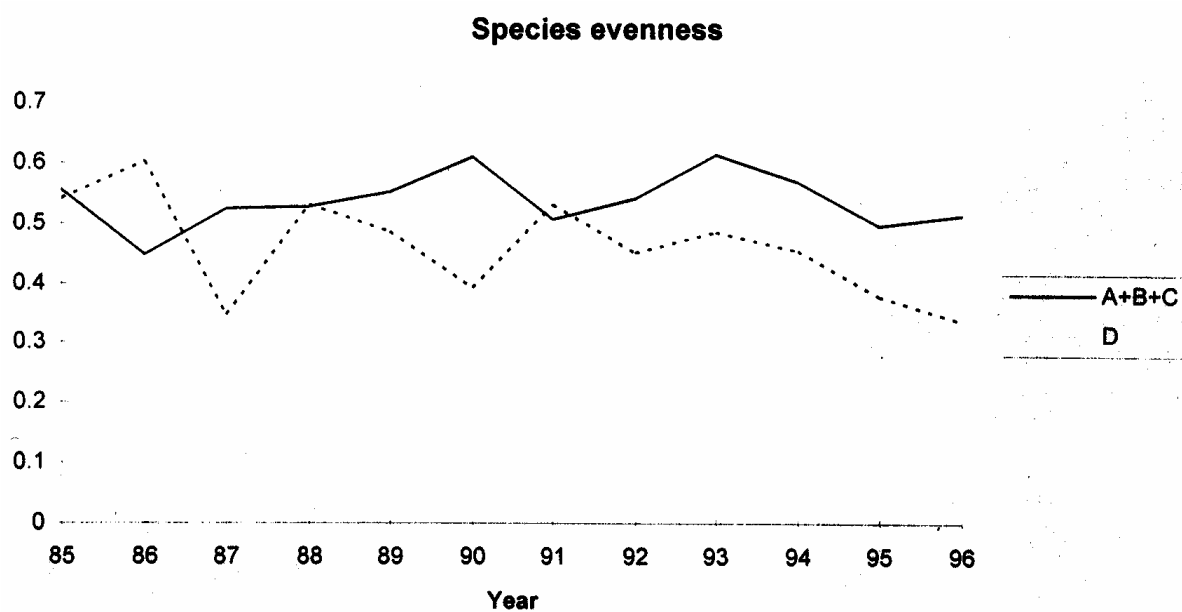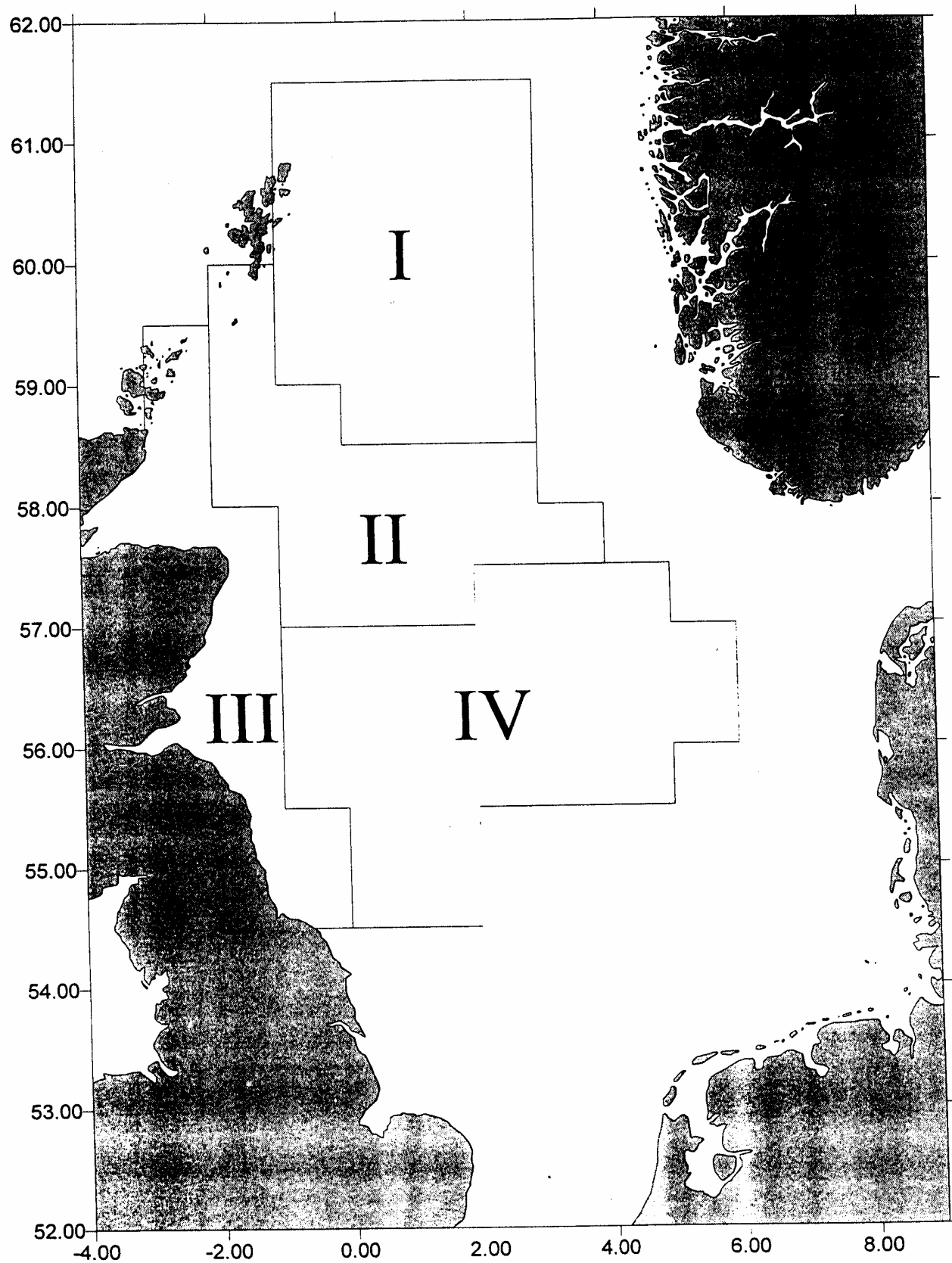**Figure 8.2.2.2.2.13** Hills N2 calculated for 2 different areas in the Barents Sea. A+B+C is the western subarea while D is the eastern subarea.

Figure 8.2.2.2.2.14.

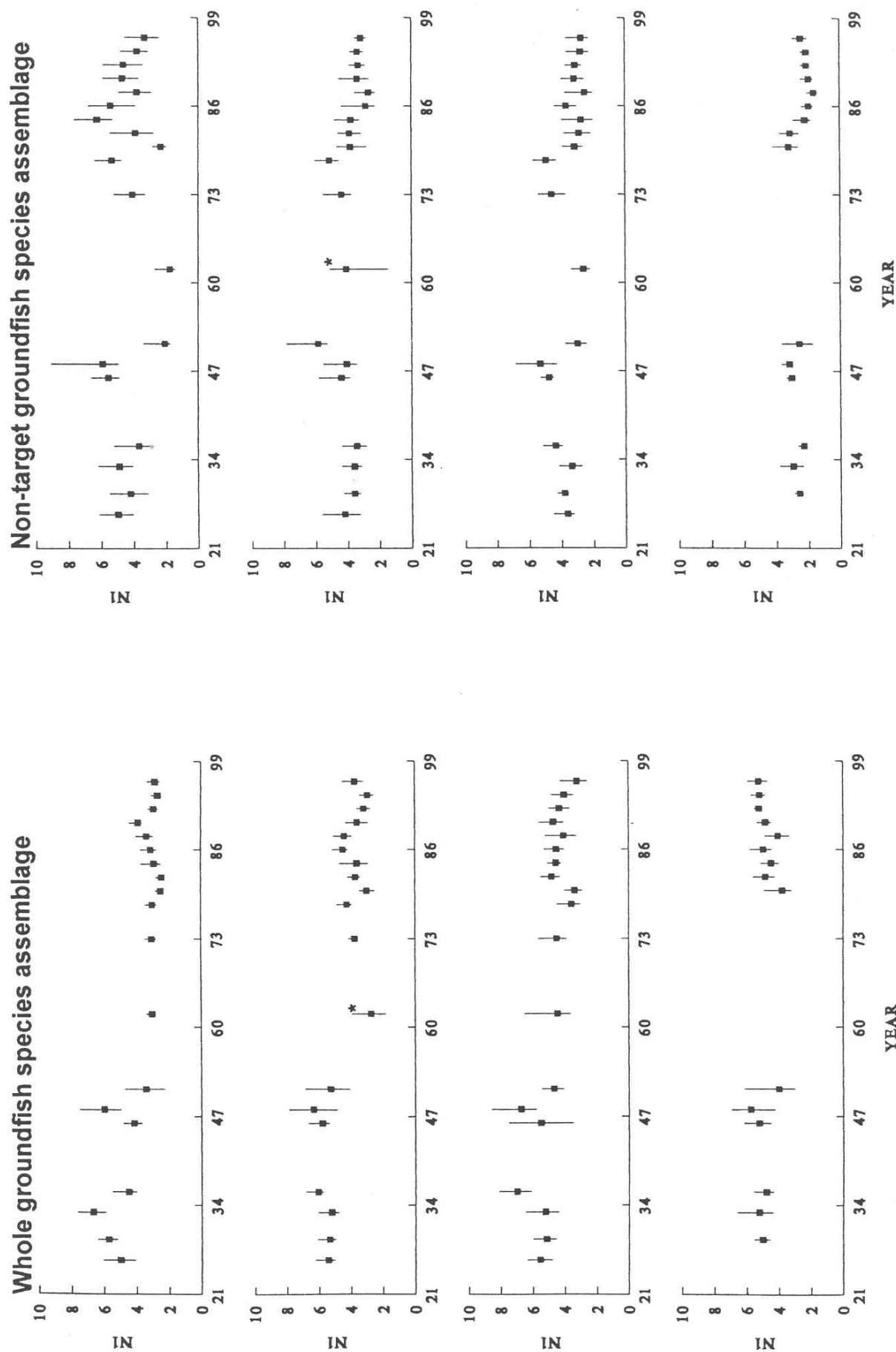**Figure 8.2.3.1.1.1**   Map of the North Sea showing four identified areas.

Non-target groundfish species assemblage

Whole groundfish species assemblage
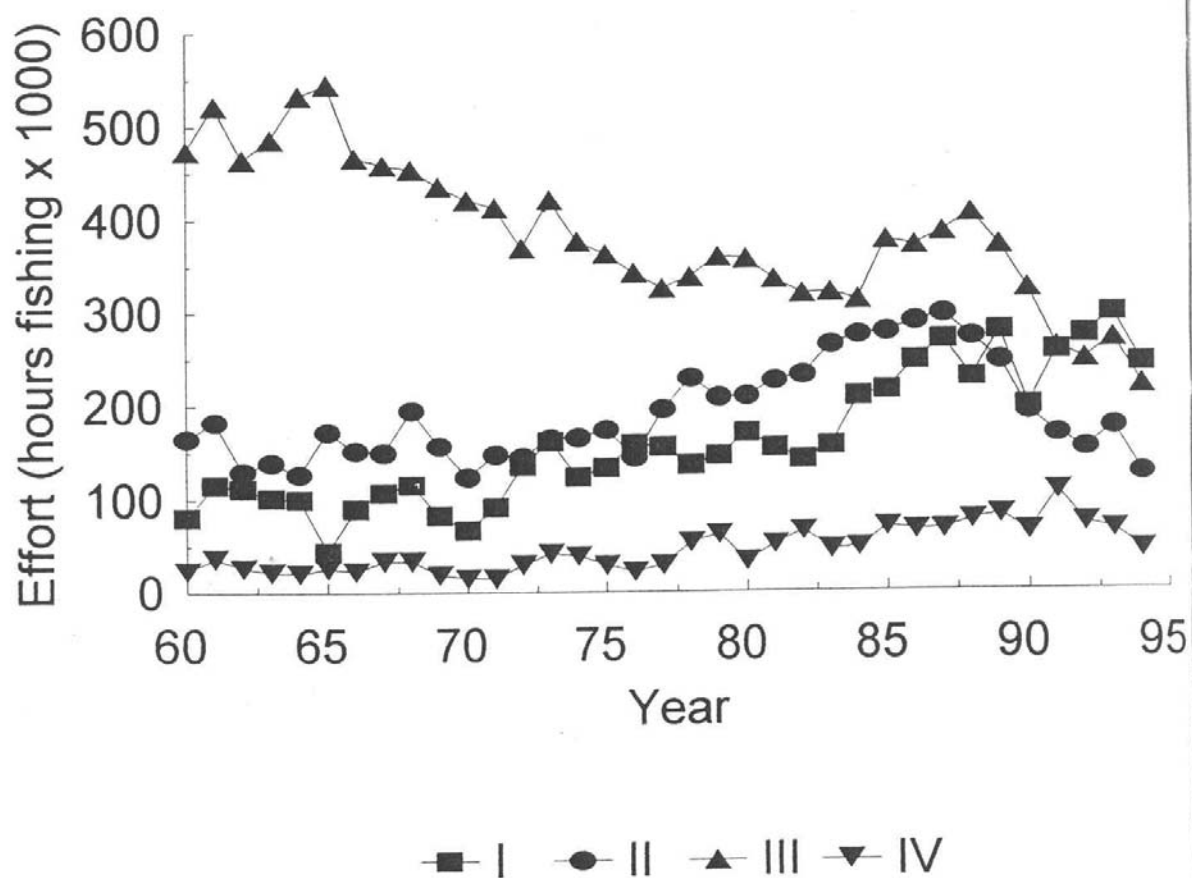
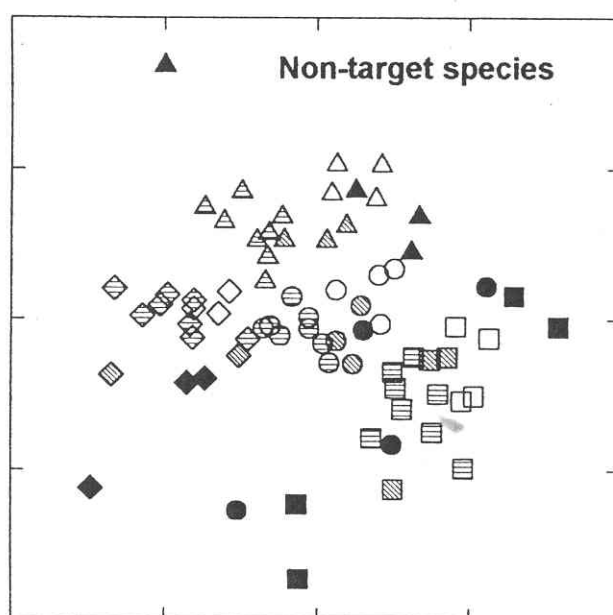**Figure 8.2.3.1.1.2**.
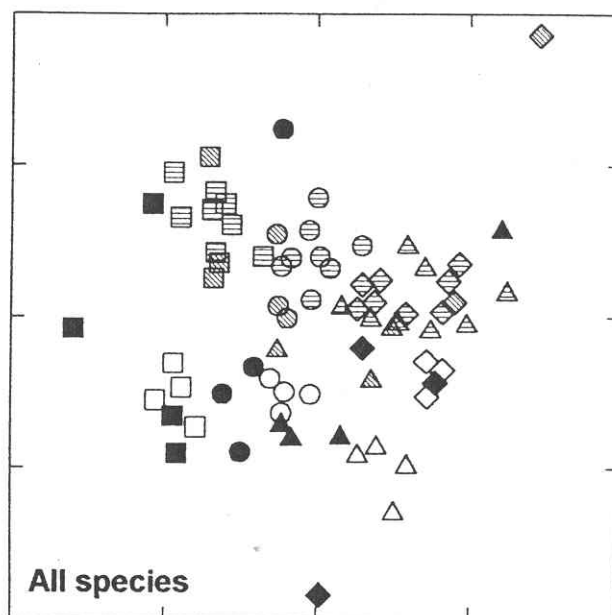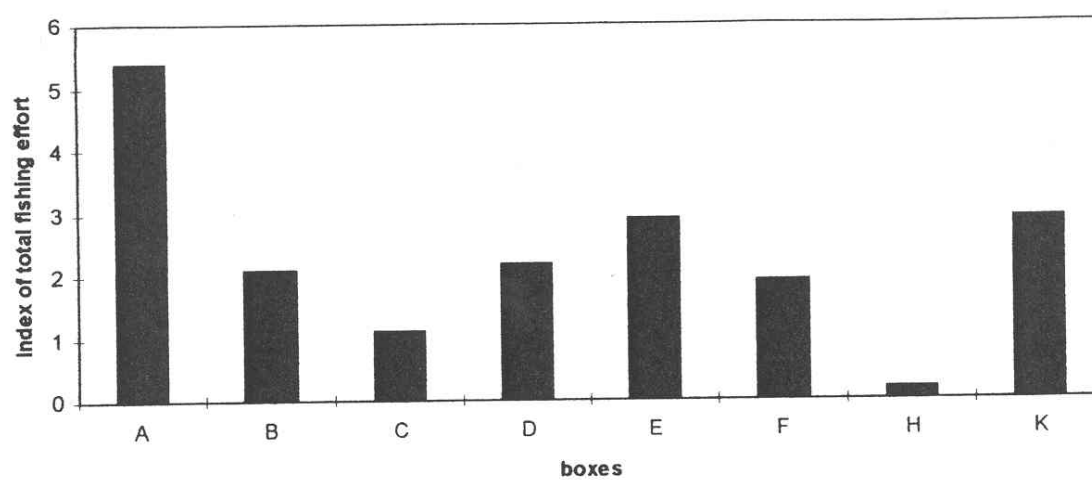
Figure 8.2.3.1.1.3.

Figure 8.2.3.1.1.4.

Figure 8.2.3.2.1.1 Index of total fishing effort in 1991 for the boxes.

Figure 8.2.3.2.1.2   Box A–D. Annual variation of Hill N1 diversity index from 1987 to 1996 of fish assemblages (including pelagic species).

Figure 8.2.3.2.1.3   Box A–D. Annual variation of Hill N1 diversity index from 1987 to 1996 of fish assemblages (excluding pelagic species).
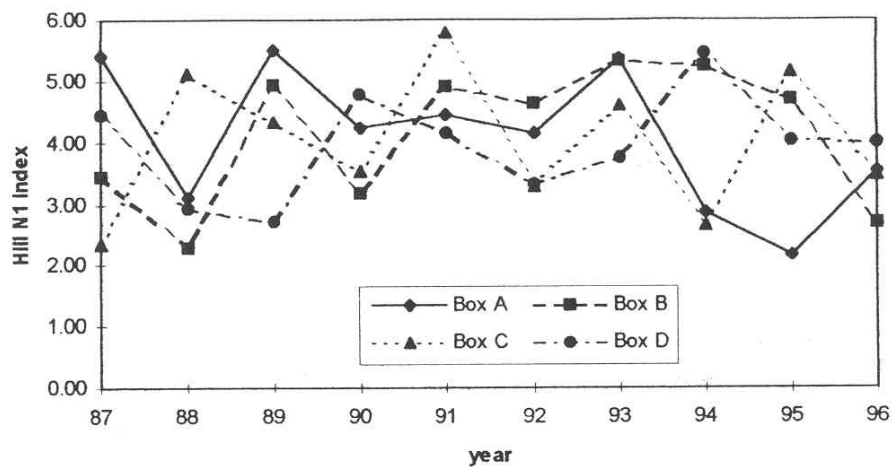
Figure 8.2.3.2.1.4   Box E–K. Annual variation of Hill N1 diversity index from 1989 to 1996 of fish assemblages (including pelagic species).

Figure 8.2.3.2.1.5   Box E–K. Annual variation of Hill N1 diversity index from 1989 to 1996 of fish assemblages (excluding pelagic species).
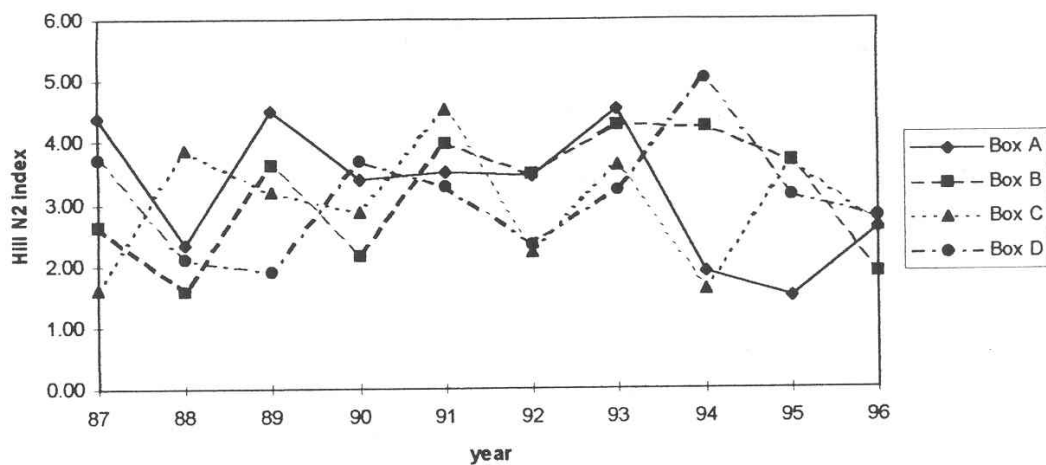
Figure 8.2.3.2.1.6   Box A–D. Annual variation of Hill N2 diversity index from 1987 to 1996 of fish assemblages (including pelagic species).
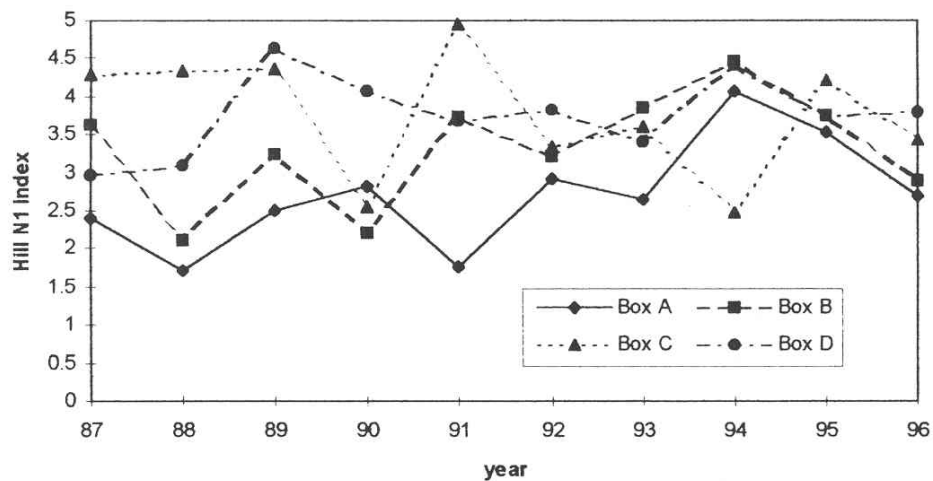

.
Figure 8.2.3.2.1.7   Box A–D. Annual variation of Hill N2 diversity index from 1987 to 1996 of fish assemblages (excluding pelagic species).
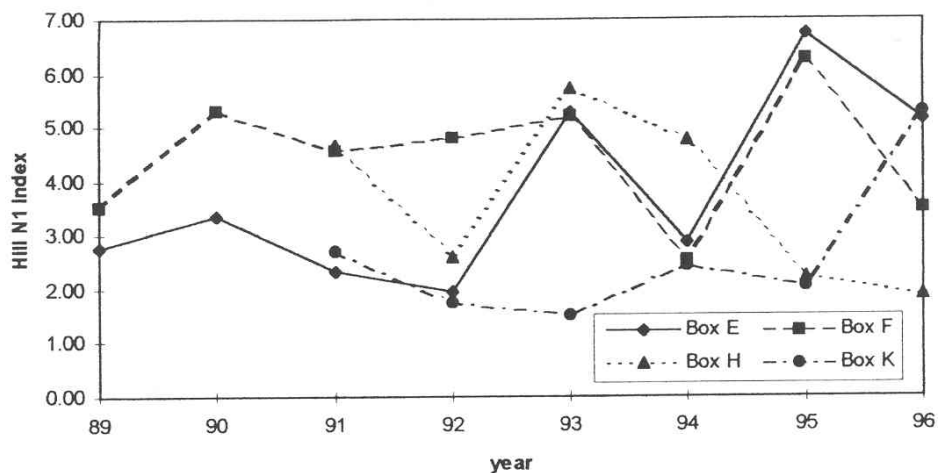
Figure 8.2.3.2.1.8   Box E–K. Annual variation of Hill N2 diversity index from 1989 to 1996 of fish assemblages (including pelagic species).
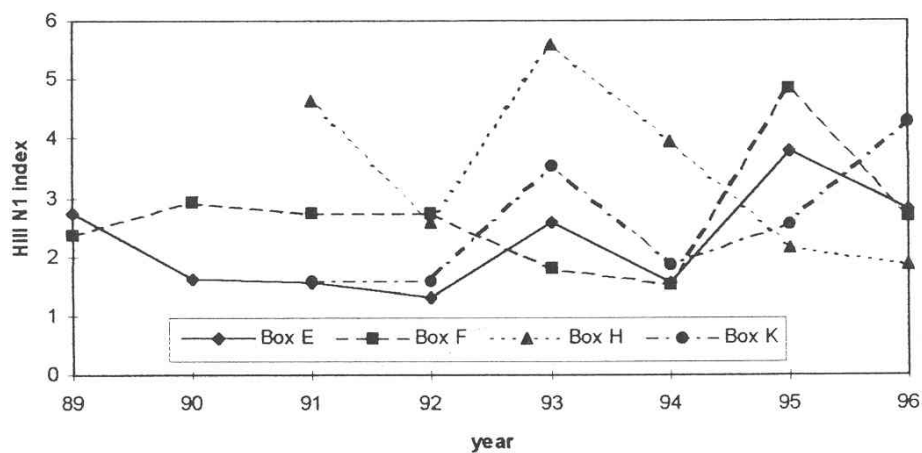

Figure 8.2.3.2.1.9   Box E–K. Annual variation of Hill N2 diversity index from 1989 to 1996 of fish assemblages (excluding pelagic species).
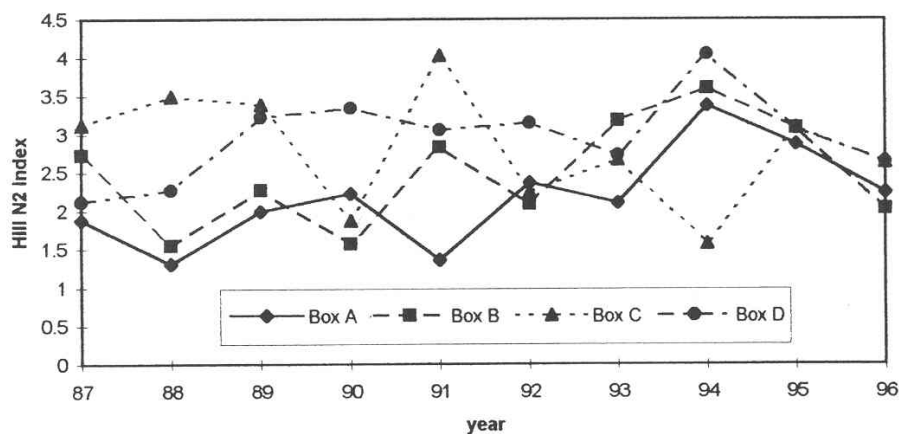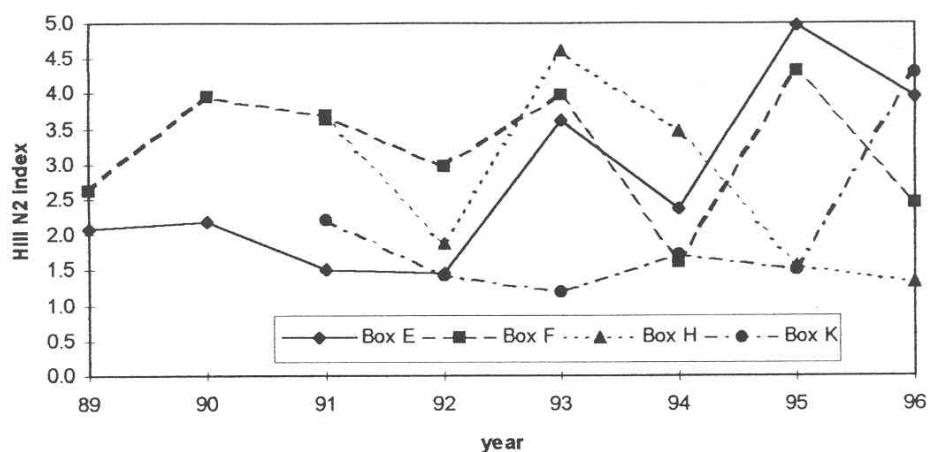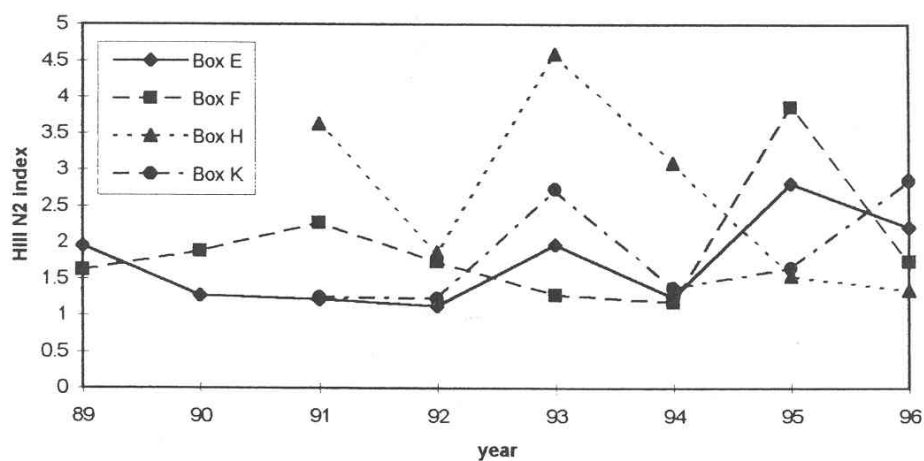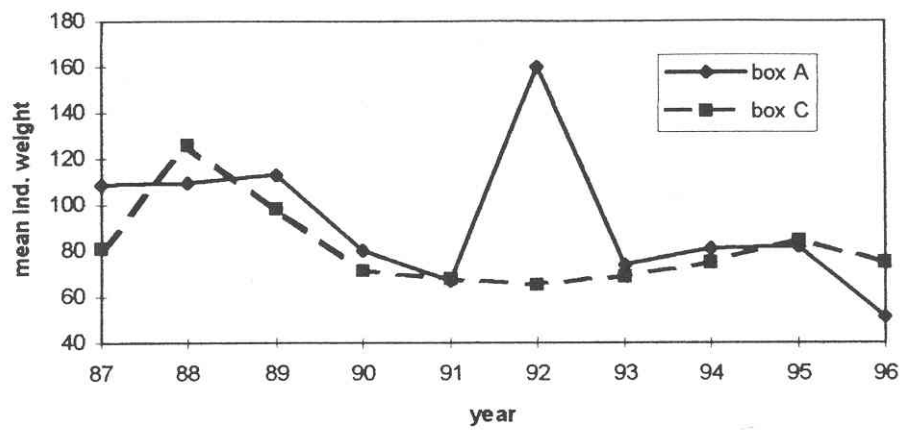
Figure 8.2.3.2.1.10 Changes in mean individual weight in the boxes A and C.

9 Theory and Performance of Ecosystem Models as a Basis for Choosing Metrics of Ecosystem Status and Evaluating Indirect Effects of Fishing

Several modelling approaches were explored, with regard to their ability to provide insight into the ways that fisheries affect ecosystems and their components. First, the various modelling approaches were illustrated in applications prepared by modellers with expertise in the various approaches. These subsections (9.1-9.4) generally present the positive aspects of each approach. A more comparative and critical evaluation of the alternative approaches is presented in Section 9.5, where performance of models are compared to each other, and evaluated against external standards. Limitations as well as potential strengths of model-based approaches to selecting metrics receive more attention in these sections.

9.1 Mass-Balance Models – Theory and Performance

9.1.1 Concepts of aggregate ecosystems models – ECOPATH, ECOSIM, and ECOSPACE

All three of these aggregate ecosystem models were used in the study carried out by Sanchez and Olaso (1999). The use of these models in this study is summarised here by way of an illustration of how they might be applied to examining the indirect effects of fishing on marine ecosystems.

The study used ECOPATH to estimate the following: the trophic level of the fishery, the transfer efficiency between trophic levels, the mixed trophic impact with the fishery as both an impacted and an impacting component, and the historical mean trophic level of the fishery. The input data required were:

- An estimation of biomass (obtained from ICES, ICCAT and survey data in this study; when this value could not be estimated it was back calculated from the model using an alternative value for ecotrophic efficiency (EE) of 0.95);
- An estimation of mortality (obtained from ICES, ICCAT and survey data in this study);
- An estimation of consumption-to-biomass ratios and diet composition of each predator; and
- Landings and discard data from the fishing fleet.

Feeding habits form the links between groups derived from predator-prey relationships determined by stomach content analysis, or from literature values. Discards were considered and 60% were returned to the model (the percent that falls to the seabed), and the 40% taken by seabirds (not included in the model) were accounted for in the category 'exports'. The model was parameterised as 'top-down' such that flow at low trophic levels was set so as to match the food demands of the top levels. Achieving this energy balance assumes a steady-state equilibrium, and is perhaps one of the major drawbacks of the ECOPATH model.

Outputs: Major biomass flow diagram for the studied ecosystem for a given time, showing which groups determine the main flow and which groups form the link between pelagic and demersal groups. The flow of discards can be mapped, and the groups which are most impacted by these discards are identified. This model also provides insight to the direct and indirect trophic interactions within the ecosystem.

ECOSIM is a dynamic extension of ECOPATH (Walters *et al*., 1997). It includes biomass and size structure dynamics, and uses the same mass balance assumptions as ECOPATH for parameter estimation. The study by Sanchez and Olaso (1999) used ECOSIM to simulate the effect of different fishing regimes on the flow structure of the ecosystem.

Outputs: Data showing the evolution of biomass trends over time for trophic groups as a function of different fishing rates. The relationship between equilibrium biomasses at different fishing levels can thus be graphically displayed. These relationships can be predetermined as being 'bottom-up', 'top-down' or 'mixed' controlled systems. The predictions can also be plotted over time. This representation then shows how different species groups are affected by different levels of trawling and how some populations would decline and others could recover.

ECOSPACE is a mesoscale spatial simulation tool for predicting spatial patterns and runs the ECOPATH model through ECOSIM to model the response of the ecosystem (Walters *et al*., 1998). Sanchez and Olaso's (1999) study used ECOSPACE to explore trophic and spatial relationships and to evaluate management options.

The input data required were:

ECOPATH model plus habitat-related parameters, thus replicating ECOSIM dynamics over a spatial grid of 'homogeneous' cells and linking the cells allowing for the dispersal of organisms and changes in fishing effort. ECOSPACE also allows for spatial variation in productivity and can represent habitat 'preferences' according to different criteria (differential dispersal, predation rates, feeding) (Walters *et al.*, 1998).

Outputs: A series of base maps illustrating the evolution of trophic group abundance in different situations. The example of Sanchez and Olaso (1999) illustrates the ecosystem response to a closed area instigated to protect hake recruits.

These three models provided predictions on the functioning of the ecosystem, and can provide predictions about ecosystem effects of each fishery type on different functional groups in the foodweb. This would provide an idea of the importance of each fishery for the trophic dynamics throughout the system.

9.1.2 Case studies using mass-balance models to compare the trophic structure of ecosystems – pelagic upwelling systems

9.1.2.1 Datasets description

A comparison of mass-balance models of trophic flows in the four large marine upwelling areas (Jarre-Teichmann, 1998) was reviewed to illustrate the potential use of this class of models for intersystem comparisons. For this study, the Ecopath II software (Christensen and Pauly, 1992) was used to balance models for different species dominance regimes in the upwelling systems off northern-central Peru (1964–1971 and 1973–1981), California (1965–1972 and 1977–1985), northwest Africa (1972–1979), and Namibia (1971–1977 and 1978–1983). The data for these models had largely been assembled from published literature and/or reports, and the models were built in strictly the same way such as to allow for intersystem comparisons. The study has been expanded since to include the southern Benguela ecosystem as well.

9.1.2.2 Description of the modelling and analysis methodology

9.1.2.2.1 Construction of the models

Assuming mass-balance over an appropriate period of time, the production of each component of an ecosystem (e.g., a sub-population, species or a group of species) is balanced by its predation by other components in the system (predation mortality), its exports from the system (fishing mortality and other exports), and the baseline mortality. Thus,

Production by (i) = All predation on (i) + nonpredatory biomass losses of (i)
+ fishery catches of (i) + other exports of (i)

The terms in this equation may be replaced by

Production by (i) = $B_i * P/B_i$
Predatory losses of (i) = $\Sigma_j (B_j * Q/B_j * DC_{j,i})$
Other losses of (i) = $(1 - EE_i) * B_i * P/B_i$

For any component in the system, this leads to the linear equation

$$B_i * P/B_i * EE_i - \Sigma_j (B_j * Q/B_j * DC_{j,i}) - Ex_i = 0$$

Where

i   indicates a component (stock, species, species group) of the model,
j   any of the predators of i,
B_i the biomass of i,
P/B_i the production i per unit of its biomass (= total mortality under steady-state conditions),
Q/B_i the consumption of a component per unit of its biomass,
$DC_{j,i}$ the average fraction of i in the diet of j (in terms of mass),
EE_i the ecotrophic efficiency of i (the fraction of the total production consumed by predators or exported from the
        system),

Ex$_i$ the export of i from the system (e.g., by emigration, or fishery catch).

The energy balance of each component is given by

Consumption = Production + Respiration + Non-assimilated food

wherein consumption is composed of consumption within the system and consumption of imports (i.e., consumption 'outside the system'), and production may be consumed by predators, exported from the system, or be a contribution to detritus.

This structure defines the necessary parameters for the model. For each component, an estimate of its biomass, P/B and Q/B ratios, diet composition, its exports from the system, and its assimilation and ecotrophic efficiencies are required. However, for each component one of the parameters B, P/B, Q/B or ecotrophic efficiency may be unknown, because it is estimated when solving the system of linear equations, along with the respiration of that component. The model is regarded as balanced when realistic estimates of the missing parameters have been achieved for all components of the ecosystem.

Analysis of the models

After a model has been balanced, the various estimates of biomass and turnover rates are checked, and if they are mutually compatible, then the balanced model represents a possible and consistent picture of the energy flows in the system. Only after this process has been completed is it meaningful to perform further analyses of the model, e.g., for interactions between its components and/or the role of the fishery. A rich theoretical framework exists for the analysis of energy flows or cycling in ecosystems, notably building on the theories of Odum (1969) and Ulanowicz (1986).
Direct trophic interactions, i.e., predation and fishery, can straightforwardly be assessed by analysing partial mortality coefficients of the prey (or target) groups, and by calculating trophic levels. An additional assessment of indirect trophic interactions, e.g., competition, is possible by mixed trophic impact analysis (Ulanowicz and Puccia, 1990). This approach assesses the relative impact that the change in biomass of a given group would have on the biomass of the other groups in the system. The method is, however, based on the assumption that its trophic structure does not change. Consequently, it is not possible to use it for predictions, but instead as a sensitivity analysis of the cascading effects of changes in an ecosystem's food web.

The partitioning of trophic flows among different consumer groups in an ecosystem can further be illustrative of the role of these consumers in a system, and of their development over time. While fish usually take the largest fraction of fish production (e.g., Bax, 1991; Jarre *et al.*, 1991), the fishery is often the second largest consumer, and often in direct competition with marine mammals.

The fisheries in different ecosystems cannot readily be compared based on their total catch alone, because the species composition of the catch can be rather different. This is in part a result of the specific oceanographic and biological conditions that determine the distribution of a species, but also a result of both fishery management (selection of target species) and fishing practice (selection of fishing gear). Fish are situated at different levels in the food web of an ecosystem, and trophic pathways of different length are therefore required to sustain them. Therefore, the exploitation of fish on lower trophic levels is less expensive in ecological terms than the exploitation of fish on higher trophic levels, and a common currency is needed to compare the ecological cost of fishing among different time periods or systems. Primary production equivalents, as suggested by Pauly and Christensen (1995), are one possibility. Following their approach, a particular end flow in question (e.g., the fishery catch of a species) is traced backwards through the food web, using the ratios of production and consumption of the various components along the path as magnification factors. The sum of the flows leading from the basis of each path (i.e., from the producers' level) to the end flow in question is then the total primary production needed to sustain it.

9.1.2.3   Results

The results of the study indicated that the four upwelling systems ranked rather distinctly after the size parameters primary production, total biomass sustained in the system, fishery catches, and total system throughput (Figure 9.1.3.1). They were set apart in geographical rather than in regime-specific (or temporal) order, although considerable changes in energy flows occurred in some of the systems. Mixed trophic impact analysis showed the importance of primary and secondary production, but also the competition of predatory fish with the fishery, and top-down control aspects like the inhibition of semipelagic fish such as hake through the fishery (Figure 9.1.2.1). The fishery took 20-30% of the production of the five dominant species anchovy, sardine, mackerel, horse mackerel, and hake in all systems except off California where fishing moratoria applied for part of the period under investigation (Figure 9.1.2.2). In this system, a comparatively large fraction of the fish production was consumed by top-predators which are valued more highly by the tourism industry than in the other upwelling areas.

The analysis of primary production required to sustain the fishery (Figure 9.1.2.3) reflected changes in the fishing strategy in systems over time. In the Peruvian system, where the magnitude of the catches was reduced by a factor of more than three between the two periods investigated, the primary production required to sustain the fishery decreased only by 10% as the fishery increasingly targeted hake, a predatory fish, in the later period. The fishery thus remained just as costly in ecologic terms as it had been during the peak period of anchovy exploitation in the 1960s.

9.1.2.4 Discussion

Proponents of these modelling approaches argue that fisheries-oriented construction and analysis of trophic models bear the advantages that:
(i)   they are relatively straightforward to construct,
(ii)   a trophic flow diagram allows to put the commercially exploited species (and the fishery) into the entire ecosystem, giving an immediate visual impact of the trophic flows in the system,
(iii)   a whole toolkit of established methods of network analysis is available to assess, e.g., indirect trophic interactions and top-down control processes,
(iv)   flows can be compared directly between different periods in the same system, or among similarly structured systems.

The comparison of upwelling systems showed, among other results, that the systems are not only driven by food availability as repeatedly suggested, but a number of top-down control mechanisms exists. The position of the small pelagic species in the food web, the low transfer efficiency between trophic levels, and the mixed trophic impact of the lower trophic levels appeared to be rather global properties. By identifying similarities between ecosystems, experiences in their fisheries management could consequently become transferable.

9.1.2.5 References

Bax, N. 1991. A comparison of the fish biomass flow to fish, fisheries and mammals in six marine ecosystems. *ICES Marine Science Symposia*, 193:217–224.

Christensen, V., and Pauly, D. 1992. ECOPATH II—a software for balancing steady-state ecosystem models and for calculating network characteristics. Ecological Modelling, 61:169–185.

Jarre-Teichmann, A. 1998. The potential use of mass-balance models for the management of upwelling ecosystems. Ecological Applications 8(1), Supplement: 93–103.

Jarre, A., Muck, P., and Pauly, D. 1991. Two approaches for modelling fish stock interactions in the Peruvian upwelling ecosystem. *ICES Marine Science Symposia*, 193:171–184.

Odum, E.P. 1969. The strategy of ecosystem development. Science, 242:132–1139.

Pauly, D., and Christensen, V. 1995. Primary production required to sustain global fisheries. Nature, 374:255–257.

Ulanowicz, R.E. 1986. Growth and development: ecosystem phenomenology. Springer Verlag, New York. 203 pp.

Ulanowicz, R.E., and Puccia, C.J. 1990. Mixed trophic impacts in ecosystems. Coenoses, 5(1):7–16.

9.1.3 Case studies using mass-balance models to compare the trophic structure of ecosystems – application to the Baltic Sea – 1900 to the present

9.1.3.1 Description of data

The Baltic Sea is a comparatively young, brackish, boreal ecosystem the coasts of which have been inhabited by people for a long time. Primary productivity increased markedly during the past 90 years due to eutrophication. Increased primary production has contributed to approximately a three-fold increase fish biomass from around 1900 to the present (Thurow, 1997). Fishery catches in the Baltic Sea increased about ten-fold in that period. Marine mammals (three species of seals plus harbour porpoise) were abundant at the beginning of this century, but have almost vanished now due to excessive hunting.

As a case study, the seasonal mass-balance models of carbon flows in the central Baltic Sea of Jarre-Teichmann (1995), which were based on Elmgren (1984), Wulff and Ulanowicz (1989), and ICES (1996), were re-arranged to give an annual average of trophic flows at the end of the 1980s, when cod biomass in the Baltic was very low. Food composition of mysids was updated based on Hansson *et al*. (1997). This model of recent trophic flows was compared to a model of carbon flows in the Baltic around 1900, which constructed during the working group based on Elmgren (1989) and Thurow (1997). The ECOPATH software, explained in Section 9.1.2 of this report, was used for balancing and analysing the models. The results of the comparisons are given in Table 9.1.3.1 and in Figure 9.1.3.1.

9.1.3.2 Results and discussion

Odum's (1969) theory divides measures of ecosystem maturity into five groups; community structure and energetics, life history, nutrient recycling, selection pressure, and system homeostasis. Slightly rearranged, measures of (i)

community structure, (ii) structure of the food web, (iii) nutrient cycling, and (iv) system homeostasis were addressed for this case study (Table 9.1.3.1.). A high community production/respiration ratio indicates a rather immature system *sensu* Odum. Production per unit biomass is relatively high although it has decreased during the past 90 years. More biomass is supported per unit energy flow at present, but the fraction is still relatively low. The same holds true for the total biomass in the system. These indicators of community structure point at a rather immature system, as should be expected from the relatively young age in the Baltic.

Connectance index and system omnivory index, both metrics of the diversity of trophic flows in the model, indicate that trophic niches became narrower during the past 90 years, a result well in line with Elmgren's (1989) description of lost food chains due to bottom anoxia in the deeper parts of the Baltic Sea. The loss of the marine mammals as top predators (and their exploitation) resulted in the loss of at least one level in the trophic flow pyramid. However, the average path length in the system remained approximately the same, as cod took over the former role of the mammals. The transfer efficiency of flows between trophic levels increased, which could be an indicator of stress.

Three metrics of food we structure indicate that the food web stayed approximately the same, apart from the loss of some food chains and the mammals as top predators. This is largely due to unchanged flow patterns at the lower trophic levels. It is not clear to what extent this is an artefact of model construction, nor if so, what other implications the artefact would have for model results.

Cycling, nutrient regeneration and the role of detritus in nutrient regeneration all indicate a loss in maturity *sensu* Odum. The loss of maturity could have been caused by both increased primary production (less food limitation) and stress due to pollution. The increasing oxygen depletion in the Baltic in periods of stagnation (no inflow of high saline, oxygen-rich water from the North Sea) has been extensively discussed. However, its effects appear to show in the system summary metrics derived from this relatively simple model.

Summarising, the seemingly contradictory results from the metrics pertaining to community structure and nutrient recycling, respectively, may be explained in the following way. Energy throughput in the Baltic Sea has increased due to eutrophication, making the system 'larger'. With the loss of homeothermic predators, the average organism size has increased along with the increase of fish biomass. However, the additional nutrients appear not to be worked up as well as before, as there is more standing biomass of fish not being consumed by marine mammals. Their mortality leads to a two-fold increase in detritus accumulation and thus, through increased areas suffering from oxygen depletion, to relatively decreased recycling of nutrients.

The metric that within this framework is the closest to being an indicator for system stability, is the system overhead on exports (calculated according to Ulanowicz, 1986). This metric is slightly higher than for large upwelling systems, and lies in the lower range of other shelf ecosystems. The increasing factor of mutual information (Ulanowicz, 1986) points at increasing certainty about the destination of a given unit of flow in the system. This reflects the lower connectance and system omnivory indices, so there are fewer pathways in the system now than earlier. The metrics of system homeostasis thus indicate a shelf-like system, but one which is more vulnerable than typical shelf ecosystems.

The primary production required to sustain the fishery catches increased from 5% at the turn of the century (this already included intense hunting for mammals) to 15% at the end of the 1980s. Although the fishery as a total is probably sustainable in the Baltic, this is only the case because herring and sprat are comparatively lightly fished. Cod, on the other hand, is outside of safe biological limits (ICES, 1997). An assessment of the total ecological cost of the fishery in a system can therefore not replace the assessment of its impact by species, but it can indicate its general compatibility with the flows in the ecosystem.

While mammals consumed about 35% of the total fish production around 1900, their consumption is now lower than 1%. The fishery took slightly less than 11% of the total fish production at the turn of the century, this fraction increased to 36% at the end of the 1980s. Consumption of fish by fish has been relatively constant, approximately 44% of the total fish production. 9% and 19% of fish production were directed to other sinks in the two periods, respectively.

Assuming the same diet composition for mammals as used at the beginning of the century, there would at present be enough food to sustain slightly less than half of their biomass at the turn of the century, i.e., 3 mg C m^{-2}. The fish production which would be available to mammals appears at present not to be directly consumed in the system, but to enter the detrital food chain. The observation that the seal population in the Baltic is presently increasing at a high rate supports the assumption of available food in the system. However, it is without doubt that the present level of fishing overlaps with the food requirements for mammals at their historic population size. Which of the two forms of consumption in the ecosystem is to be preferred is necessarily a choice of society, balancing, e.g., cultural preferences, economic returns, and a commitment to sustain biodiversity. Whichever the choice, it must be ensured that habitat requirements beyond food supply are also met.

The results of this case study need to be viewed with caution, as the model around 1900 pertains to the entire Baltic Sea, while the model of the late 1980s was constructed for the central Baltic (ICES SDs 25, 26, 28, 29), excluding the western Baltic, Gulf of Riga, the Bothnian Bay, and the Gulf of Finland. However, those conducting the work believe that the trends which emerged here are correct, as it was the more vulnerable areas of the Baltic which were excluded from the more recent model, while the bulk of the fish production has always taken place in the central Baltic Sea. Furthermore, the balancing of the models can only be regarded as preliminary due to time constraints.

The study also showed that the Baltic was far from an unexploited system around 1900, and a considerable further step backwards in time may be required to arrive at a system which was not subject to major anthropogenic influence.

9.1.3.3   Metrices addressing the impact of fishing in this case study

Fishing practices in the Baltic have changed substantially from the beginning of this century. Not only have the catches increased by one order of magnitude, but at the turn of the century a coastal fishery existed which was largely directed towards herring, in combination with seal hunting. At present, the herring fishery continues, but removals of sprat and particularly cod have increased by factors of about 25.

There are three metrics which in the framework of a mass-balance model directly address the impact of a fishery in the ecosystem. The trophic level of the fishery puts the fishery into the ecosystem as a predator, and the models show that the fishery continues to be the top predator in the Baltic Sea. The transfer efficiency between trophic levels, which increased during the past 90 years, reflects the increased productivity (= mortality) of the exploited fish species. The primary production required to sustain the fishery reflects the increased ecological cost of fishing, taking into account the position of the targeted species in the food web (as discussed in connection with comparative modelling of upwelling systems in an earlier section of this report). The increase of the ecological cost of fishing by a factor of three, agrees well with the observed removal of total fish production.

Direct trophic interactions, i.e., predation and fishery, can be assessed directly by analysing partial mortality coefficients of the prey (or target) groups. In addition, mixed trophic impact analysis allows assessment of the indirect trophic interactions, taking into account, e.g., competition of predator groups for prey. The mixed trophic impact of the fishery changed markedly during the past 90 years. Whereas at the turn of the century mammals were strongly impacted by hunting, cod, herring and sprat were only inhibited very lightly. The mixed trophic impact of the fishery on herring increased (in the negative way) by a factor of 5 from the turn of the century to present. A slight inhibition of sprat at the turn of the century turned to a slight favouring (by inhibiting its competitor and predator at present), consistent with observed trends of increasing biomasses in the Baltic. Cod are at present strongly inhibited by the fishery, which shows through an increase of the mixed trophic impact index by an order of magnitude in the negative way.

9.1.3.4   References

Elmgren, R. 1984. Trophic dynamics in the enclosed, brackish sea. Rapp. P.-v. Réun. Cons. Int. Explor. Mer, 193:152–169.

Elmgren, R. 1989. Man's impact on the ecosystem of the Baltic Sea: Energy flows today and at the turn of the century. Ambio, 18(6): 326–332.

Hansson, S., Hobbie, J.E., Elmgren, R., Larsson, U., Fry, B., and Johansson, S. 1997. The stable nitrogen isotope ratio as a marker of food-web interactions and fish migration. Ecology, 78(7):2249–2257.

ICES. 1996. Report of the Working Group on Multipecies Assessment of Baltic Fish. ICES CM 1996/Assess:2.

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

Jarre-Teichmann, A. 1995. Seasonal mass-balance models of carbon flow in the central Baltic Sea with emphasis on the upper trophic levels. ICES CM 1995/T:6. 26 pp.

Odum, E.P. 1969. The strategy of ecosystem development. Science, 164:262–270.

Thurow, F. 1997. Estimation of the total fish biomass in the Baltic Sea during the 20th century. *ICES Journal of Marine Science*, 54:444–461.

Ulanowicz, R.E. 1986. Growth and development: ecosystem phenomenology. Springer Verlag, New York. 203 pp.

Wulff, F., and Ulanowicz, R.E. 1989. A comparative anatomy of the Baltic Sea and Chesapeake Bay ecosystems, pp. 232–256. *In* Network analysis in marine ecosystems. Ed. by F. Wulff, J.G. Field, and K.H. Mann. Springer Verlag, Berlin. 284 pp.

Table 9.1.3.1   Results of the comparison of trophic flows in the Baltic Sea around 1900 and in the late 1980s.

| Metric/Period	ca. 1900	ca. 1990
Overview		
Primary production (g cm^{-2}yr^{-1})	79	160
Sprat production (g cm^{-2}yr^{-1})	0.013	0.258
Herring production (g cm^{-2}yr^{-1})	0.112	0.351
Cod production (g cm^{-2}yr^{-1})	0.016	0.085
Fishery catches (g cm^{-2}yr^{-1})	0.027	0.295
Mammal consumption (g cm^{-2}yr^{-1})	0.087	<0.001
Trophic level of the fishery	4.30	4.36
Community structure		
Community P/R (yr^{-1})	1.60	1.69
Community P/B (yr^{-1})	37.3	16.8
Biomass supported by unit energy flow (g cm^{-2}yr^{-1})	0.010	0.017
Net community production (g cm^{-2}yr^{-1})	43.5	62.6
Total organic matter sustained (g cm^{-2})	3.54	9.51
Food web structure		
Connectance index	0.209	0.189
System omnivory index	0.137	0.108
Average path length	3.41	
No. of discrete trophic levels	>8	7
Transfer efficiency between trophic levels	9.3%	12.4%
Nutrient regeneration		
Finn's cycling index (%)	12.6	12.0
Nutrient regeneration (overhead on exports) (%)	4.2	3.6
Role of detritus in nutrient regeneration (%)	59	50
Residence time (B/(R+Ex))		
System homeostasis		
Stability (% system overhead)	71.0	73.0
Information content of flows (bytes per trophic linkage)	1.24	1.89
Primary production required to sustain fishery catches (%)	4.7	14.8
Mixed trophic impact analysis of the fishery		
... impacting sprat	-0.018	0.008
... impacting herring	-0.002	-0.014
... impacting cod	-0.032	-0.275
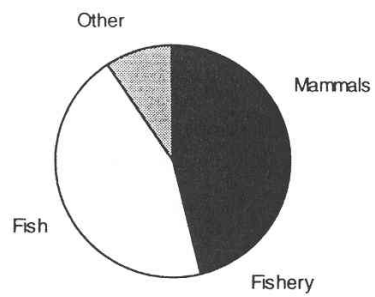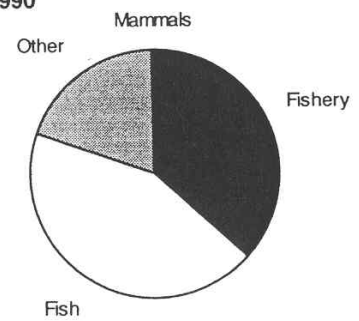
Figure 9.1.3.1 Partitioning of the total fish production in the Baltic Sea by major consumer groups.

9.2 Theory on Size and Diversity Spectra

9.2.1 Size spectra

Studies of the amount of biomass in various size categories has suggested that the logarithm of the biomass in log weight intervals should be approximately constant (Sheldon *et al.*, 1972). Recognizing that the amount of biomass in each size category would depend on the width of the size categories, Platt and Denman (1978) standardized the spectrum by using log biomass divided by the width of the interval on the abscissa and found the standardized spectrum to have a slightly negative slope. Models have been developed which explain the slope and intercepts of the biomass spectrum as a function of the energy-transfer between adjacent trophic levels (e.g., Borgman, 1987; Thiebaux and Dickie, 1993). These models have later been used to predict fish biomass in lakes, but with variable success (Cyr and Peters, 1996).

Fisheries biologists have studied abundance-size rather than biomass-size relations. Pope and Knights (1982) compared the size composition of demersal fish caught by bottom trawl surveys in the North Sea and at the Faroe Islands and found that a straight line fitted log numbers per size class *versus* size in both cases. Subsequent comparisons of size spectra from various parts of the world have confirmed that log numbers per size group often are linearly related to the size of the fish (Pope *et al.*, 1988; Murawski and Idoine, 1992; Gobert, 1994). They have also suggested that the slope of this relationship could be related to fishing intensity. Spectra from areas subjected to different fishing intensities have thus shown that the slope of the size spectra is steeper in heavily fished areas than in less fished areas. The slope is more negative in the heavily fished North Sea than it is in Faroe waters and on the Georges Bank (Pope and Knights, 1982; Murawski and Idoine, 1992). Haedrich and Barnes (1997) linked decreases in biomass and numbers as well as decreases in the mean size of target and non-target species of fish on the northeastern Newfoundland and Labrador shelf to increases in fishing effort. Similar changes in mean size over time was found in analyses of survey data from west Greenland. Analysis of data from two independent bottom trawl surveys in the North Sea revealed an increase in the intercept and a decrease of the slope with time (Section 9.2.3).

In work reviewed at the meeting, for the Bay of Biscay, no significant time trend was found in either slope or intercept of the size spectrum of pelagic and demersal species of fish caught from 1987–1995. When considering only demersal species, a significant increasing trend in the slope and a significant decreasing trend in the intercept was found, but the interpretation was somewhat confounded by significant year/depth strata interactions. Other analyses investigated changes in the species and size composition of the fish catch obtained in five beam trawl surveys in the southern North Sea. The analyses revealed a general increase in the proportion of the smaller size classes of fish in the catch. This increase was tentatively attributed to a decrease in predation leading to an increase in the number of small fish. The findings suggest that the fish assemblage has changed from a top-down regulated assemblage towards a bottom-up regulated assemblage in which increased competition could limit the growth rate of the smaller fish. Although alternative explanations are possible, this interpretation is in accordance with the decrease in weight-at-age of small plaice and the increase in growth of larger plaice (> 35 cm) found by Rijnsdorp and van Leewen (1996).

Section 8.2 of this Chapter provides additional examples of spatial and temporal changes in size spectra within the North Atlantic.

Rice and Gislason (1996) compared the North Sea size spectrum with a size spectrum derived from the numbers at length estimated from the output of MSVPA (Sparre, 1991; ICES, 1997). The changes in the slopes and intercepts of the two spectra were similar. When single and multispecies fish stock assessment models were used to predict changes in the slope and intercept of the size spectrum of the commercially exploited fish in the North Sea in response to fishing it was found that both variables were approximately linear functions of overall fishing effort (ICES, 1996; Gislason and Rice, 1996). The linearity was a consistent feature irrespective of whether a single-species model with constant recruitment was used, a stock recruitment model was added to this model, or a multispecies model (MSFOR) was used to predict the change. In all cases the slope was inversely proportional and the intercept directly proportional to overall fishing mortality. A sensitivity analysis showed that the response of the size spectrum to changes in fishing mortality was virtually unaffected by the level of natural mortality assumed. The response was far more sensitive to changes in growth and stock recruitment dynamics. Changes in growth resulted in major changes in the relationship between fishing mortality and the slope and intercept of the size spectrum. When growth was reduced, the slope and intercept of the size spectrum became much more sensitive to changes in fishing mortality. When growth was increased, sensitivity decreased.

Gislason and Lassen (1997) analysed the mathematical background for the linearity of the change in slope with fishing mortality. Assuming that natural mortality was a function of 1/length:

$$M = a + \frac{b_1}{L} + \frac{b_2}{L^2}$$

where:

M: natural mortality
L: length
a, b_1, b_2:   constants

and that growth could be described by the von Bertalanffy growth equation, and it was shown that the slope of the size distribution for a single-species could be described by:

$$slope_{species} = \frac{\partial \, log \, N(L)}{\partial L} =$$

$$-\frac{b_2}{kL_\infty^2} * (\frac{1}{L(t)} + \frac{L_\infty}{L^2(t)} + \frac{1}{L_\infty - L(t)}) - \frac{b_1}{kL_\infty}(\frac{1}{L(t)} + \frac{1}{L_\infty - L(t)}) - \frac{a+f}{k}\left(\frac{1}{L_\infty - L(t)}\right)$$

where:

N(L):   numbers at length
L_∞, k;   von Bertalanffy growth parameters

Differentiating with respect to fishing mortality the rate of change of the slope of the size distribution with fishing mortality could be described by:

$$r_{species} = \frac{\partial^2 \, log \, N(L)}{\partial f \partial L} = -\frac{1}{k}\left(\frac{1}{L_\infty - L(t)}\right)$$

which means that the slope is directly proportional to fishing mortality for a given length. The rate at which the slope will change depends, however, on the growth parameters, but not on natural mortality.

The size distribution of the biomass will respond in a similar way. Assuming standard isometric growth it follows that:

$$\frac{\partial \, log \, B(L)}{\partial L} = \frac{\partial}{\partial L}\left[log(q * L^3) + log \, N(L)\right] = \frac{3}{L(t)} + s_{species}$$

The slope of the size distribution of the biomass should therefore respond to changes in fishing mortality in exactly the same way as the size distribution of the numbers.
The size spectrum of the entire fish assemblage is estimated by summing up the abundance at size of the individual species:

$$log \sum_{species} N_{species}(L)$$

The slope of the assemblage size spectrum is therefore:

$$\bar{s} = \frac{\sum_{species} N(L)\dfrac{\partial log \, N(L)}{\partial l}}{\sum_{species} N(L)} = \frac{\sum_{species} N(L) * s_{species}}{\sum_{species} N(L)}$$

which is equivalent to the weighted (with abundance) mean of the individual slopes. As the individual slopes decrease with increasing fishing mortality the overall slope will also decrease.

     For the size spectrum we now have sufficient theoretical and empirical evidence to be confident that changes in fishing mortality should result in a long-term change in the slope of the size spectrum. Provided that the growth and the relative recruitment of the constituent species do not change, the change in the slope should be directly proportional to the change in fishing mortality.

Over shorter time-spans the spectrum will change due to interannual changes in recruitment. Over longer time-spans changes in recruitment levels might also affect the slope. Murawski and Idoine (1992) thus suggested that the size composition was a conservative property of demersal fish assemblages, and that species replacement would counteract the effect of fishing on the size spectrum slope. Similarly it cannot be ruled out that a general environmental change could result in changes in the level of recruitment that were different for large and small species. If the level of recruitment for large species declined relatively to the level of recruitment for small species, the slope of the size spectrum would decrease in a way which might be indiscernible from the influence of an increase in overall fishing mortality. Finally, the response of the spectrum is sensitive to changes in growth, and growth changes might influence the slope of the size spectrum in way similar to fishing. With these possibilities in mind, and interpreted with care, the slope of the size spectrum seems to be a useful indicator of changes in fishing effort.

9.2.2 Diversity spectra

Temporal patterns in species diversity with size have been investigated several times (ICES CM 1994/Assess/Env:1; ICES CM 1996/Assess/Env:1). The rationale behind this work has been that fishing would effect larger slower growing and late maturing species to a larger extent than smaller species with a more rapid turnover. If this is the case, changes in diversity with size are expected with changes in fishing effort. There is now some evidence that this might take place. Analyses have found that the abundance of species with a large size at maturity decreased while those with a small size at maturity increased in beam trawl surveys in the southern North Sea. These results suggest that diversity by size group should change with fishing, but at different rates and possibly even in different directions in different size groups.

Where patterns in diversity by size do show some changes over time, the results are far from easy to interpret and difficult to link theoretically to fishing effort. Indeed, the modelling study of Gislason and Rice (1996) suggests that the diversity spectrum would be among the less useful measures of changes in fishing effort. The way in which the slope and intercept of the diversity spectrum changed with fishing effort differed between single-species models with and without stock/recruitment relationships and multispecies (MSFOR) models. Furthermore, none of the models predicted the higher evenness at low levels of fishing mortality suggested by the analyses of survey data from the North Sea from 1906–1909 and 1990–1995 made by Rijnsdorp *et al*. (1996).

Furthermore, species diversity is assessed with a multitude of diversity indices. Each of these indices combines information on species richness and evenness into a single number. High evenness occurs when species are equal or approximately equal in abundance, low evenness when the species composition is dominated by a few abundant species. Due to the relative importance each index gives to evenness and richness, it is difficult to compare the indices. A Working Paper by Rogers *et al*. (in press) compared diversity of coastal demersal fish faunas in the northeast Atlantic by diversity profiles calculated from:

$$H_\alpha = (\log \sum_i p_i^\alpha) / (1 - \alpha)$$

Substituting 0.1 and 2 for the scale parameter α, H_α, will be directly related to species richness, Shannon's entropy and Simpsons dominance index, respectively. Thus for α near zero, the index will be dominated by richness, while for larger values of α, species evenness will have progressively more effect.

Without a theory to provide a causal link between fishing intensity and diversity, it will be difficult to know whether diversity is a useful measure of fishing impact. Recent work by Hall and Greenstreet (2000) suggests, however, that there are patterns in relationships between species richness, individual abundance, and size which might be linked to fisheries effects at the community level.

Hall and Greenstreet described the relationships between species diversity, the abundance of individuals, and body size in a demersal fish community. They investigated patterns in different geographic regions in the northwestern North Sea and over a 60-year period. A striking similarity with previously reported data for insect communities was observed. A dome-shaped relationship between both species richness (S) and individual abundance (I) with body size was found when data were categorised in logarithmic (to base 2) weight classes. The same power law relationship between S and I, of the form $S = aI^b$, existed for both types of fauna. The coefficient *b* of this relationship did not differ between regions or over time, whereas the intercept *a* declined over time. This decline could not be accounted for by sampling artefacts and Hall and Greenstreet suggest that it may provide an informative measure of the effect of fisheries exploitation on the community. They also demonstrated that rank abundance relationships within body size classes exhibited a similar pattern to that found in insects, of the form $A \, r - m$ (where A = abundance and r = species rank). These similarities with insects and the robustness of the patterns for fish when compared over large spatial (100 km) and temporal (decadal) scales, suggest that common explanations may underlie the organisation of these communities. With respect to fisheries effects, it would appear from these data that the coefficient *a* of the power law relationship $S = I^b$, when data are categorised into weight classes, might be a valuable measure of the effect on fish species assemblages of fishing disturbance.

The empirical studies of changes in species diversity have been inconsistent and the theoretical understanding has not advanced to a state where the underlying process can be modelled. More work is therefore needed before predictions can be made about how fishing would affect the diversity spectrum.

9.2.3 Diversity profiles

Improvements in the measurement and interpretation of diversity have recently been made using methods of diversity ordering (Tothmeresz, 1995), where a range of diversity indices within a family show varying sensitivities to rare and abundant species. These profiles display graphically a family of diversity indices obtained by changing the scale parameter a. There are several available, but one that is recommended for large datasets is Renyi's diversity index family.

$$H_a = (\log \Sigma \, p^a_{\ i}) / (1 - a)$$

When substituting 0, 1, and 2 for the scale parameter a, H_a will be directly related to the species richness (i.e., is the log. of the species number), Shannon's entropy and Simpsons dominance index, respectively (Hill, 1973). Thus for a near zero, richness will have more effect on H_a, but for larger values of the scale parameter, species evenness has more effect. For scale parameters which increase from 1 to 4 the influence of rare species will be gradually replaced by the influence of dominant species. One community is more diverse than another if its' diversity profile is equal to or above that of another, over the whole range of the scale parameter. If the two profiles intersect at any point then they can be considered non-comparable (i.e., different diversity indices would rank the communities differently).

Diversity profiles were calculated for the demersal fish catches (number/8m beam trawl/hour) from the coastal waters of the northeast Atlantic (Rogers *et al.*, in press). Results suggest that this is a robust technique for identifying differences in diversity between assemblages, which takes account of all combinations of species richness and evenness.

9.3 Community metrics models

These models begin with basic concepts from community ecology, developed primarily in terrestrial ecosystems, and attempt to represent marine ecosystem dynamics that would follow from the community processes.

9.3.1 Huston's Dynamic Equilibrium Model

Initial theoretical community ecology models presupposed that the species diversity of communities was underpinned by competitive exclusion until ultimately competitive equilibrium was achieved (MacArthur, 1970; MacArthur and Levins, 1967). Many early studies supported these ideas (e.g., Park, 1948, 1954; Pianka, 1973, 1975, 1983; Fenchel, 1975; Davidson, 1977), but not all. Communities were identified which, while apparently stable, were clearly not at competitive equilibrium (e.g., Paine, 1966; Patrick, 1975). These suggested that predation was the dominant process governing the structure of communities. In its absence communities progressed to competitive equilibrium, in its presence an alternative stable, or even unstable, state existed (e.g., Connell, 1975). The apparently opposing effects of competition and predation led Huston (1979) to propose the 'dynamic equilibria model' of species diversity, the fundamental predictions of which are encompassed in Figure 9.3.1.1. Huston considered predation to be simply one source of 'disturbance', i.e., a factor that caused mortality and limited the scope for population growth. In the application of this model, fishing can be considered as a disturbance resulting in mortality, causing at worst, reductions in population size, or at best, limiting the extent of population increase. Variation in productivity puts limits on the amount of food resources available to competing predators.

The theory assumes a unimodal response of species diversity to both disturbance and productivity. Thus for example, nutrient enrichment events have been shown to result in both increases and decreases in species diversity (e.g., Mirza and Gray, 1981; Russo, 1982), i.e., both limbs of the unimodal relationship. At low disturbance frequency, diversity is highest at the intermediate to low end of the productivity spectrum. At the lowest productivity, population growth is so slow that extinction is a distinct possibility. At higher productivity, the scope for growth is sufficiently high that the populations of the dominant competitors will be able grow fast enough, so as to be able to capture an increasingly unequal share of the resources. Growth of the subordinate competitors will be impeded and they will be excluded before the next disturbance occurs. At higher disturbance frequencies, diversity will be lower at low productivity levels; some populations will be unable to grow fast enough to recover sufficiently between disturbance events and they will be driven to extinction. At intermediate productivity levels, populations will be able to grow fast enough to avoid extinction, but the growth of the dominant competitors will be checked, so preventing competitive exclusion. Species diversity in areas of high disturbance frequency and high productivity will be lower than in areas of intermediate levels of both because the high frequency of disturbance is likely to eliminate k-selected species and so reduce the total species pool available. Huston (1994) examined a large number and variety of species assemblage data sets and generally found that variation in species diversity could be explained in terms of the dynamic equilibria model.

What insight can this model provide regarding the impact of fishing disturbance on North Sea groundfish species assemblages? Figure 9.3.1.2 shows that the relationship between diversity and disturbance is strongly influenced by environmental productivity. Consider an increase in disturbance over the range of 20 to 80 units at the three levels of productivity marked on Figure 9.3.1.1. In productive environments increasing disturbance results in increased species diversity, while in unproductive environments the opposite trend is observed. In moderately productive environments diversity increases as disturbance rate increases to intermediate levels, thereafter diversity decreases. Secondly, the model suggests that without accounting for variation in productivity in some sort of multivariate analysis, it could prove difficult to determine any relationship at all. Figure 9.3.1.3 looks at the mean (and standard deviation) species diversity predicted by the dynamic equilibrium model at various disturbance levels over the whole productivity range. The three levels of fishing disturbance shown on Figure 9.3.1.1 are indicated. This graph suggests that we would be very unlikely to detect any significant variation in species diversity among the three fishing disturbance regimes from sets of random samples taken across the full productivity range. Finally, Figure 9.3.1.4 examines the response of species diversity to variation in productivity at the three levels of fishing disturbance indicated on Figure 9.3.1.1. In low productivity environments, species diversity is highest at low disturbance and least at high disturbance, a ranking which intuition suggests might be expected. However, at intermediate productivity the ranking starts to alter, so that where productivity is high the order is completely reversed; diversity is greatest where disturbance is most prevalent and *vice versa*.
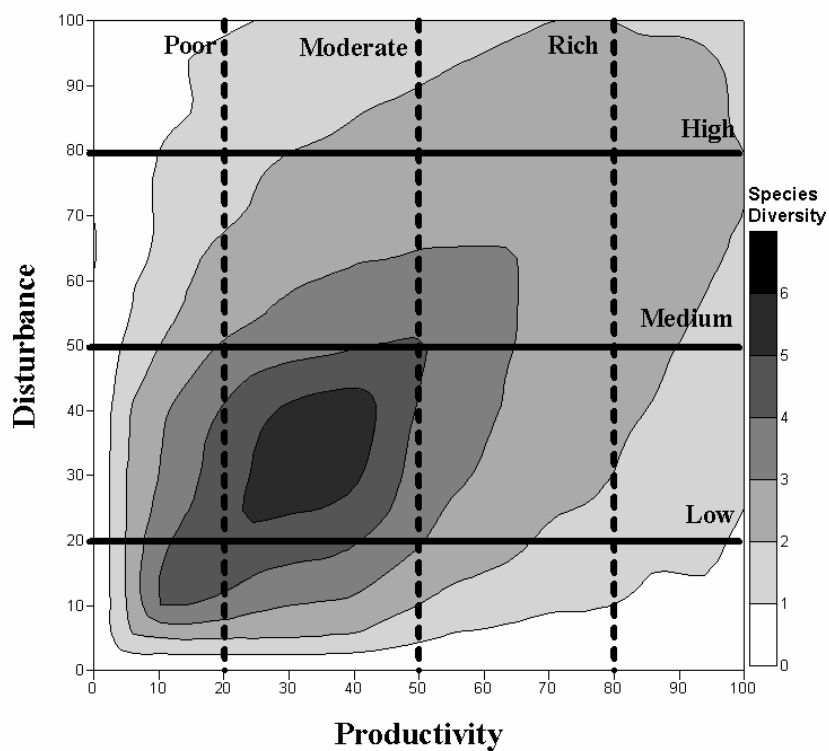
Figure 9.3.1.1   Huston's (1979) 'dynamic equilibria model' of species diversity.
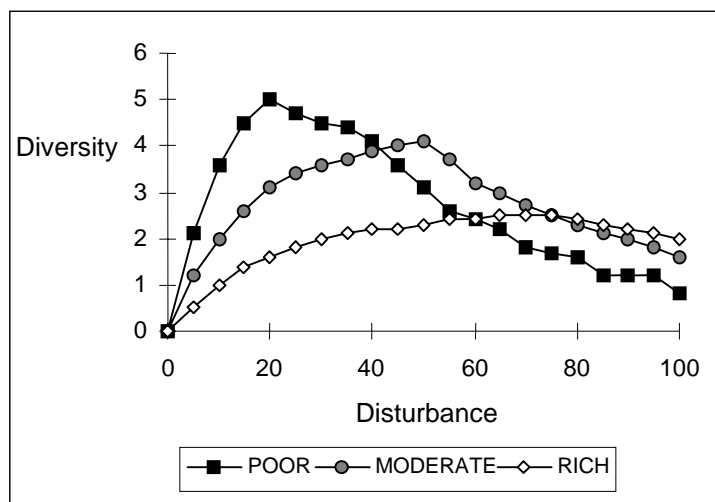

Figure 9.3.1.2   Relationship between species diversity and fish disturbance intensity at three levels of environmental productivity**.**
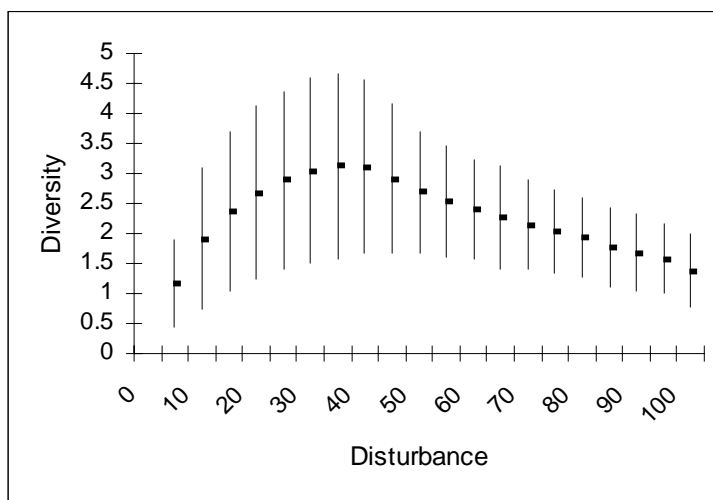
Figure 9.3.1.3   Effect of disturbance on the mean diversity and standard deviation estimated over the full productivity range. Low, medium, and high disturbance ranges shown on Figures 9.3.1.2 and 9.3.1.4 correspond to regions labelled low, medium, and high in Figure 9.3.1.1.
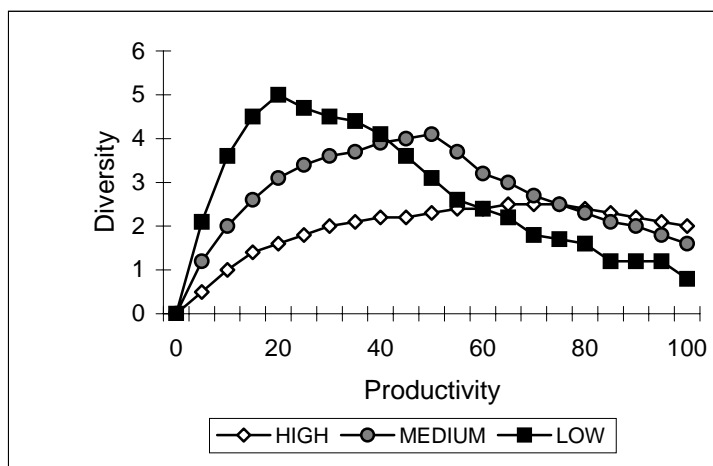

Figure 9.3.1.4   Relationship between species diversity and productivity at three fishing disturbance intensities.

The effects of disturbance on community species diversity are thus far from clear-cut (e.g., Death and Winterbourn, 1995). They may be particularly complex in marine ecosystems (Russo, 1982; Menge and Sutherland, 1987), underlining the necessity for the provision and validation of underlying theory. One of the difficulties in applying such models, however, is the frequent absence of the necessary data to fully parameterize it, in this case a direct measure of system productivity. One approach would be to find suitable correlates, such as water depth in marine systems (Rosenzweig and Abramsky, 1993). However, the use of such substitutes requires caution (Rosenzweig, 1995). A second approach may be to use empirical relationships (e.g., Brey, 1990; Duineveld *et al*., 1991; also see Bryant *et al*., 1995) to estimate benthic production from benthic standing crop biomass data. Such data may be available for the North Sea, at least in basic form, collected during the ICES North Sea Benthos Survey (Heip *et al*., 1992; Basford *et al*., 1993), and additional data continue to be collected. This raises the possibility of examining the relationships in spatial variation in fishing disturbance, benthic production and demersal fish species diversity to explore the value of Huston's model as a tool to enable us to predict the consequences of changes in fishing practice on groundfish species diversity.

9.3.2   An age/size-structured ecosystem model—European Regional Seas Ecosystem Model (ERSEM)

Most of the early models used to describe energy/carbon/nitrogen flow through marine food webs essentially assumed a steady-state dynamic equilibrium situation; they were 'static' models (e.g., Steele, 1974; Billen, 1978; Jones, 1982, 1984; Cohen *et al*., 1982; Mommaerts *et al*., 1984). The alternative view is that the food web is not at equilibrium and is instead constantly responding to environmental and/or anthropogenic forcing. Describing such situations required the application of 'dynamic' time-evolving models (Fransz and Verhagen, 1985; Billen and Lancelot, 1988). Both of these approaches require simplified representation of the ecosystem by the aggregation of species into functional groups. The degree of species aggregation varies considerably between models. An overview of many of these earlier models is provided by Fransz *et al*. (1991).

More recently many research institutes around the North Sea were involved in a project to develop a spatially explicit model of carbon pathways through the North Sea ecosystem (Baretta *et al.*, 1995). The model used the ten ICES areas to achieve a relatively coarse spatial resolution. The five offshore boxes were split into two by depth to model the effect of the thermocline. Primary production occurred in the upper box, whilst the lower box included the links to the benthos. A general circulation model was used to generate daily horizontal exchange rates of dissolved and suspended constituents between the ten surface boxes. Vertical transport between the five upper and lower boxes was based on determined sinking rates for the sedimentation of particulates and turbulent diffusion for the dissolved constituents.

The physical model contained information specific to the area to be modelled, whereas the biological/chemical sub-models were constructed to be non-site-specific. The biological variables were represented as functional groups expressed as units of organic carbon and the chemical variables as internal pools in the biological variables and as dissolved inorganic pools in the water and the sediment, expressed in units of N, P, and Si.

The model is modular in its construction, with each module dealing with a related collection of functional groups, thus the Zoobenthos model included the functional groups: Benthic Carnivores; Suspension Feeders; Deposit Feeders; and Meiobenthos. The modules were set up to run within the ten ICES boxes and the five lower layer boxes using the Software Environment for Simulation and Analysis of Marine Ecosystems (SESAME) (Ruardij *et al.*, 1995). The modules were linked to allow the exchange of carbon and nutrients between the modules. These were then routed through the functional groups comprising each module. Size and age structure was explicitly represented in the fish groups, but the remaining biological components were modelled as unstructured populations. This was deemed appropriate for taxa having short generation times relative to the annual cycle, and for taxa which did not cross more than one trophic level in their lifetime.

This model has not been used to examine the indirect effects of fishing on the emergent properties of the ecosystem. However, it would be a relatively easy step to introduce some rate of fishing mortality to the age-structured fish module. Such mortality could even be applied to take into account spatial and seasonal variation in the behaviour of different fisheries. The consequent changes arising from alteration to the carbon flow through the various modules could then be tracked over any given time period (1 to 10 years, or longer if appropriate). Examples of the sort of ecosystem food web alterations for which predictions could be obtained are: the consequences of sandeel fisheries operating in specific locations to top predators in the area, and to the zooplankton populations on which the sandeels prey. Furthermore, predictions could also be obtained regarding the effects of all these higher trophic level changes on the benthos-pelagic coupling in the system.

9.4 Evaluating Ecosystem Effects of Fishing: Predictions from Ecosystem Dynamics Models

Building on the illustrations of the major modelling approaches, WGECO next used the principal models of ecosystem dynamics to develop specific predictions regarding the ecosystem effects of fishing;

9.4.1 Inventory of models of ecosystem dynamics

There has been a multitude of models constructed, each of which purports to illustrate the dynamics of ecosystems. A useful classification of these models was provided by a flowchart in Hollowed *et al.* (2000) (Figure 9.4.1). This flow chart provides a useful guide to grouping together models based on similar constructs, requiring similar input variable data and producing similar output predictions. We have used this flow chart to produce the key to ecosystem models presented in Sections 9.3.1-9.3.3, and other models from the literature. We have used the key to assign models gleaned from the theoretical ecology and fisheries science literature to various categories or 'families' of models. Models within a family will provide essentially the same sort of insight into how fishing may affect the ecosystem. It may be that each model may require slightly different input data, thus some models within a family may be more or less appropriate, or practical, when attempting to address a particular issue. Different families of models will essentially address different issues, or provide different insights into ecosystem operation. Chapter 1 concluded that there is no evidence that we will fail to safeguard the marine ecosystems as long as single-species issues are addressed adequately, such that no one component or species within the ecosystem is subject to unsustainable mortality, and habitats are protected. This message is confirmed in our current deliberations. Only a few of the models mentioned below consider single-species in a way that can be directly compared with current assessment models, dealing as they do in the most part with multispecies interactions. Multispecies models therefore provide a means of examining how fishing disturbance might affect the emergent properties of ecosystems, in particular food-web dynamics and change in species diversity. This, in turn, might provide the definitive answer to questions about the risk posed to emergent ecosystem properties in systems where individual species are not at risk. In Section 9.4 we briefly describe some of the models in each of the model families defined by the key. We indicate what they do, some of the key assumptions, and the type of output they provide.
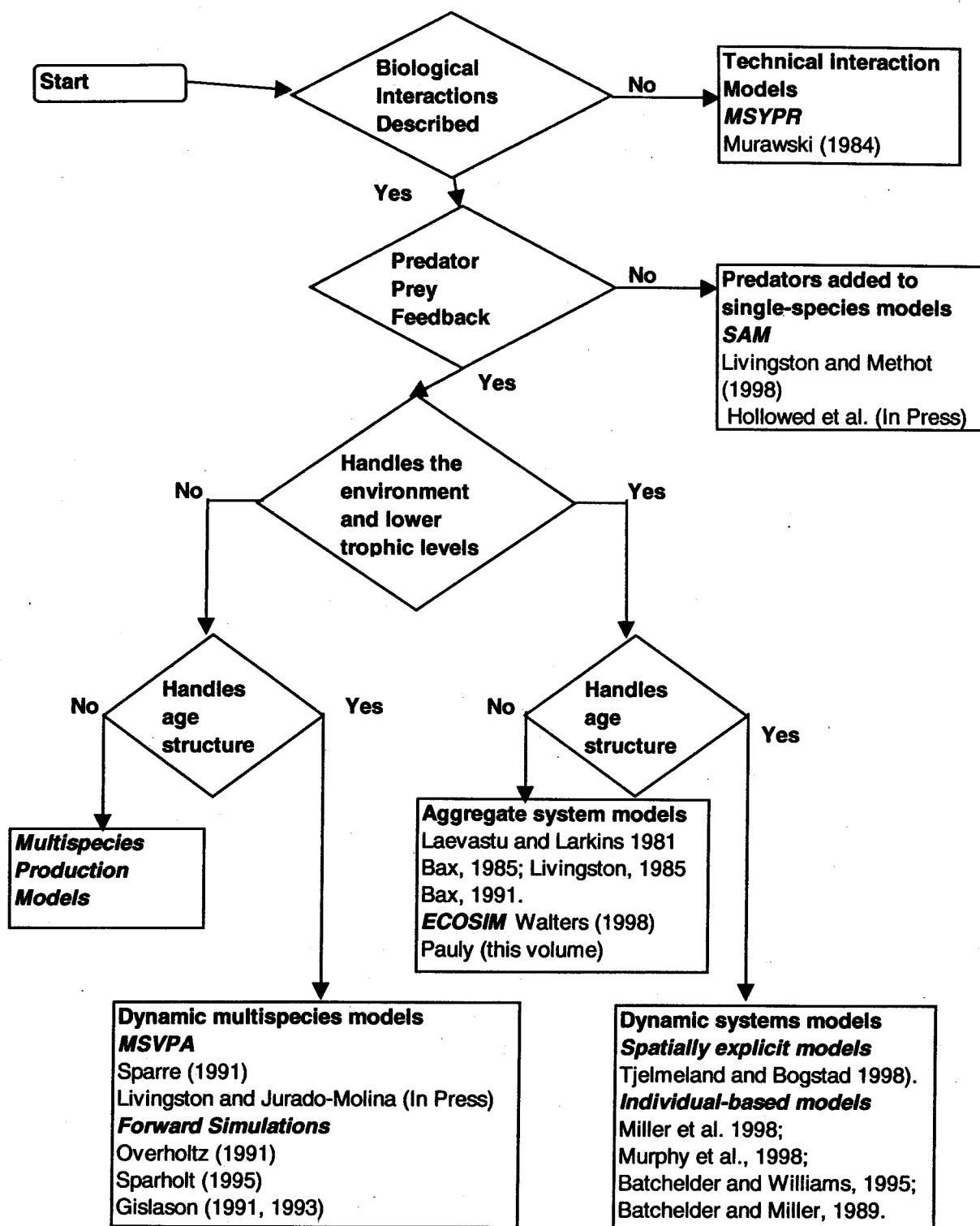
Figure 9.4.1   Flow chart summarizing classification of multispecies models. **Bold** letters indicate model classification, *italicized* letters indicate sub-categories of models. References for classes and sub-categories of models are provided in the text of the paper by Hollowed *et al.* (2000).

9.4.2   Model type key

1) Is habitat suitability explicitly included?
    No   go to 2
    Yes   Basin model (MacCall, 1990)

2) Is model based on community metrics?

No   go to 3
Yes   Dynamic equilibrium model (Huston, 1979, 1994)
      Size-spectrum model (Pope *et al.*, 1988)

3) Predator-prey-feedback included?
    No   Prey added to single-species models (Furness, 1978)
         Predators added to single-species models (Livingston and Methot, 1998)
    Yes go to 4

4) Are the environment and lower trophic levels included?
    No   go to 5
    Yes go to 7

5) Age structure?
    No   Multispecies production models
         a. Multispecies Lotka-Volterra models (May *et al.*, 1979)
         b. Predator-prey models with non-linear interactions (Collie and Spencer, 1994; Spencer and Collie, 1996;
             Basson and Fogarty, 1997)
         c. Spatially-explicit predator-prey models (Pascual and Levin, 1999)
    Yes go to 6

6) Dynamic multispecies models with age-structure [a], variable growth [g], multiple fleets [f], spatial structure [s]
    a. MSVPA (Sparre, 1991) [a, f]
    b. MSGVPA (Gislason, 1999) [a, g, f]
    c. Length-based MSVPA (Dobby *et al.*, 1999) [g, f]
    d. MULTSPEC (Tjelmeland and Bogstad, 1998) [a, f, g, s]
    e. Bormicon (Stefànsson and Pàlsson, 1997) [a, f, g, s]
    f. 4M (Vinter and Thomsen, 1998) [a, f, g, s]

7) Age-structured?
    No   go to 8
    Yes go to 9

8) Aggregate system models with time dynamics [t], spatial dynamics [s]
    a. N→P→Z models
    b. ECOPATH (Polovina, 1984)
    c. ECOSIM (Walters *et al.*, 1997) [t]
    d. ECOSPACE (Walters *et al.*, 1998) [t,s]

9) Age/size structured ecosystem models
    a. Andersen and Ursin (1977) North Sea model [t][*]
    b. ERSEM (Baretta *et al.*, 1995) [t, s]
    c. Individual based models (Murphy *et al.*, 1998; Batchelder and Williams, 1995)
    d. OSMOSE (Shin and Cury, 1999) [a, t, s]

9.4.3   Description of models and predictions for the ecosystem effects of fishing

The classes of models correspond to the key above. We list the principal properties of each class of model without reviewing their validity or usefulness. Generic predictions are made about the ecosystem effects of fishing if each of the models were a correct description of the ecosystem. Several of the models have been described more fully in Section 9.2. A more thorough review will require a more complete description of each model's properties, consideration of the underlying assumptions or theory, and empirical evidence for the model. For those models that have been implemented and parameterised, estimates of the ecosystem effects of fishing can be made.

Habitat-based models

Properties
The population growth rate depends on habitat suitability. Realised suitability is a function of local population density. Individuals distribute themselves with an ideal-free distribution such that realised suitability is equal over the entire range of habitats (see MacCall, 1990).

---

[*]This model is no longer in use.

As fishing reduces the total population size, the population range will contract to the most suitable habitats. Catchability will remain constant in the core habitat as total population size declines because local population density is highest in the most suitable habitats.

Models based on community metrics

Properties
These models are generally applied at the community level and do not describe the abundance of individual species from one generation to the next. According to the dynamic-equilibrium model, species diversity is a function of ecosystem productivity and natural disturbance levels (see more detailed description of this model in Section 9.3.1). The size-spectrum model describes the decrease in number of individuals with increasing length classes.

Predictions
The dynamic equilibrium model predicts that species diversity will change in response to fishing disturbance. The direction and magnitude of the change depends on levels of production and natural disturbance in the unimpacted ecosystem. The slope of the size spectrum is expected to become steeper when the community is exploited. The relationship between the slope of the size spectrum and fishing mortality has been investigated by simulation (Gislason and Lassen, 1997) and by comparing the size spectra from fish communities around the world (Bianchi *et al*., in press).

Single-species models with variable prey or predators

Properties
These are dynamic single-species models with either variable prey abundance (e.g., fish abundance for seabirds) or variable predator abundance (e.g., predators on fish populations). Trophic feedback is in one direction only.

Predictions
In the case of variable prey, harvesting the prey species (forage fish) will reduce the reproductive rate or growth rate of the predators (seabirds). In the case of variable predation, harvesting the predator will reduce the natural mortality of the prey species and may lead to higher prey recruitment (Walters *et al*., 1986). Harvesting the prey would not affect the predators because of the lack of prey-predator feedback. These models could perhaps be used to examine some of the immediate effects of alteration in the abundance of particular single-species, i.e., the type of change which has commanded most attention throughout the remainder of the report.

Multispecies production models

Properties
The simplest examples derive from the Lotka-Volterra model and have first-order linear interaction terms. Density-dependent regulation of the prey is necessary for system stability.

Predictions
Harvesting the predator increases prey abundance and hence the potential prey yield (May *et al*., 1979). Harvesting the prey reduces predator abundance and the potential predator yield.

Predator-prey models with non-linear interactions

Properties
Non-linear functional responses and time delays in the effects of predation can give rise to multiple equilibrium levels of predators and prey populations. With a type-two predator functional response, there may be two equilibria: a stable upper equilibrium and a lower unstable equilibrium. With a type-three predator functional response, there may be two stable equilibria separated by an unstable equilibrium (Steele and Henderson, 1981). Habitat suitability can be linked to species carrying capacity (Sainsbury, 1991).

Predictions
The effects of fishing are generally the same as for other multispecies production models (see above). In addition, incremental changes in either the predator or prey exploitation rates can cause the system to flip rapidly from one equilibrium level to the other (Collie and Spencer, 1994; Spencer and Collie, 1996). Prey populations can get trapped in 'predator pits' such that recovery may be delayed and require substantial reductions in fishing mortality. Fishing can reduce habitat suitability and selectively reduce the abundance of species which depend on that feature of the habitat (Sainsbury, 1991).

Dynamic spatially structured multispecies models

Properties
Predator-prey dynamics are simulated at small spatial scales. The spatial structure may include prey refugia. The dynamics of the populations can be examined by aggregating the spatial grid into larger 'windows.' Such models can exhibit oscillatory behaviour and multiple equilibria when examined on intermediate spatial scales.

Predictions
Spatial models of this type generally have not explicitly included fishing, but fishing mortality could be added.

Dynamic age/size-structured multispecies models

Properties
These models track the fates of cohorts backward in time (MSVPA, MSGVPA, 4M) or forward from recruitment (Multspec, Bormicon). Natural mortality is a dynamic function of predator abundance and prey availability. Length-based MSVPA removes the dependence on age-length keys by incorporating growth models. In the models with variable growth rates, size at age is a function of food availability, as observed in boreal ecosystems. Models with spatial structure explicitly consider the seasonal overlap between predators and their prey.

Predictions
Harvesting the predators reduces natural mortality of the prey species, resulting in increased prey abundance and increased potential prey yields. In models with variable growth, harvesting the prey will reduce predator growth rates, resulting in reduced predator biomass and potential predator yields. The indirect effects of harvesting one species on other species in the community may be positive or negative depending on the food web structure. Spatially explicit models allow for local depletion of prey abundance ('understocking') even when total prey abundance remains undepleted.

Aggregate ecosystem models

Properties
These models derive from food webs and energy budgets; the units may be energy, carbon or biomass. Species are aggregated into functional groups, especially at the lower trophic levels. The trophic interactions are generally first-order linear. Static mass-balance models (e.g., ECOPATH) assume that the production of each ecosystem component is balanced by losses due to predation, non-predation mortality and export. Dynamic ecosystem models require some form of density dependence to prevent species extinctions. In ECOSIM, this stabilisation is provided by partitioning the functional groups into available and unavailable pools (Walters *et al.*, 1997). Spatially explicit ecosystem models (e.g., ECOSPACE) can model prey availability explicitly and can also define habitat suitability.

Predictions
In static ecosystem models, fishing mortality must be balanced by increased production or reduced predation on that functional group. Dynamic ecosystem models can describe the time dynamics of these changes to a new steady state. In general, harvesting a functional group will reduce its own biomass and the biomass of those groups that feed on it, while increasing the biomass of those groups which are preyed on. These changes can propagate along the food chain, resulting in a trophic cascade (Carpenter *et al.*, 1985).

Age/size-structured ecosystem models

Properties
This class of models is distinguished from the aggregate ecosystem models in that the individual functional groups are generally less aggregated and their dynamics are described with greater temporal resolution. Given the high level of detail, these models are often constructed as simulations and individual based models (e.g., Shin and Cury, 1999) as opposed to statistical models fit to data. Coupled biophysical models that incorporate flow fields concentrate on the lower trophic levels and planktonic life stages.

Predictions
Many of the existing models of this type do not include the upper trophic levels (i.e., fish) and therefore cannot be used to predict the effects of fishing in their present form.

9.5 References

Andersen, K.P., and Ursin, E. 1977. A multi-species extension to the Beverton and Holt theory of fishing, with accounts of phosphorus circulation and primary production. Meddelelser fra Danmarks Fiskeri- og Havundersøgelser. N.S., 7: 319–435.

Baretta, J.W., Ebenhoh, W., and Ruardij, P. 1995. The European regional seas ecosystem model, a complex marine ecosystem model. Netherlands Journal of Sea Research, 33: 233–246.

Basford, D.J., Eleftheriou, A., Davies, I.M., Irion, G., and Soltwedel, T. 1993. The ICES North Sea benthos survey: the sedimentary environment. *ICES Journal of Marine Science*, 50: 71–80.

Basson, M., and Fogarty, M.J. 1997. Harvesting in discrete-time predator-prey systems. Mathematical Biosciences, 141 (1): 41–74.

Batchelder, H.P., and Williams, R. 1995. Individual-based modelling of the population dynamics of *Mertridia lucens* in the North Atlantic. *ICES Journal of Marine Science*, 52: 469–482.

Bianchi, G., Gislason, H., Graham, K., Hill, L., Koranteng, K., Manickchand-Heileman, S., Paya, I., Sainsbury, K., Sanchez, F., Jin, X., and Zwanenburg, K. In press. Impact of fishing on demersal fish assemblages. *ICES Journal of Marine Science*.

Billen, G. 1978. A budget of nitrogen cycling in North Sea sediments off the Belgian coast. Estuarine and Coastal Marine Science, 7: 127–146.

Billen, G., and Lancelot, C. 1988. Modelling benthic nitrogen cycling in temporate coastal ecosystems. *In* Nitrogen Cycling in Coastal Marine Environments, pp. 343–378. Ed. by T.H. Blackburn and J. Sorensen. Wiley and Sons Ltd., London.

Borgman, U. 1987. Models on the slope of, and biomass flow up, the Biomass Size Spectrum. Can. J. Fish. Aquat. Sci., 44(Suppl.2): 136–140.

Brey, T. 1990. Estimating productivity of macrobenthic invertebrates from biomass and mean individual weight. Meeresforschungen, 32: 329–343.

Bryant, A.D., Heath, M.R., Broekhuizen, N., Ollason, J.G., Gurney, W.S.G., and Greenstreet, S.P.R. 1995. Modelling the predation, growth and population dynamics of fish within a spatially-resolved shelf sea ecosystem model. Netherlands Journal of Sea Research, 33: 407–421.

Carpenter, S.R., Kitchell, J.F., and Hodgson, J.R. 1985. Cascading trophic interactions and lake productivity. BioScience, 35: 634–639.

Cohen, E.B., Grosslein, M.D., Sissenwine, M.P., Steimle, F., and Wright, W.R. 1982. Energy budget of Georges Bank. *In* Multispecies Approaches to Fisheries Management Advice, pp. 95–107. Ed. by M.C. Mercer. Canadian Special Publication in Fisheries and Aquatic Science, 59.

Collie, J.S., and Spencer, P.D. 1994. Modeling predator-prey dynamics in a fluctuating environment. Canadian Journal of Fisheries and Aquatic Sciences, 51: 2665–2672.

Connell, J.H. 1975. Some mechanisms producing structure in natural communities. *In* Ecology and Evolution of Communities, pp. 460–490. Ed. by M.L. Cody and J.M. Diamond. Belknap Harvard University Press, Cambridge, Massachusetts.

Cyr, H., and Peters, R.H. 1996. Biomass-size spectra and the prediction of fish biomass in lakes. Can. J. Fish. Aquat. Sci., 53: 994–1006.

Davidson, D.W. 1977. Species diversity and community organization in desert seed-eating ants. Ecology, 58: 711–724.

Death, R.G., and Winterbourn, M.J. 1995. Diversity patterns in stream benthic invertebrate communities: the influence of habitat stability. Ecology, 76: 1446–1460.

Dobby, H., Veitch, A.R., Banks, D., and Gurney, W.S.C. 1999. Incorporating length-based predation into multispecies fisheries modelling. Submitted for publication.

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.

Fenchel, T. 1975. Character displacement and coexistance in mud snails (Hydrobiidae). Oecologia, 20: 19–32.

Fransz, H.G., Mommaerts, J.P., and Radach, G. 1991. Ecological modelling of the North Sea. Netherlands Journal of Sea Research, 28: 67–140.

Fransz, H.G., and Verhagen, J.H.G. 1985. Modelling research on the production cycle of phytoplankton in the Southern Bight of the North Sea in relation to river-borne nutrient loads. Netherlands Journal of Sea Research, 19: 241–250.

Furness. R.W. 1978. Energy requirements of seabird communities: a bioenergetics model. Journal of Animal Ecology, 47(1): 39–53.

Gislason, H. 1999. Biological reference limits for interacting species in the Baltic. *ICES Journal of Marine Science*, 56: 571–583.

Gislason, H., and Rice, J. 1996. Modelling the effect of changes in fishing effort on the size and diversity spectra of exploited fish assemblages. ICES CM 1996/Mini:15.

Gislason, H., and Lassen, H. 1997.On the linear relationship between fishing effort and the slope of the size spectrum. ICES CM 1997/DD:05.

Gobert, B. 1994: Size structures of demersal catches in a multispecies multigear tropical fishery. Fisheries Research, 19: 87–104.

Haedrich, R.L, and Barnes, S.M. 1997. Changes over time of the size structure in an exploited shelf fish community. Fisheries Research, 31: 229–239.

Hall, S.J., and Greenstreet, S.P.R. 2000. Submitted. Diversity, abundance and body size in fish: how robust are the patterns and what do they tell us? American Naturalist.

Heip, C., Basford, D., Craeymeersch, J.A., Dewarumez, J.-M., de Wilde, P., Dorjes, J., Duineveld, G., Eleftheriou, A., Herman, P.M.J., Niermann, U., Kingston, P., Künitzer, A., Rachor, E., Rumohr, H., Soetaert, K., and Soltwedel, T. 1992. Trends in biomass, density and diversity of North Sea macrofauna. *ICES Journal of Marine Science,* 49: 13–22.

Hill, M.O. 1973. Diversity and evenness: a unifying notation and its consequences. Ecology, 54: 427–432.

Hollowed, A.B., Bax, N., Beamish, R., Collie, J.S., Fogarty, M., Livingston, P., Pope, J., and Rice, J. 2000 In press. Are multispecies models an improvement on single-species models for measuring impacts on marine ecosystems? *ICES Journal of Marine Science.*

Huston, A.H. 1979. A general hypothesis of species diversity. American Naturalist, 113: 81–101.

Huston, A.H. 1994. Biological diversity: The coexistence of species on changing landscapes. Cambridge University Press. 681 pp.

ICES. 1996. Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 1996/Assess/Env:1.

ICES. 1997. Report of the Multispecies Assessment Working Group. ICES CM 1997/Assess:16.

Jones, R. 1982. Species interactions in the North Sea. Canadian Special Publication of Fisheries and Aquatic Sciences, 59: 48–63.

Jones, R. 1984. Some observations on energy transfer through the North Sea and Georges Bank food webs. *Rapports et Procès-Verbaux des Réunion du Conseil International pour l'Exploration de la Mer*, 183: 204–217.

Livingston, P.A., and Methot, R. 1998. Incorporation of predation into a population assessment model of eastern Bering Sea walleye pollock. Fishery stock assessment models of the 21st century. Alaska Sea Grant College Program Publication, AK-SG-98-01.

MacArthur, R.H. 1970. Species packing and competitive equilibrium for many species. Theoretical Population Biology, 1: 1–11.

MacArthur, R.H., and Levins, R. 1967. The limiting similarity, convergence and divergence of coexisting species. American Naturalist, 101: 377–385.

MacCall, A.D. 1990. Dynamic geography of marine fish populations. University of Washington Press, Seattle, WA. 153 pp.

May, R.M., Beddington, J.R., Clark, C.W., Holt, S.J., and Laws, R.M. 1979. Management of multispecies fisheries. Science, 405: 267–276.

Menge, B.A., and Sutherland J.P. 1987. Community regulation: variation in disturbance, competition and predation in relation to environmental stress and recruitment. American Naturalist, 130: 730–757.

Mirza, F.B., and Gray, J.S. 1981. The fauna of benthic sediments from the organically enriched Oslofjord, Norway. Journal of Experimental Marine Biology and Ecology, 54: 181–207.

Mommaerts, J.P., Pichot, G., Ozer, J., Adam, Y., and Baeyens, W. 1984. Nitrogen cycling and budget in Belgian coastal waters: North Sea areas with and without river inputs. *Rapports et Procès-Verbaux des Réunion du Conseil International pour l'Exploration de la Mer*, 183: 57–69.

Murawski, S.A., and Idoine, J.S. 1992. Multispecies size composition: a conservative property of exploited fishery systems ? J. Northw. Atl. Fish. Sci., 14: 79–85.

Murphy, E.J., and nine co-authors. 1998. Interannual variability in the South Georgia marine ecosystem: biological and physical sources of variation in the abundance of krill. Fisheries Oceanography, 7: 219–234.

Paine, R.T. 1966. Foodweb complexity and species diversity. American Naturalist, 100: 65–75.

Park, T. 1948. Experimental studies of interspecific competition. I. Competition between populations of the flour beetles *Tribolium confusum* Dival and *Tribolium castaneum* Herbst. Ecological Monographs, 18: 267–307.

Park, T. 1954. Experimental studies of interspecific competition. II. Temperature, humidity and competition in two species of *Tribolium*. Physiological Zoology, 27: 177–238.

Pascual, M., and Levin, S.A. 1999. From individuals to population densities: searching for the intermediate scale of nontrivial determinism. Ecology, 80: 2225–2236.

Patrick, R. 1975. Structure of stream communities. *In* Ecology and Evolution of Communities, pp. 445–459. Ed. by M.L. Cody and J.M. Diamond. Belknap Harvard University Press, Cambridge, Massachusetts.

Pianka, E.R. 1973. The structure of lizard communities. Annual Review of Ecology and Systematics, 4: 53–74.

Pianka, E.R. 1975. Niche relations of desert lizards. *In* Ecology and Evolution of Communities, pp. 292–314. Ed. by M.L. Cody and J.M. Diamond. Belknap Harvard University Press, Cambridge, Massachusetts.

Pianka, E.R. 1983. Evolutionary Biology, 3rd Edition. Harper and Row, New York.

Platt, T., and Denman, K. 1978. The structure of pelagic marine ecosystems. *Rapports et Procès-Verbaux des Réunion du Conseil International pour l'Exploration de la Mer*, 173: 60–65.

Polovina, J.J. 1984. Model of a coral reef ecosystem I. The ECOPATH model and its application to French Frigate Shoals. Coral Reefs, 3: 1–11.

Pope, J.G., and Knights, B.J. 1982. Comparison of the length distributions of combined catches of all demersal fishes in surveys in the North Sea and at Faroe Bank. *In* Multispecies approaches to fisheries management, pp. 116–118. Ed. by M.C. Mercer. Can. Spec. Publ. Fish. Aquat. Sci., 59.

Pope, J.G., Stokes, T.K., Murawski, S.A., and Idoine, S.I. 1988. A comparison of fish size composition in the North Sea and on Georges Bank. *In* Ecodynamics*:* contributions to theoretical ecology, pp. 146–152. Ed. by W. Wolff, C.J. Soeder, and F.R. Drepper.

Rice, J., and Gislason, H. 1996. Changes in the abundance and diversity size spectra of the North Sea, as reflected in surveys and models. *ICES Journal of Marine Science*, 53: 1214–1225.

Rijnsdorp, A.D., van Leewen, P.I., Daan, N., and Heessen, H.J.L. 1996. Changes in abundance of demersal fish species in the North Sea between 1906–1909 and 1990–1995. *ICES Journal of Marine Science*, 53(6):1054–1062

Rijnsdorp, A.D., and van Leewen, P.I. 1996. Changes in growth of North Sea plaice since 1950 in relation to density, eutrophication, beam trawl effort and temperature. *ICES Journal of Marine Science*, 53: 1199–1213.

Rogers, S.I., Maxwell, D., Rijnsdorp, A.D., Damm, U., and Vanhee, W. In press. Comparing diversity of coastal demersal fish faunas in the northeast Atlantic. Fisheries Research.

Rosenzweig, M.L. 1995. Species diversity in time and space*.* Cambridge University Press, UK. 436 pp.

Rosenzweig, M.L., and Abramsky, Z. 1993. How are diversity and productivity related? *In* Species diversity in ecological communities: historical and geographical perspective, pp. 52–65. Ed. by R. Ricklefs and D. Schluter. University of Chicago Press, Chicago, Illinois.

Ruardij, P., Barretta, J.W., and Barretta-Bekker, J.G. 1995. SESAME, a software environment for simulation and analysis of marine ecosystems. Netherlands Journal of Sea Research, 33: 261–270.

Russo, A.R. 1982. Temporal changes in fish community structure near a sewage ocean outfall, Mokapu, Oahu, Hawaii. Marine Environmental Research, 6: 83–98.

Sainsbury, K.J. 1991. Application of an experimental approach to management of a tropical multispecies fishery with highly uncertain dynamics. *ICES Marine Science Symposia*, 193: 310–320.

Sanchez, F., and Olaso, I. 1999. Fisheries impacts in the Cantabrian Sea using a mass-balance model. WGECO 1999/10. 19 pp.

Sheldon, R.W., Prakash, A., and Sutcliffe, W.H., Jr. 1972. The size distribution of particles in the ocean. Limn. Oceanogr., 17: 337–340.

Shin, Y.-J., and Cury, P. 1999. OSMOSE: a multispecies individual-based model to explore the functional role of biodiversity in marine ecosystems. Ecosystem considerations in fisheries management. Alaska Sea Grant College Program, AK-SG-99-01.

Sparre, P. 1991. Introduction to multispecies virtual population analysis. *ICES Marine Science Symposia*, 193: 12–21.

Spencer, P.D., and Collie, J.S. 1996. A simple predator-prey model of exploited marine fish populations incorporating alternative prey. *ICES Journal of Marine Science*, 53: 615–628.

Steele, J.H. 1974. The structure of marine ecosystems. Blackwell, Oxford. 128 pp.

Steele, J.H., and Henderson, E.W. 1981. A simple plankton model. American Naturalist*,* 117: 676–691.

Stefànsson, G., and Pàlsson, Ó.K. 1997. BORMICON. A Boreal Migration and Consumption Model. Marine Research Institute Report 58, Reykjavík. 223 pp.

Thiebaux, M.L., and Dickie, L.M. 1993. Structure of the body-size spectrum of the biomass in aquatic ecosystems: a consequence of allometry in predator-prey interactions. Can. J. Fish. Aquat. Sci., 50: 1308–1317.

Tjelmeland, S., and Bogstad, B. 1998. Multspec—a review of multispecies modelling project for the Barents Sea. Fisheries Research, 37: 127–142.

Tothmeresz, B. 1995. Comparison of different methods for diversity ordering. Journal of Vegetation Science, 6: 283–290.

Vinter, M., and Thomsen, L. 1998. *Ad hoc* multispecies VPA tuning, an extension to 4M. ICES Study Group on Multispecies Model Implementation in the Baltic. Working Document 1998.

Walters, C.J., Christensen, V., and Pauly, D. 1997. Structuring dynamic models of exploited ecosystems from trophic mass-balance assessments. Reviews in Fish Biology and Fisheries, 7: 139–172.

Walters, C.J., Pauly, D., and Christensen, V. 1998. Ecospace: a prediction of mesoscale spatial patterns in trophic relationships of exploited ecosystems, with emphasis on the impacts of marine protected areas. ICES CM 1998/S:4. 22 pp.

Walters, C.J., Stocker, M., Tyler, A.V., and Westreim, S.J. 1986. Interaction between Pacific cod (*Gadus macrocephalus*) and herring (*Clupea harengus pallasi*) in the Hecate Strait, British Columbia. Canadian Journal of Fisheries and Aquatic Sciences, 43: 830–837.

10   Testable Ecological Hypotheses About Fishing Effects

Having reviewed both empirical and model-based approaches to measuring impacts of fishing on ecosystem properties, the next step was to refine what hypotheses should be tested. WGECO considered it important that the evaluation of impacts be conducted, to the fullest extent possible, in a hypothesis testing framework, rather than just as an exercise in describing patterns in data or model results.

10.1   Development of Testable Hypotheses for Evaluating which Components of the Marine Ecosystem are Most Vulnerable to Trawl Impacts

WGECO was concerned with the apparent circularity in logic prevalent in the identification of vulnerable species. Rarely were such species proposed *a priori* on the basis of theoretical expectations. More commonly, vulnerable species were identified on the basis of their observed response to changes in fishing pressure. This circularity leads to uncertainty in the interpretation of data, and contributes to a degree of confusion regarding the value of particular species, or particular traits, as indicators of the impact of fishing on the ecosystem.

A more useful approach might be to first try to classify species on the basis of life history characteristics in a ranking order for vulnerability. Then one would test the hypothesis that those species which have been classified as the most vulnerable have actually shown a decline in response to fishing and vice versa. Such a classification should take place before any analysis of the data. Results could be taken a step further to address the question "can mean-values of specific characteristics of these populations, weighted by species abundance, be used to monitor the effects of fishing on the entire fish community?".

A list of characteristics and traits for which it was thought there could be sound theoretical, or common sense, grounds for being able to predict a clear directional response to variation in fishing impact is developed below. The behaviour of as many of these characteristics and traits in some real data sets has been examined in Section 9.2, but will be tested formally in Section 10.4. Both spatial and temporal analyses were undertaken, looking for differences among areas differing in the level of fishing disturbance to which they have been subjected, and variation over time in areas where fishing impact has either increased or decreased. The analyses have been restricted to the potential impacts of fishing on fish species, and on trends in the mean characteristic value determined for the entire fish community or sub-components of it. A similar approach could be adopted for other components of the marine ecosystem, such as birds, marine mammals, invertebrates and benthic communities.

The effect of fishing on life history characteristics within a particular species is not considered here. (For example, does the age at maturity of individuals within a fished cod population decline?) This is another complex problem which, although of great interest, was beyond the scope of the WG to address in the time available.

10.2   Specific Hypotheses Regarding the Impact of Fishing on the Characteristics and Traits of Fish Communities

In this section many traits and characteristics of both species and fish communities are considered, *a priori* hypotheses are stated, and other initiatives are set out.

10.2.1   Specific hypotheses about populations and species abundances

When trying to characterise species based on life history characteristics there is extensive literature that distinguishes K-strategists from r-strategists on theoretical grounds and how their relative abundance in a community depends on the stability of their environment. This division is debatable and the categorisation of species into K- and r- strategists in this section is heuristic, and should not be interpreted as WGECO accepting the unqualified validity of this approach. Rather, the section explores how life history reasoning can be used in interpreting the results of studies of impacts of fishing. We ask, "Given the theoretical framework, can predictions of fishing effects be made and tested on a more *a priori* basis?"

According to this paradigm, K-strategists are adapted to living in stable and predictable environments and have greater competitive ability. They have longer life-spans, larger body size, reproduce later, produce few young, and are more likely to exhibit parental care. In contrast, r-strategists live in unpredictable or disturbed environments. They are small organisms with short life-spans, early reproduction and high fecundity. All gradations between the two extremes are possible and in practice it may be difficult to characterise a particular species as belonging to either strategy.

Therefore the approach chosen here was to use the values of life history parameters to indicate a species' position on the r/K continuum.

Fishing pressure increases mortality in all species. When this increase in mortality is placed in the context of life history theory it may have several consequences. Life history traits that change as a response to fishing will change in the same direction for all species, but the rates of change in the life history parameters should differ in important and informative ways. For a specific level of fishing mortality, populations of species with K-selected traits will decline faster than species with r-selected traits. Furthermore the life history characteristics will change faster for species at the K-end of the continuum. Thus the predicted responses by individual species to increased fishing disturbance are expressed below as testable hypotheses relative to what would be expected for r-selected species:

- Species with large ultimate body length (L_{max} or L_{inf}) should decline;
- Species with slow growth rates (e.g., k from the von Bertalanffy equation) should decline;
- Species with older age at maturity (A_{mat}) should decline;
- Species with longer length at maturity (L_{mat}) should decline;
- Species with a low fecundity and lower life-time reproductive output should decline.

For a given level of fishing mortality, at the community scale the percentage of the community composed of species with K-selected traits will decline. Because of responses predicted for individual species, the response of each of these characteristics calculated across the assemblage as a whole is predictable. The community average character values, weighted by species abundance, should respond to an increase in fishing disturbance as follows:

- L_{max} Decrease
- L_{inf} Decrease
- Growth Rate   Increase
- Fecundity   Increase
- Life-time reproductive output   Increase
- A_{mat} Decrease
- L_{mat} Decrease

These changes in growth rate and fecundity should affect the productivity of the fish assemblage. Thus:

- The overall production to biomass (P/B) ratio of the fish community should be higher in more intensively fished areas, and it should increase as fishing disturbance increases.

The trophic level at which fish feed is strongly size dependent; larger fish in the community tend to be piscivores, smaller fish are planktivores and/or benthivores. With the decline of larger fish in more heavily fished areas, or as fishing in an area increases, the trophic structure of the community should change (e.g., Pauly *et al.,* 2001).

- Species that feed at higher trophic level will be more sensitive and should decline as fishing effort increases, or have a lower abundance in heavily fished areas.
- The average trophic level of the fish community should decline as fishing intensity increases, and be lower in more heavily fished regions.

Because of an increase in the amount of damaged and killed benthic organisms left lying on the seabed as a consequence of demersal fishing, species best able to utilise this resource are likely to increase in abundance.

- The proportion of fish that can be considered scavengers should have increased in intensively fished areas.

Species with obligate habitat requirements should decline in abundance when such habitat is lost as a consequence of fishing activity.

- Species that depend on a three-dimensional habitat (e.g., a fragile biogenic habitat) should decline in abundance and have a lower abundance in areas where habitat is altered by increased levels of trawling.

WGECO 2000 suggested that species richness should decline more in intensively fished areas than in less disturbed areas. This can be tested in two ways:

- Spatially, species richness should be lower in areas of high fishing intensity;
- Temporally, species richness should decline in areas where fishing intensity is increasing.

Many factors could confuse the response of species diversity to changes in fishing levels. Huston's (1994) dynamic equilibrium model suggests that the response of species diversity to disturbance is dependent upon local productivity.

- Species diversity should decline in response to increased fishing disturbance in areas of low productivity. In areas of high productivity, increased fishing could cause species diversity to increase.

Fish also pass through the meshes of the gear and can become damaged in the process, which could increase mortality and susceptibility to disease.

- The prevalence of fish showing sub-lethal effects (scarring, scale loss, external lesions, etc.) in intensively fished areas should be higher compared with fish in relatively undisturbed regions.
- Species which are particularly sensitive to the effects of scale loss, etc., are likely to decline in abundance as fishing intensity increases, and to have lower abundance in areas of high fishing activity, relative to insensitive species.

10.2.2 Spatial Hypotheses

Many attributes of the two- and three-dimensional distribution of species or groups of species have not been well explored. This section attempts to develop some spatial metrics and apply them in a provisional way to fisheries survey data. In Atlantic Canada two metrics of distribution are commonly reported in single-species assessments: the area covered by a species and an index of concentration, which is the area containing the densest portion of the resource (Branton and Black, 2000).

The proposed metrics are applicable to single-species and are aggregated into community or group indices. It is not clear, or perhaps even likely, that the aggregated indices will be more valuable than those for specific single-species. As the work is exploratory it is offered as a stimulus to further work as opposed to a definitive or proscriptive study.

There is an implicit hypothesis that the distribution of animals may affect their viability and further that some species will be more sensitive to displacement than others. Further if species are perturbed, their community may also be affected. For example, if they are scattered too widely, they may be subject to higher predation or compromised recruitment. Conversely, if they are concentrated into a small area, they could suffer increased fishing mortality per unit effort (Paloheimo-Dickie effect (Paloheimo and Dickie, 1964)). Another consideration is displacement from traditional spawning areas. It is further assumed that fishing activity (or pollution) could affect these distributions. At this time hypotheses about which species or species groups will be most affected have not been posed.

10.3 Approach

The analysis performed by WGECO had two purposes:

Firstly, to use the most comprehensive data available to test a set of specific hypotheses with the purpose of identifying those characteristics and traits of fish species and communities that might be most useful as metrics of trawling impact. This required three basic types of information:

1) Trawl survey data providing information on the species' abundance in samples of fish. These data must extend over sufficient time so as to have substantial contrast in fishing events if the hypotheses being tested involve the evaluation of temporal trends. Alternatively, if the tests involve spatial comparisons, a reasonably large and dynamic geographic ranged is required.
2) Information on species characteristics or traits, for example, age at maturity, or habitat requirements of the species caught in the trawl surveys. Such information is required for a sufficient number of species so as to ensure that a reasonably large fraction of the total number of individuals caught in each sample are included.
3) Information on the variation in fishing effort, over time and/or space, is necessary for directly testing hypotheses. Ideally the temporal and/or spatial extent of the data should match that/those of the groundfish survey data.

Secondly, to explore the potential use of these metrics as indicators of impending ecological problems for managers. Thus temporal and spatial trends in the abundance of potentially sensitive species were explored, as well as trends in the metrics calculated for the entire fish community in an area where fishing effort data were not available. The reasoning behind this analysis was to explore whether the fish community concerned was affected by fishing activity in the area.

10.4 Analysis of the Data Sets

10.4.1 Northwest North Sea (Scottish August groundfish surveys)

In this section we use Scottish August Groundfish Survey (SAGFS) data, international and Scottish fishing effort data, and information on life history characteristics of the species encountered in the survey data, to test some specific hypotheses derived from the theoretical expectations presented in Section 10.2. A primary objective of the section is to identify which, if any, of the life history parameters examined might hold potential as a metric of fishing-induced change in the fish community, and the particular circumstances where the greatest insight might be gained. The data

presented cover 75 ICES statistical rectangles located in the northwestern North Sea where data coverage is most reliable (Figure 10.4.1.1).

10.4.1.1 Species characteristics

Information regarding four life history characteristics (L_{inf}, Growth Rates, A_{mat}, L_{mat}) was available for 32 of the species (Jennings *et al.*, 1998, 1999a) encountered in the SAGFS, listed in Table 10.4.1.1.1. The L_{inf} and Growth Rate were the parameter values determined from the von Bertolanffy growth equation calculated for each species. The von Bertolanffy parameter is not strictly a rate value, but is used here as an index equivalent to growth rates. A_{mat} and L_{mat} values were determined by observation, either from recent survey data or with recourse to the literature. These 32 species accounted for over 99% of the individuals sampled by the SAGFS in each of the 75 statistical rectangles. No life history characteristic information was available for the remaining 24 species included in the database. These species were among the rarest sampled, and combined they represented less than 1% of the total number of individuals sampled in any rectangle. Their influence on the mean value of each characteristic could only have been negligible. For the purposes of this analysis, therefore, abundance data for these species were excluded. In the final temporal analysis, the proportion of the sampled fish assemblage in any time-period/"treatment" cell never dropped below 98.5%.

10.4.1.2 Effort

International otter trawl, beam trawl, and Seine net fishing effort (hours fished) for the period 1990 to 1995 were available from the database compiled as part of the EC "Monitoring Biodiversity…" project (Jennings *et al.*, 1999b, 2000). Average annual effort values were calculated to provide estimates of the spatial distribution of fishing effort across the 75 ICES statistical rectangles for which groundfish survey data were available. Total annual average fishing effort across the 75 statistical rectangles amounted to 963 216 hours of fishing, 67% of which consisted of otter trawling, 12% beam trawling, and 21% Seine netting.

The possibility that the life history composition of the groundfish assemblage was affected not only by the absolute amount of fishing effort in any statistical rectangle, but also by recent trends, was also considered. The international fishing effort database covered only the years 1990 to 1995 and so does not provide much of a time-series. The Scottish fishing effort database extends further back in time (Greenstreet *et al.*, 1999b). Furthermore, Scottish vessels landing in Scotland account for most of the fishing effort in this part of the North Sea. Indices of annual rates of change in otter trawl, beam trawl, and Seine net effort were therefore determined for each of the 75 statistical rectangles using the Scottish data. Effort data for Seine net and otter trawl were available for the period 1970 to 1994 for each rectangle. Average annual effort for the five-year periods 1970 to 1974 and 1990 to 1994 were computed for both gears. The difference between these values was divided by 20 to provide average annual rates of change for each gear in each rectangle over the 25-year period. Beam trawling is a relatively recent phenomenon in the northwestern North Sea, and effort data for this gear were only recorded from 1984 onwards. The same approach described above was adopted, except that the start point five-year period was 1984 to 1989, and the divisor was 5, thus providing an annual rate of change in beam trawl use index over a ten-year period.
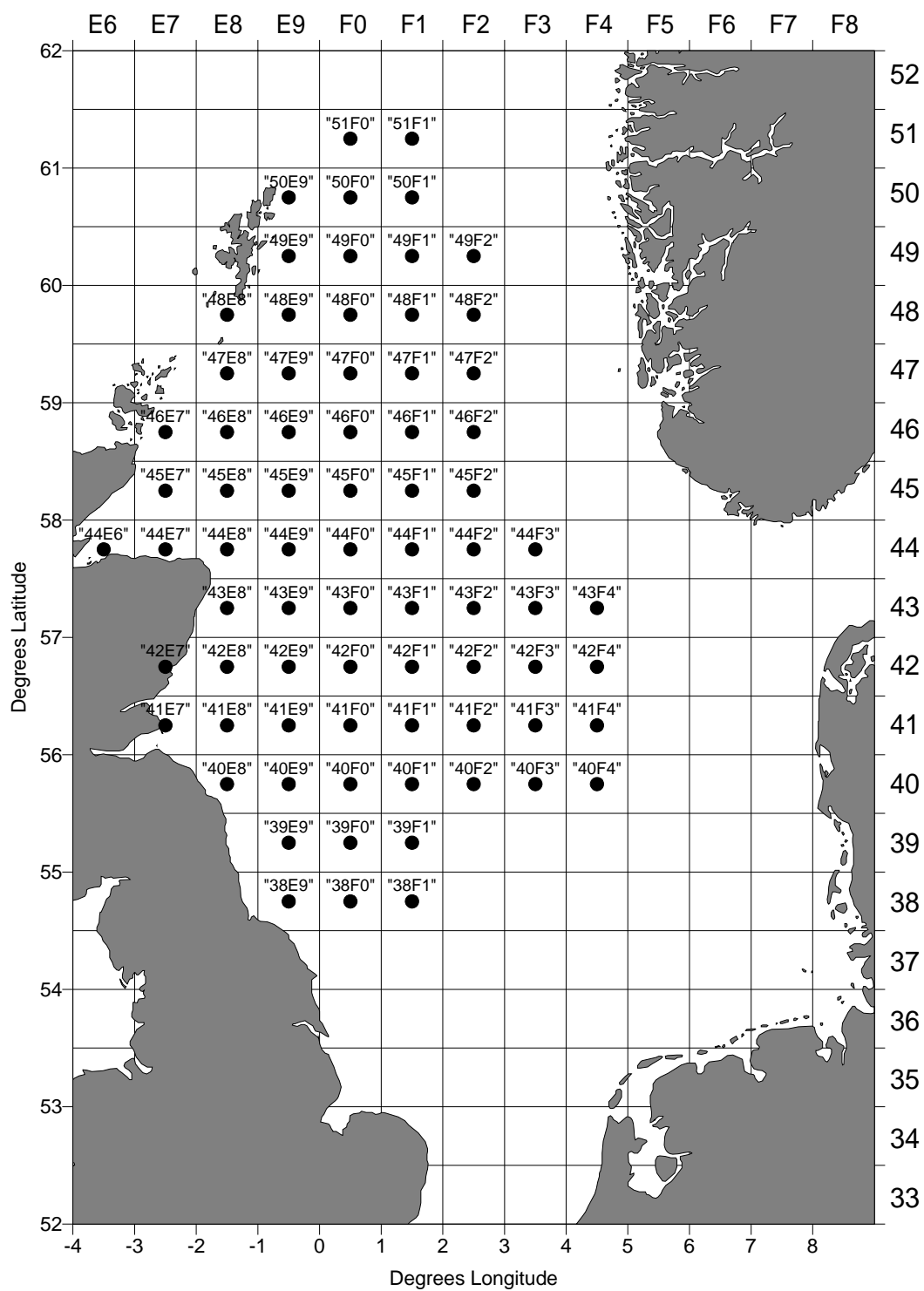
Figure 10.4.1.1   Area of the North Sea covered by the data sets analysed in this section.

Table 10.4.1.1.1   List of species encountered in the SAGFS for which life history character information was available.

Angler	*Lophius piscatorius*
Bull rout	*Myoxocephalus scorpius*
Catfish	*Anarhichas lupus*
Cod	*Gadus morhua*
Common dab	*Limanda limanda*
Cuckoo ray	*Raja naevus*
Dover sole	*Solea solea*
Dragonet	*Callionymus lyra*
Four-bearded rockling	*Enchelyopus cimbrius*
Grey gurnard	*Eutrigla gurnardus*
Haddock	*Melanogrammus aeglefinus*
Hake	*Merluccius merluccius*
Halibut	*Hippoglossus hippoglossus*
Hooknose	*Agonus cataphractus*
Lemon sole	*Microstomus kitt*
Lesser spotted dogfish	*Scyliorhinus canicula*
Long rough dab	*Hippoglossoides platessoides*
Megrim	*Lepidorhombus whiffiagonis*
Norway pout	*Trisopterus esmarki*
Plaice	*Pleuronectes platessa*
Poor cod	*Trisopterus minutus*
Saithe	*Pollachius virens*
Skate	*Raja batis*
Spotted ray	*Raja montagui*
Spurdog	*Squalus acanthias*
Starry ray	*Raja radiata*
Thornback ray	*Raja clavata*
Three-bearded rockling	*Gaidropsarus vulgaris*
Torsk	*Brosme brosme*
Turbot	*Scophthalmus maximus*
Whiting	*Merlangius merlangus*
Witch	*Glyptocephalus cynoglossus*

10.4.1.3 Survey (catch) data

Groundfish survey data collected as part of the Scottish August Groundfish Surveys were examined. Trawl species abundance data were extracted for 75 ICES statistical rectangles in the northwestern North Sea covering a period of 14 years from 1983 to 1996. Up to four trawl samples were then excluded as necessary in order to reduce the number of samples to 10 in all rectangles. For one rectangle, data for the years 1983, 1985, 1987 and 1995 were missing. In reducing the number of trawls to 10 in the remaining rectangles, samples from these years were selected and deleted at random as required. All ten trawl samples in each rectangle were then pooled to provide a single aggregated sample for each rectangle. The samples for each rectangle were thus standardised as far as possible, given the type of sampling involved. All trawl samples were collected by the FRV "Scotia (II)", using a 48-foot Aberdeen otter trawl, towed for one hour (Greenstreet and Hall, 1996; Greenstreet *et al.*, 1999a). The number of trawl samples per rectangle was the same for all rectangles, thus avoiding any sample-size dependency issues.

Species abundance data were converted to the number of individuals with particular characteristic values, and the mean value for each characteristic for each rectangle was computed. Data were available only for groundfish species likely to be well sampled by the gear. Pelagic species and other species not well sampled by the 48-ft Aberdeen otter trawl, such as herring, sprat, and sandeels, were all excluded from the data set. The results therefore only apply to the demersal groundfish community occupying the area.

For the final analysis, looking at long-term temporal trends in rectangles varying in the level of fishing effort to which they had been subjected, data from the full time-series were used. As in Greenstreet *et al.* (1999b) data were pooled into groups of two or three years to ensure adequate sampling effort in each time-period/"treatment" cell.

10.4.1.4 Analysis and results

This section is hypothesis driven. A series of specific hypotheses are presented, all of which are related to or derived from the theoretical discussion presented in Section 10.2. The data are then analysed so as to test each hypothesis.

Hypothesis: Groundfish assemblage Growth Rates should be positively correlated, and L_{inf}, L_{mat} and A_{mat} should be negatively correlated in space, with fishing effort.

Life history charactistic data were plotted against international effort data. Relationships between assemblage average life-history characteristics and otter trawl and beam trawl effort in each of the 75 statistical rectangles were similar. Also, since otter trawl effort exceeded beam trawl effort by a factor of six, data for both gears were aggregated. The effects of the two gears combined on each of the life history characteristics were then examined (Figure 10.4.1.4.1). Correlation coefficients were computed and used as a guide to identify life history characteristics that could potentially be useful as metrics indicative of an effect of fishing on groundfish communities. All four life history characteristics responded to increased trawling effort in a manner predicted by our hypothesis; however, only the correlations for A_{mat} and L_{mat} were significant at the 5% level. Furthermore, some caution is required in interpreting the significance of any of the correlations presented in this section, since the degrees of freedom applied take no account of the fact that these are essentially spatial analyses. Because of the strong possibility of spatial auto-correlation between many of the data points used, rendering them not truly independent of each other, the actual number of degrees of freedom is likely to be less than, in this case, 73.
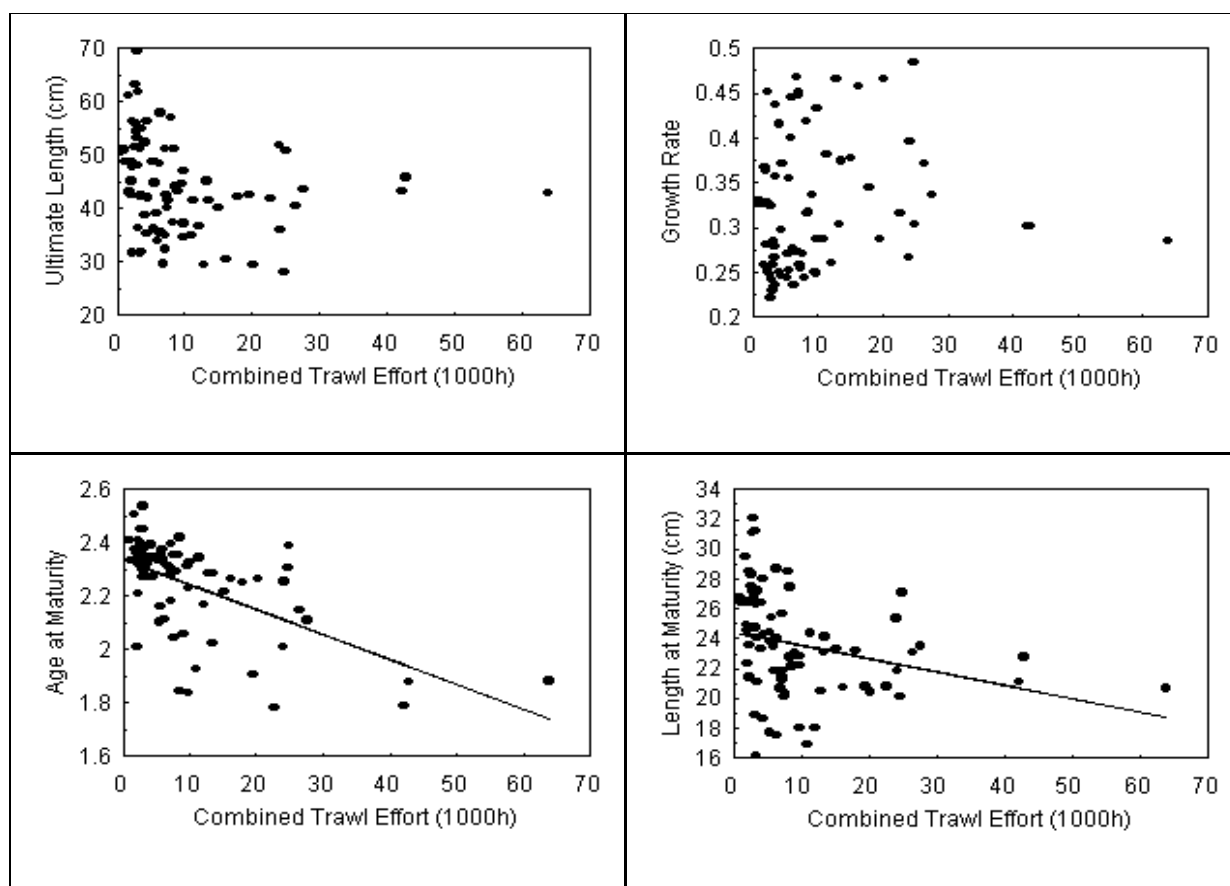
Figure 10.4.1.4.1   Relationship between combined otter and beam trawl effort in 75 ICES statistical rectangles and the average and Length$_{Infinity}$, Growth Rate, Age$_{Maturity}$ and Length$_{Maturity}$ determined for 32 species making up >99% of the total number of individuals sampled in each rectangle.

By the early 1990s, use of Seine net gear in this part of the North Sea had declined from the high levels characteristic of the 1960s. Nevertheless, the relationship between spatial variation in Seine net effort and the life history characters of the groundfish assemblage in each statistical rectangle was also explored (Figure 10.4.1.4.2). Three characters were correlated at the 5% level of significance: Growth Rate, A_{mat} and L_{mat}. However, only the relationship for Growth Rate was in the direction predicted by our hypothesis. Seine net used to be the predominant type of gear used in a large part of the northwestern North Sea. Over recent decades this gear has largely been replaced by otter trawls. It is possible that the relationships displayed in Figure 10.4.1.4.2 have been influenced by this change in fishing practice, such that the relationships between any character and Seine net use have been affected by the impact of otter trawling in the same rectangles.
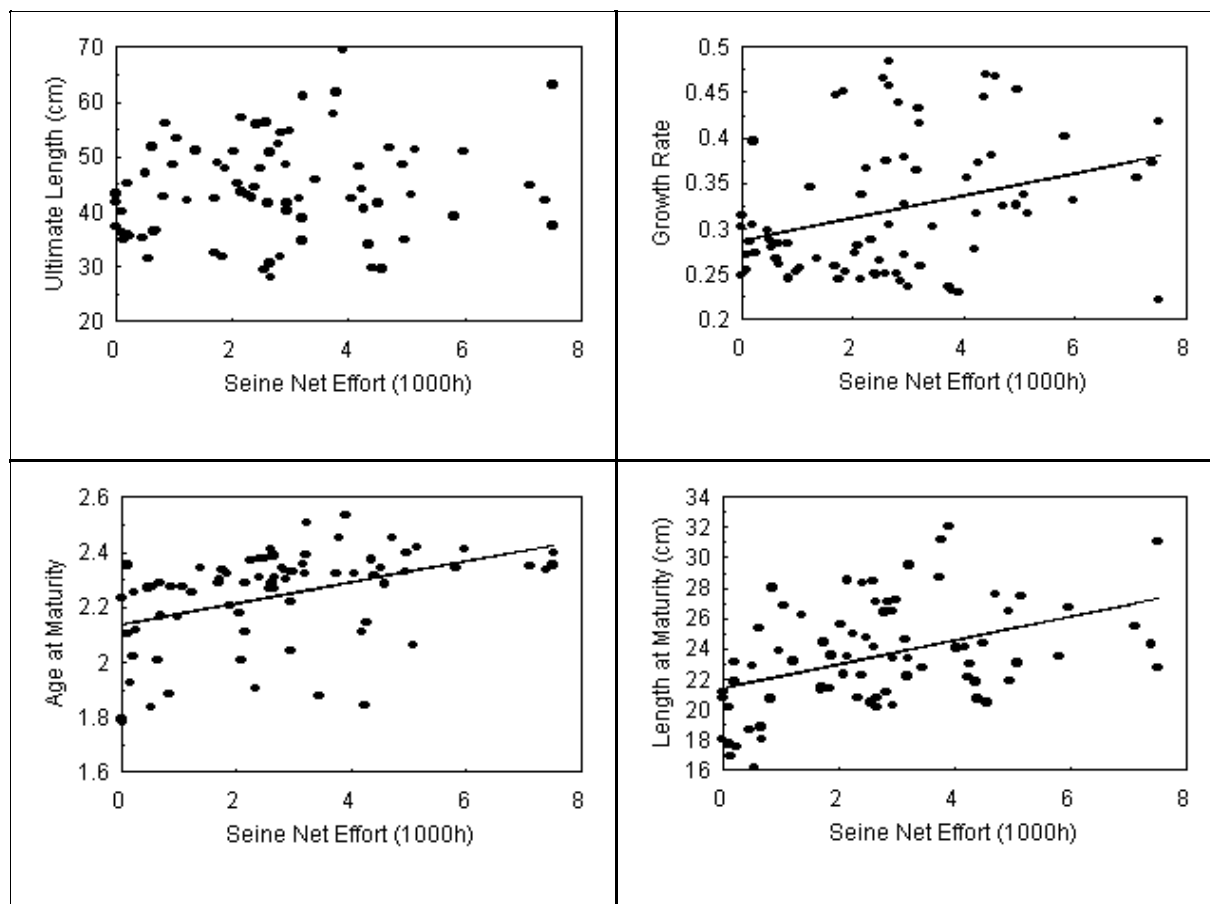
Figure 10.4.1.4.2.   Relationship between Seine net effort in 75 ICES statistical rectangles and the average and Length$_{Infinity}$, Growth Rate, Age$_{Maturity}$ and Length$_{Maturity}$ determined for 32 species making up >99% of the total number of individuals sampled in each rectangle.

Hypothesis: Groundfish assemblage Growth Rates should be positively correlated, and L_{inf}, L_{mat} and A_{mat} should be negatively correlated in space with rates of change in fishing effort over recent years.

Relationships between mean assemblage life history characteristics and annual rates of change in fishing effort in each rectangle were examined (Figure 10.4.1.4.3). The correlations for the L_{inf} and Growth Rate were significant at the 5% level, whilst those for A_{mat} and L_{mat} were not. These two sets of results raise the possibility that assemblage mean L_{mat} and A_{mat} might provide indicators of the effect of absolute levels of fishing effort on the life history composition of the groundfish communities, while mean L_{inf} and Growth Rate could reflect recent changes in fishing effort. However, a small number of points have very high leverage in these calculations, so patterns must be viewed with extra caution.
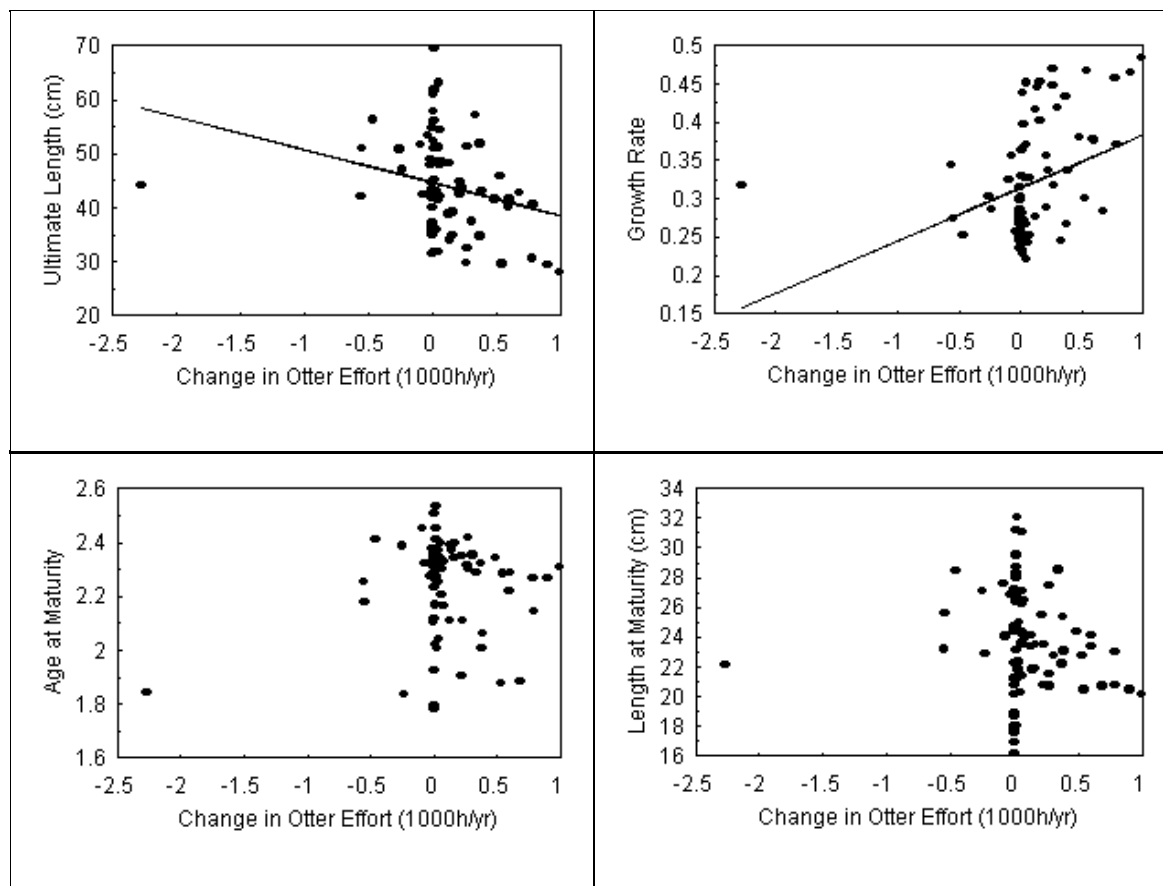
Figure 10.4.1.4.3   Relationship between annual average rate of change in otter trawl effort in 75 ICES statistical rectangles over the period 1970–1994 and the average and Length$_{Infinity}$, Growth Rate, Age$_{Maturity}$ and Length$_{Maturity}$ determined for 32 species making up >99% of the total number of individuals sampled in each rectangle.

Hypothesis: Groundfish assemblage Growth Rates should be higher, and L_{inf}, L_{mat} and A_{mat} should be lower in areas with higher fishing effort.

Levels of international otter trawl fishing effort ranged from 645 hr yr^{-1} to 63 794 hr yr^{-1} across the 75 ICES statistical rectangles. The rectangles were sorted into three groups varying in the intensity to which they had been fished during the early 1990s: a low-intensity group of 40 rectangles where effort varied from 0 to 4999 hr yr^{-1}; a medium-intensity group of 25 rectangles in which effort varied from 5000 to 19 999 hr yr^{-1}; and a heavily fished group of 10 rectangles in which effort exceeded 20 000 hr yr^{-1}. The mean, and standard error of the mean, of each life history characteristic was determined for each group of rectangles (Figure 10.4.1.4.4). Differences, tested using one-way ANOVA, were found to be significant at the 1% level for all four characteristics. In each case, the trend was consistent with the hypothesis.
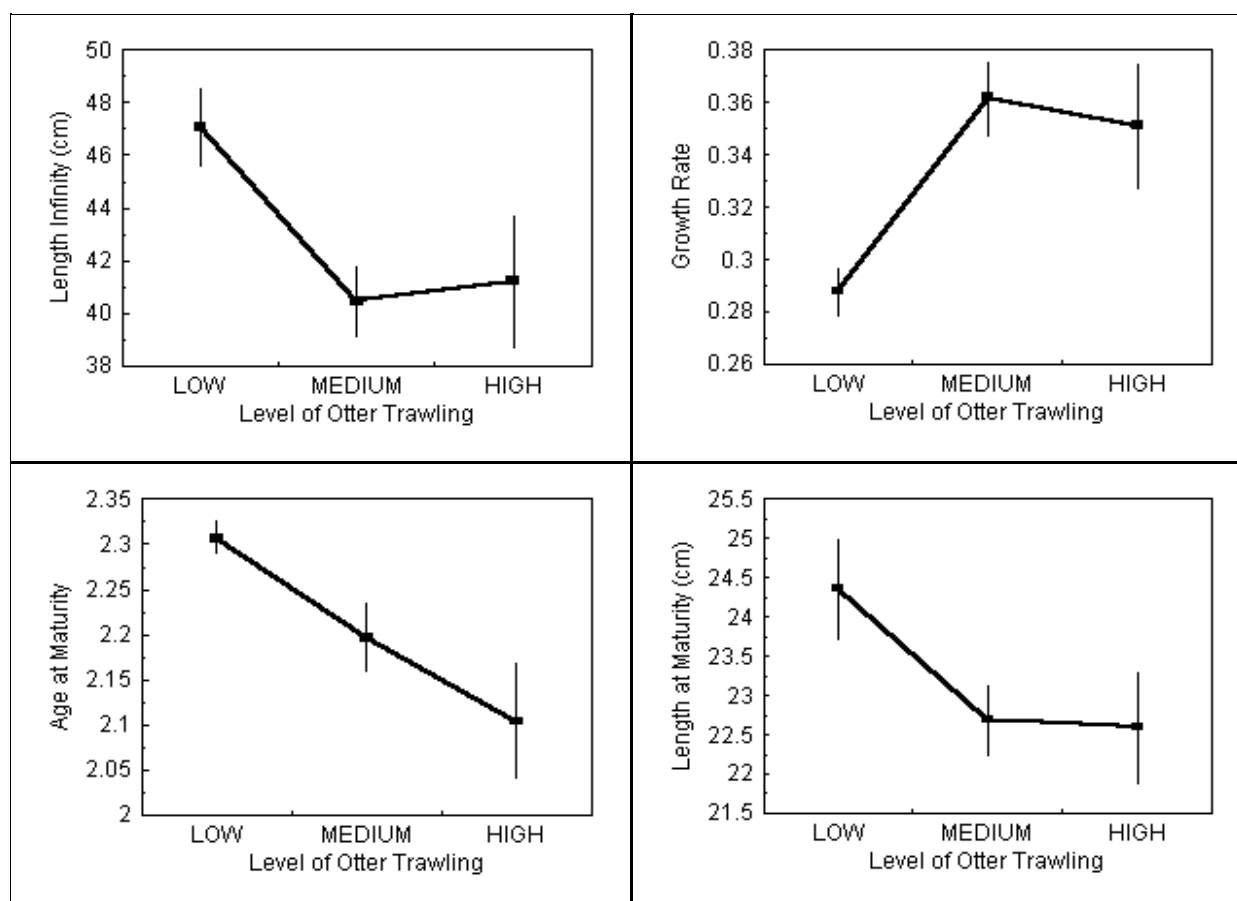
Figure 10.4.1.4.4   Variation in the mean ($\pm$ 1 S.E.) Length$_{Infinity}$, Growth Rate, Age$_{Maturity}$ and Length$_{Maturity}$ determined for 32 species making up >99% of the total number of individuals sampled in each rectangle calculated for groups of rectangles varying in the level of otter trawl effort to which they were subjected.

L_{inf}, Growth Rates and L_{mat} appeared to be the most sensitive characteristics, differentiating most between low and medium fishing intensity groups of rectangles. Beyond a certain level (threshold?) of perturbation, variation in these characteristics tended to level out. This raises the possibility that, as metrics, these three characteristics may be relatively insensitive in already disturbed areas. On the other hand, A_{mat} continued to decrease strongly as otter trawl effort increased from low, through medium, to high levels.

Hypothesis: Groundfish assemblage Growth Rates should be higher, and L_{inf}, L_{mat} and A_{mat} should be lower in areas where fishing effort is increasing at the greatest rate.

Annual rates of change in otter trawl effort varied from the extreme outlier of -2268 hr yr^{-1} to 991 hr yr^{-1}. Three groups of rectangles were again defined: a group of 19 rectangles where otter trawl effort was declining; a group of 35 rectangles where effort was increasing slowly, between 0 and 199 hr yr^{-1}; and a group of 21 rectangles where effort was increasing rapidly, between 200 and 991 hr yr^{-1}. The mean, and standard error of the mean, of each life history characteristic was determined for each group of rectangles (Figure 10.4.1.4.5). Differences, tested using one-way ANOVA, were found to be significant at the 1% level for all four characteristics. Only the Growth Rate behaved entirely as anticipated by the hypothesis. For the three remaining characteristics, the highest mean parameter values were observed on the rectangles with slow rates of increase in otter trawl effort.
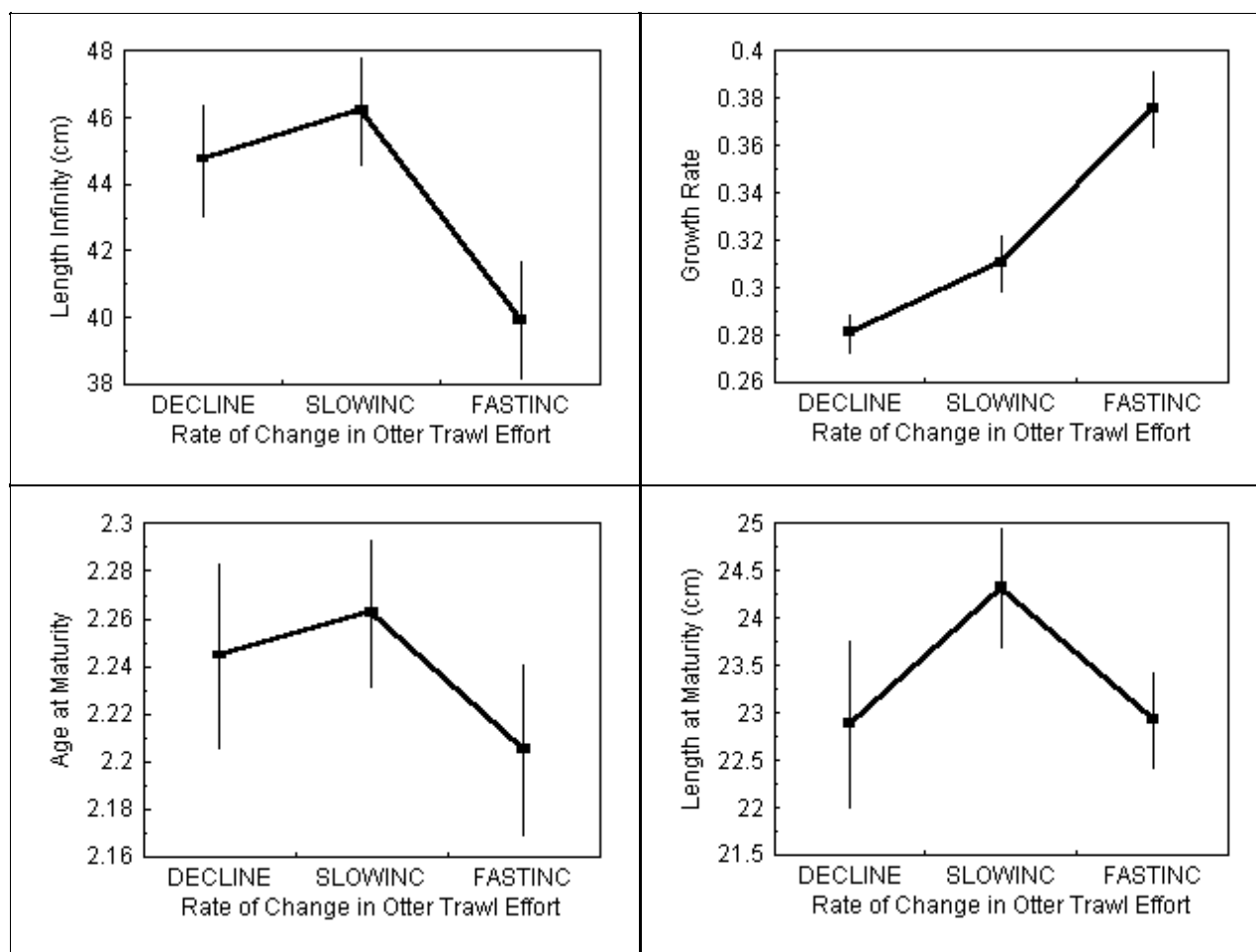
Figure 10.4.1.4.5. Variation in the mean ($\pm$ 1 S.E.) Length$_{Infinity}$, Growth Rate, Age$_{Maturity}$ and Length$_{Maturity}$ determined for 32 species making up >99% of the total number of individuals sampled in each rectangle in groups of rectangles with different annual rates of change in otter trawl effort.

Hypothesis: Long-term temporal variation in groundfish assemblage Growth Rates should show steeper positive trends, and L_{inf}, L_{mat} and A_{mat} steeper negative trends, in areas where fishing effort is higher, and in areas where recent trends in fishing effort have shown the greatest rates of increase.

This analysis used the full time-series of available groundfish survey data, from 1925 to 1996, to explore the long-term behaviour of community mean life history characteristics in areas of varying fishing intensity. The rectangles were grouped into the same three treatment levels of international otter trawl effort during the early 1990s, and for recent trends in Scottish otter trawl effort over the period 1970 to 1994. As before, abundance-weighted mean character values for the groundfish community were determined for each time/treatment cell. These were then regressed over time and the regression coefficients (± 1 S.E. of the coefficient) were plotted for each treatment and life history characteristic (Figures 10.4.1.4.6 and 10.4.1.4.7).
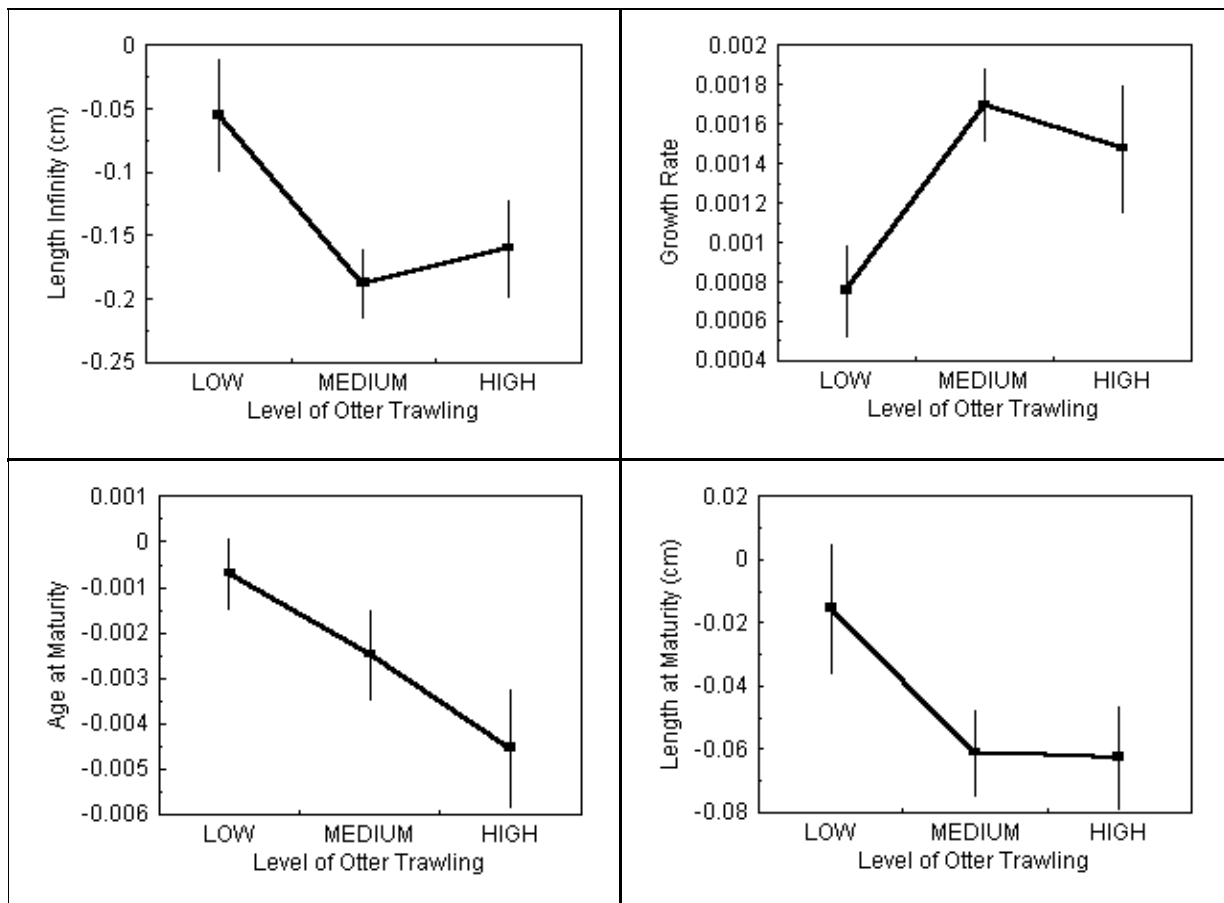
Figure 10.4.1.4.6   Variation in the regression coefficients ($\pm$ 1 S.E. of the coefficient) for the slopes of Length$_{Infinity}$, Growth Rate, Age$_{Maturity}$ and Length$_{Maturity}$ over the time period 1925 to 1995 in rectangles with different mean levels of annual international otter trawl effort over the period 1990 to 1995.

All four parameters showed very little change in rectangles where levels of otter trawl impact were low. Indeed, none of the long-term regression analyses were significant. However, in rectangles with medium and high levels of international otter trawl effort during the early 1990s, all the long-term trends were significant, and in the direction predicted by our hypotheses (Figure 10.4.1.4.6). Of interest again was the fact that L_{inf}, Growth Rate and L_{mat} all failed to differentiate between medium and high levels of fishing effort. Again, this suggests that these parameters may be able to distinguish between fished and unfished areas, but once an area is impacted, they may be relatively insensitive to further perturbation. A_{mat}, however, showed increasingly steep long-term declines as otter trawl effort increased from medium to high levels of otter trawl activity. This analysis therefore again suggests that this index may hold the greatest promise as a metric able to provide managers with an ongoing indication of the continuing effect of their actions on the life history composition of the groundfish community.

The data presented in Figure 10.4.1.4.6 are also helpful in aiding our interpretation of Figure 10.4.1.4.4 as they suggest that the current community-averaged life history parameters (e.g., Figure 10.4.1.4.4) are the result of long-term changes from some earlier common, presumably near pristine, state.
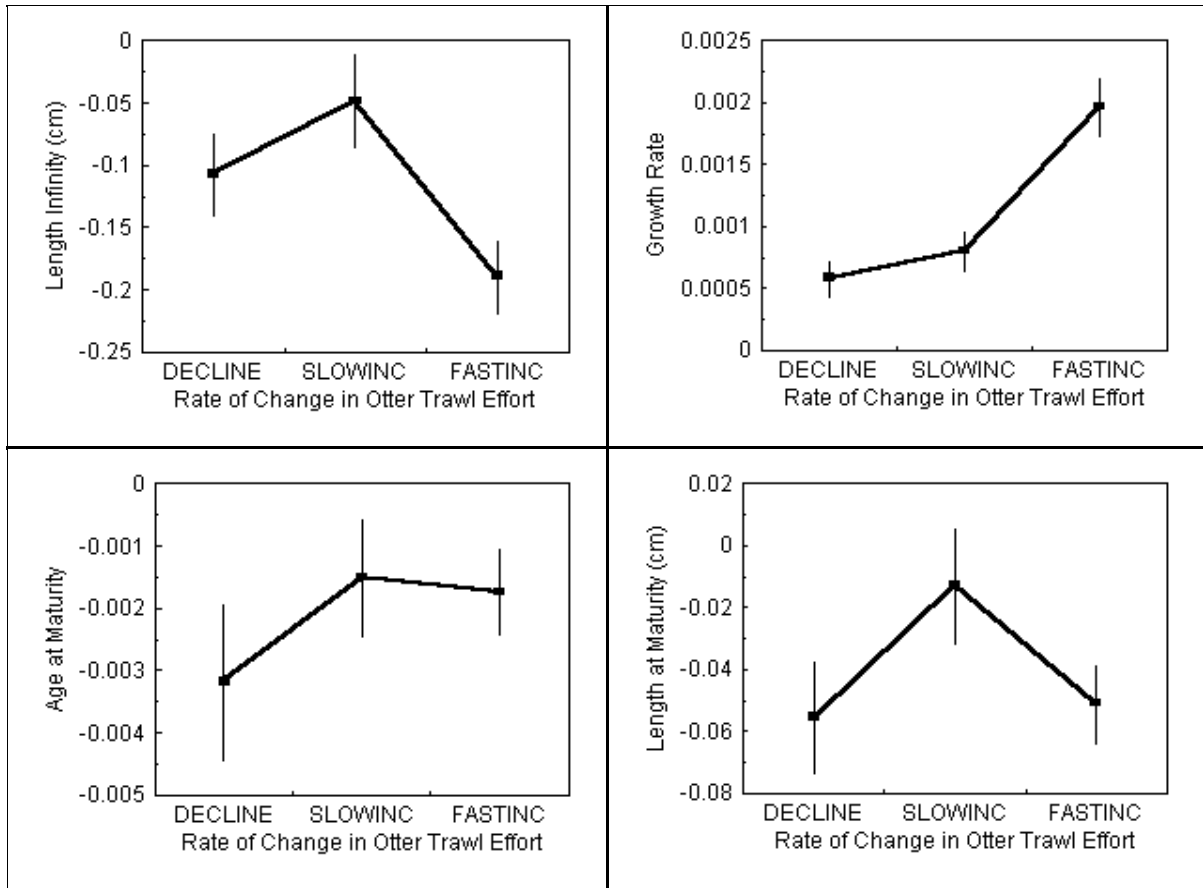
Figure 10.4.1.4.7   Variation in the regression coefficients (± 1 S.E. of the coefficient) for the slopes of Length$_{Infinity}$, Growth Rate, Age$_{Maturity}$ and Length$_{Maturity}$ over the time period 1925 to 1995 in rectangles with different annual rates of change in Scottish otter trawl effort over the period 1970 to 1994.

As with Figure 10.4.1.4.5, the interpretation of Figure 10.4.1.4.7 is more problematic. For example, Figure 10.4.1.4.7 suggests that the greatest long-term rates of decline in both A$_{mat}$ and L$_{mat}$ occurred in rectangles where otter trawl activity has actually declined over the period 1970 to 1994. This clearly contravenes the hypothesis. L$_{mat}$ does show steeper long-term declines in rectangles where otter trawling has increased most rapidly over the period 1970 to 1994. A$_{mat}$ fails even to do this. Variation in L$_{inf}$ and Growth Rate, however, both support the hypothesis. The long-term decline in L$_{inf}$ and long-term increase in Growth Rate are both steepest in the rectangles where otter trawling has increased most rapidly over the period 1970 to 1994.

Hypothesis: Species richness should be lower in areas where current levels of fishing effort are highest, and in areas where recent trends in fishing effort have shown the greatest increase.

Species richness was determined for each of the statistical rectangles from simple counts of all the different species recorded in each rectangle. Mean species richness was determined for three groups of rectangles with low (0 to 4999 hr yr^{-1}), medium (5000 to 19 999 hr yr^{-1}), and high levels (>20 000 hr yr^{-1}) of fishing intensity. The same approach was adopted for examining the effect of trends in fishing effort over a 25-year period. Three groups of rectangles were defined, characterised by their annual rates of change in otter trawl effort: a group where otter trawl effort was declining; a group where effort was slowly increasing, between 0 and 199 hr yr^{-1}; and a group where effort was increasing rapidly, between 200 and 991 hr yr^{-1}. In both cases, differences between the groups were examined by one-way ANOVA (Figure 10.4.1.4.8). In each case, significant variation was detected, but in the direction opposite to that predicted by the hypothesis. Either fishing has caused an increase in species richness, or fishing has increased most in areas where species richness is highest.
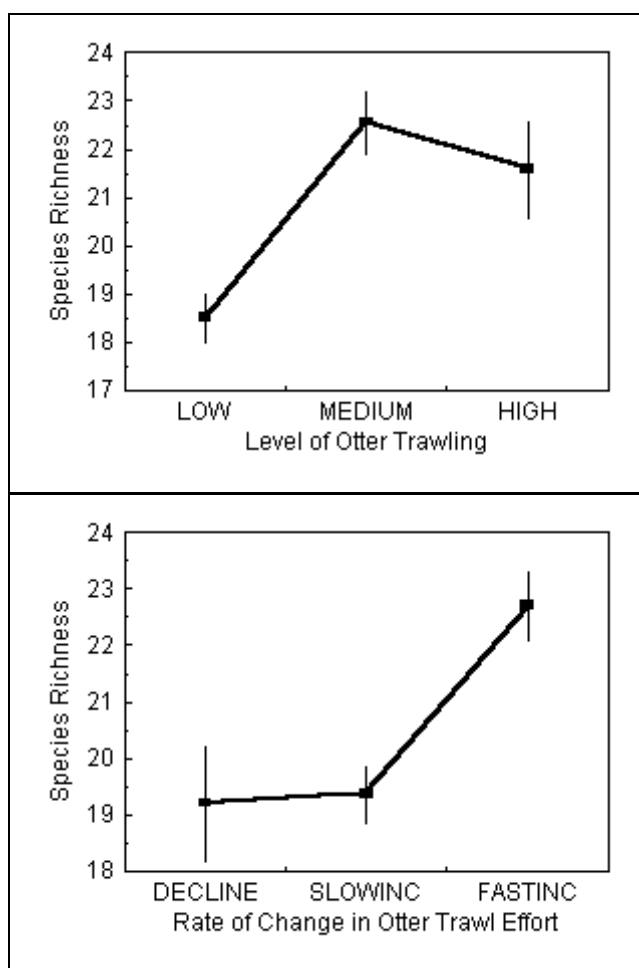

Figure 10.4.1.4.8   Variation in mean species richness (± 1S.E.) calculated for groups of rectangles varying in the level of otter trawling to which they were subjected between 1990 and 1995, and in which the annual rates of change in otter trawling differed over the period 1970 to 1994.

Hypothesis: Species diversity should be lower in areas where current levels of fishing effort are highest, and in areas where recent trends in fishing effort have shown the greatest increase.

The same treatments and analyses applied to examine the relationship between otter trawling and species richness were applied to two species diversity indices calculated from the pooled species abundance data for each of the 75 statistical rectangles. All ANOVAs were significant ($P<0.01$). In this instance, the responses of species diversity to variation in fishing effort lay in the anticipated direction. The difference in species diversity between areas of medium and high otter trawling intensity was marginal. However, both indices appeared to be sensitive to the full range of annual rates of change in fishing activity (see Figure 10.4.1.4.9).
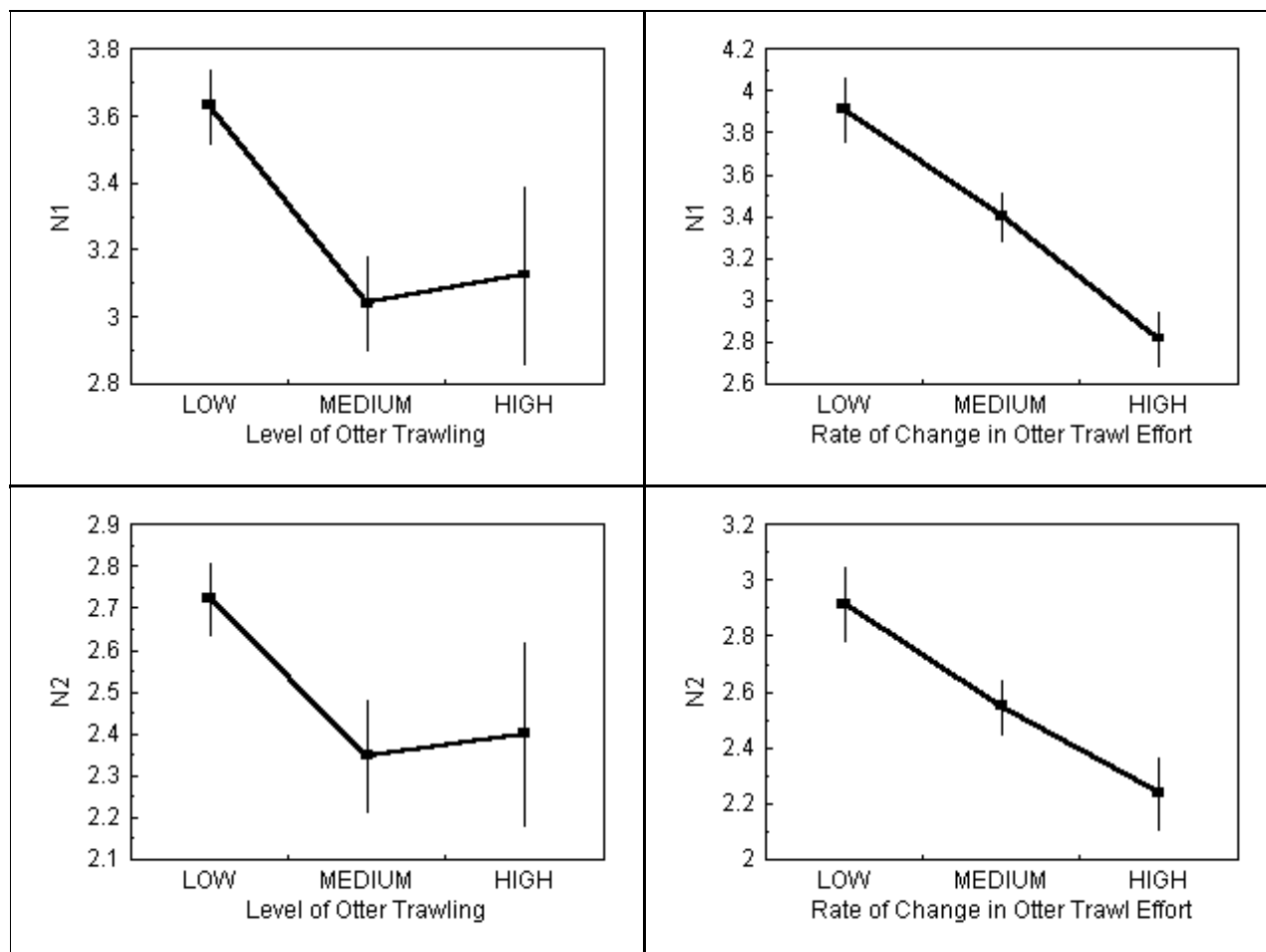

Figure 10.4.1.4.9   Variation in mean species diversity N1 and N2 (± 1 S.E.) calculated for groups of rectangles varying in the level of otter trawling to which they were subjected between 1990 and 1995, and in which the annual rates of change in otter trawling differed over the period 1970 to 1994.

10.4.1.5   Summary of Scottish AGFS results and conclusions

Table 10.4.1.5.1 summarises the results obtained in the analyses carried out on the northwestern North Sea Scottish August Groundfish Survey data. These results clearly demonstrate that mean life history characteristics can detect effects of trawling. Again these results seem to suggest, as emphasised by WGECO in the past, that the application of a suite of metrics provides more information than any single metric alone. This is particularly true for the North Sea where fishing levels are high. Several of the life history characteristics appeared particularly sensitive to the effects of fishing at low fishing intensity. Under some circumstances, these metrics may not detect any further change in impact once otter trawl effort exceeds 5000 hr yr^{-1}.

The results for the two Hill's (1973) diversity indices, N1 and N2, appeared to detect the effect of variation in fishing effort on species relative abundance in the groundfish community. They seemed particularly sensitive to changing levels of fishing disturbance. These data tend to corroborate the previous analyses of this data set. The response of species richness to variation in fishing impact was entirely contrary to the predictions of the hypothesis.

Does fishing cause an increase in species richness? An alternative explanation is that fish abundance has increased most rapidly, and is now at high levels, in areas where species richness is greatest. This highlights the shortcoming of all analyses of this type—they are still not controlled experiments. However, more detailed analysis of this data set, taking account of other sources of variation, such as depth and other environmental factors, may still provide further insight into true cause and effect.

Table 10.4.1.5.1 Summary of results on Scottish AGFS data set (- no analysis, ✔ indicates a significant result, ✗ indicates result was non-significant, footnotes provide further commentary, LTb = slope of the long-term time-series).

Independent variable or treatment	Dependent variable or measure	Analysis	Parameter						
			L_{inf}	k	A_{mat}	L_{mat}	S	N1	N2
Effort level	Parameter	Correlation	✗	✗	✔	✔	-	-	-
Rate of change effort	Parameter	Correlation	✔	✔	✗	✗	-	-	-
Effort level	Mean parameter	ANOVA	✔₁	✔₁	✔	✔₁	✔₃	✔₁	✔₁
Rate of change effort	Mean parameter	ANOVA	✔₂	✔	✔₂	✔₃	✔₃	✔	✔
Effort level	Parameter LTb	ANOVA	✔₁	✔₁	✔	✔₁	-	-	-
Rate of change effort	Parameter LTb	ANOVA	✔₂	✔	✔₃	✔₃	-	-	-

Cell entries are used to summarize patterns as:

1) Most differentiation was between low and medium levels of fishing effort. Data suggest either a threshold or a strongly non-linear effect. If used as a metric it may detect the effect of trawling impact as effort increases from low levels, but may not detect variation in impact as effort varies in relatively heavily fished areas.

2) Could be classed as insensitive—only differentiated between those rectangles where rates of change in effort varied from slow to fast increase. No difference between areas of slow increase and areas of decline in effort.

3) Significant ANOVA, but results difficult to interpret with respect to the hypothesis.

10.4.2 North Sea IBTS data

10.4.2.1 Species characteristics

A table of life history characteristics of fish species caught in the International Bottom Trawl Survey (IBTS) was modified from Daan (2001). Only a few of the life history characteristics could be obtained for the majority of the 266 species; these are maximum length, biogeographical area, habitat, lifestyle, and trophic level. For species with no maximum length (L_{max}) reported for the North Sea, the global value reported for that species was used. Habitat refers primarily to the water depth where the fish is found (e.g., shelf, slope), whereas lifestyle refers to where in the water column the fish is found (e.g., demersal, pelagic). Of these two, habitat was considered most useful for testing the above hypotheses. Trophic level was extracted from FishBase (www.fishbase.org) where it has been calculated from diet information or ECOPATH analyses of the ecosystems in which the given species live.

Additional life history characteristics that were recorded for a subset of the species include maximum age, age and length at maturity, L_{inf} and K from the von Bertalanffy equation, fecundity and egg size. However, these parametric estimates were only available for the subset of the species that are routinely sampled for age.

Ideally, each species could be ranked on an r/K continuum. One measure of the rate of increase (r) is the productivity parameter ($\propto$) from a stock-recruitment relationship. Stock-recruitment relationships can be calculated for commercially important species for which SPAs are available, but not for the entire set of species found in the trawl surveys. Hall and Collie (unpublished) found an inverse relationship between the Ricker $\propto$ and L_{inf}. In this study L_{max} is used as an approximation for L_{inf}. In this manner, L_{max} is a surrogate for the rate of increase; species with low L_{max} are at the r end of the r/K continuum and vice versa, as is expected.

10.4.2.2 Survey data

For the North Sea the hypotheses regarding the effects of fishing on traits of the fish community were tested using the International Bottom Trawl Survey (IBTS) data. The IBTS is a follow-up of the International Young Fish Surveys (IYFS) that were conducted in the North Sea and Skagerrak/Kattegat in February of each year starting in the late 1960s. Over the years, the survey has changed from a survey on young herring into one for demersal fish and herring of all ages and sizes. At the same time, the area surveyed has expanded until from 1974 onwards the whole North Sea proper, Skagerrak, and Kattegat were covered. The IBTS was conducted in international collaboration, with different research

vessels covering specific areas. Over time standardization in gear type, rigging specifications, and sampling strategy was carried out by participating countries (ICES, 1999). During the early years of the survey, a 78-foot Dutch herring trawl was recommended as the standard gear, but in 1977 it was decided to use the GOV-trawl (Grande Ouverture Verticale) as standard gear. From then onward most vessels used GOV, but it took several years before it was adopted by all vessels. The GOV has a high vertical net opening of 5 to 6 m. The horizontal opening of the net is approximately 20 m. Standard fishing speed is 4 knots measured as trawl speed over the ground. Each haul lasts 30 minutes. For the present study, only quarter 1 data from the North Sea proper (excluding the Kattegat and Skagerrak) were used for the years 1974 until present. Each year only those hauls were used where all species caught were recorded.

10.4.2.3 Analysis and results

The hypothesis tested was that "species whose maximum length recorded in the entirety of any particular data set (L_{max}) should decline".

To assess the effect that life history strategy may have on the (changes in) abundance of populations, a life history index was developed based on a species' maximum length by weighting the biomass per species in the annual IBTS catch with the maximum length as expressed in Piet (2001). The average maximum length was shown to decrease significantly ($p<0.01$) from about 41 cm at the start of the sampling period to about 38 cm at the end of that period (Figure 10.4.2.3.1), indicating a relative increase of r-selected species.
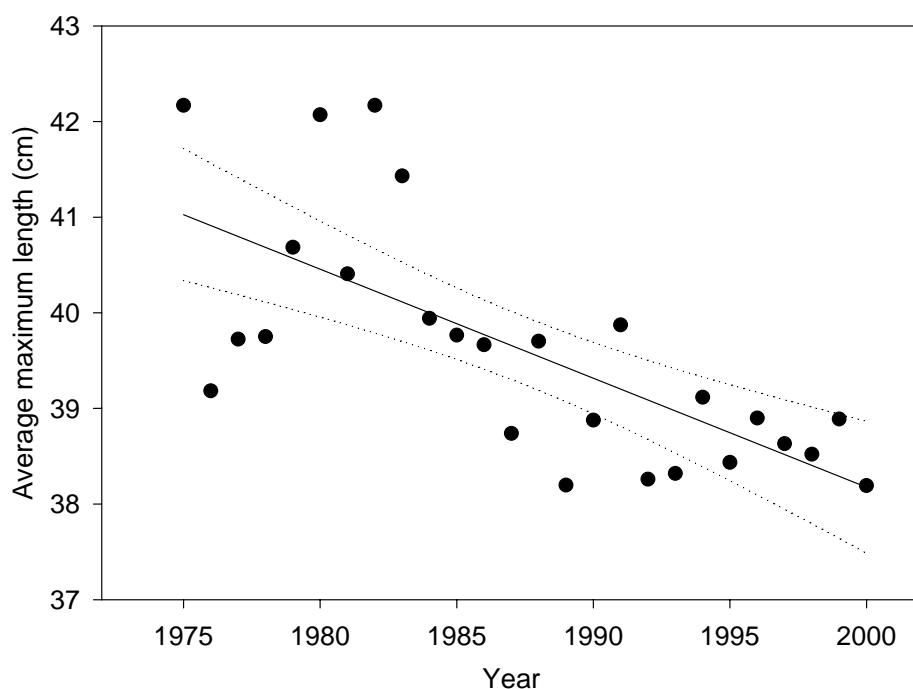

Figure 10.4.2.3.1.   Average maximum length of the fish community over time. Points are values per year, lines show fit and 95% confidence interval.

The effect of fishing effort on the maximum length index was studied by determining the mean maximum length and slope of the change in the maximum length index over time per ICES rectangle and combining these with the effort data per ICES rectangle according to Jennings *et al.* (1999a, 1999b, 2000).

The mean maximum length showed a significant ($p<0.01$) decrease with fishing effort. Combination of ICES rectangles into three effort-classes (Low≤10 000, 10 000<Medium≤30 000, High>30 000 hr yr^{-1}) showed a significantly higher mean maximum length for ICES rectangles where fishing effort was "Low" (Figure 10.4.2.3.2).
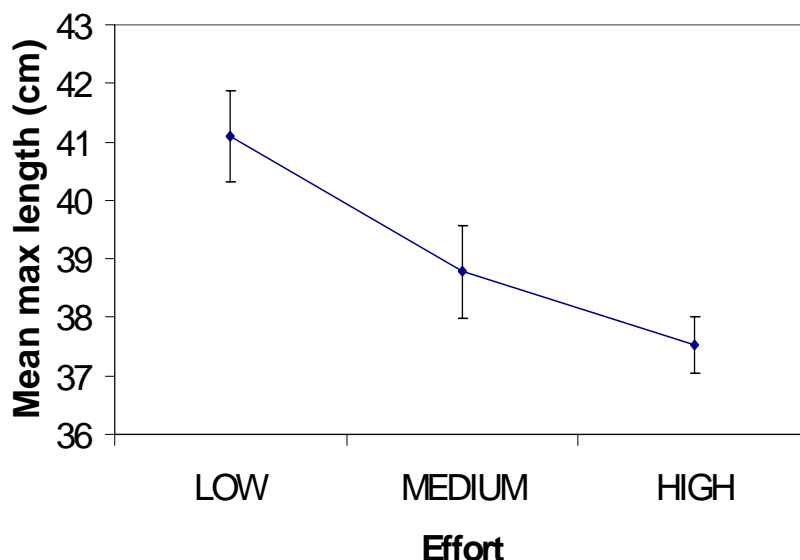
Figure 10.4.2.3.2 Mean Length$_{max}$ and 95% confidence limits for three classes of fishing effort (Low≤10 000, 10 000<Medium≤30 000, High>30 000 hr yr^{-1}). Based on 161 ICES rectangles for which IBTS and effort data were available.

The slope of the change in maximum length over time in relation to fishing effort is shown in Figure 10.4.2.3.3. Slope did not decrease with increasing effort as might be expected. The reason is that a significant inverse relationship between mean maximum length and slope was observed. This suggests that in the heavily fished ICES rectangles the composition of the fish community in terms of life history traits has stabilized at a relatively high level of r-strategists represented by a low mean maximum length.
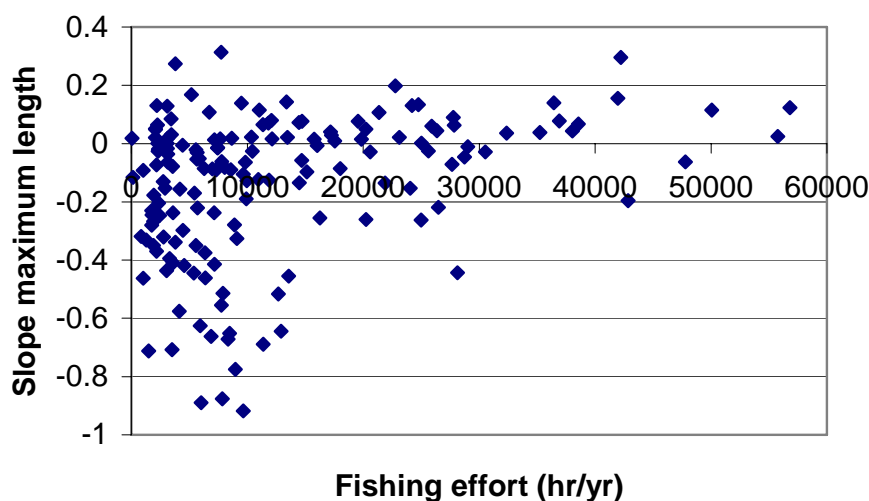
Figure 10.4.2.3.3 Relationship between the slope of the mean maximum length over time and fishing effort in 161 ICES rectangles.

10.4.3 Portuguese survey data

10.4.3.1 Species characteristics

Life history characteristics for 194 species caught in the Portuguese demersal survey were compiled by the Working Group. Of this group of species, 105 were common to the North Sea. An effort was made to standardize the sets of life history categories between regions such that these categories would be applicable to all regions of the North Atlantic. As for the North Sea, the variables that were available for most of the species were L_{max}, lifestyle, habitat and trophic level. These life history characteristics are global values for the species (extracted from FishBase, Whitehead *et al.,* 1984) and were not collected as part of the trawl survey. The 38 species for which one or more species characteristic were missing were excluded from the data set. These were all very rare species, collectively representing only 0.057% of the total number of all individuals in the data set.

10.4.3.2 Survey data

Demersal survey cruises have been carried out annually in continental Portuguese waters since 1979 and are well described in Cardador *et al.* (1997) (Figure 10.4.3.2.1). Data from the autumn (fourth quarter) surveys for 1982 and from 1989 to 2000 are used here. All stations are separated into three geographic zones (North, Centre and South) at 39.5 ºN and 37.1 ºN latitude, and into two depth strata (less than and more than 150 m). This gave six groups of data that were used for all subsequent analyses. The criteria for these choices were largely taken from Gomes *et al.* (2001). For each year and for each of these groups, the total number of individuals of each species (after being scaled up to number of individuals for 1 hour for each haul, when necessary) was determined.
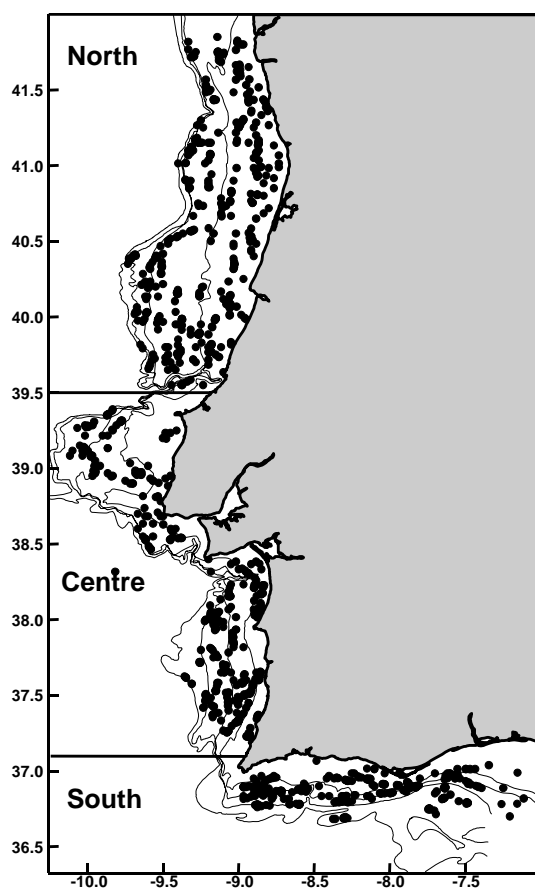

Figure 10.4.3.2.1   Map of continental Portuguese waters, showing survey stations (●), the three geographical zones (North, Centre and South) and with 100, 200, 500 and 750 m contour lines.

10.4.3.3 L_{max} and trophic level analysis and results

In order to test the hypotheses about L_{max} and trophic level presented above, the trends in these characteristics over time were analysed. This was done by calculating the weighted average value for each year based on the biomass of each species and the individual species characteristic value. These values were then plotted against time and trend lines

determined (Table 10.4.3.3.1, Figure 10.4.3.3.1). From these data the following trends were observed, as shown in Figure 10.4.3.3.1.

Table 10.4.3.3.1   Trends ($\Downarrow$ = decrease, $\Uparrow$ = increase) in weighted averages of L_{max} and trophic level. Values are R^2, a decrease with an R^2 over 0.25 is significant).

	L_{max}	Trophic level
North shallow	$\Downarrow$ 0.335	$\Downarrow$ 0.402
North deep	$\Downarrow$ 0.022	$\Downarrow$ 0.096
Centre shallow	$\Downarrow$ 0.203	$\Downarrow$ 0.130
Centre deep	$\Downarrow$ 0.006	$\Uparrow$ 0.036
South shallow	$\Uparrow$ 0.137	$\Uparrow$ 0.090
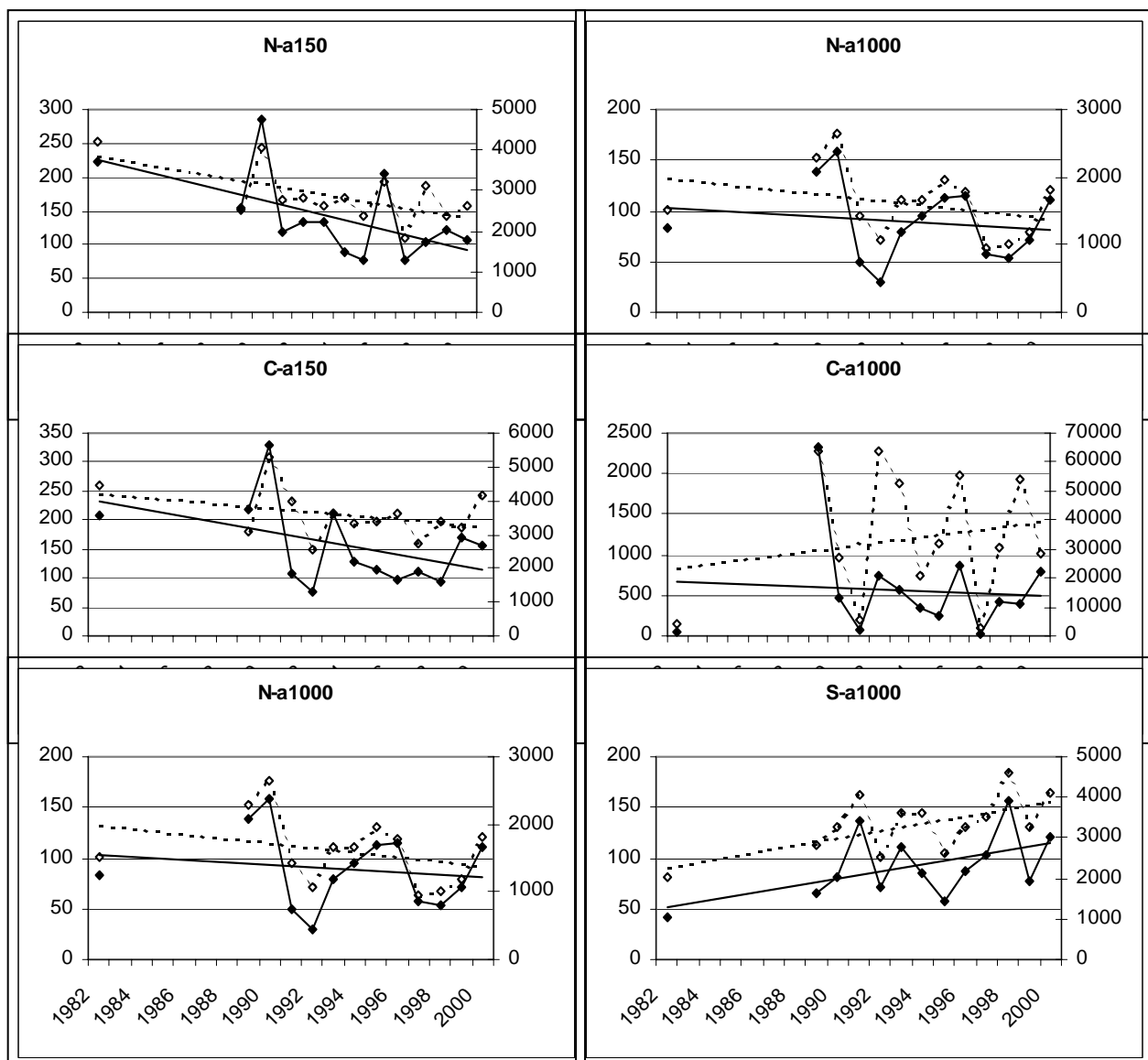
South deep	$\Uparrow$ 0.278	$\Uparrow$ 0.407

Figure 10.4.3.3.1   Plots of weighted averages of L_{max} and trophic level for 6 groups (N=North, C=Centre, S=South; a150=1–150 m, a1000=151 to 1000 m). Solid lines for L_{max}, dashed lines for trophic level.

It can therefore be seen that, except for in the north shallow group, these data do not show a significant decrease. There are a number of potential explanations for this, which could include:

- A decrease in exploitation since 1982;
- A change in faunal composition due to immigration of new species;
- The fishery targets small species;
- The shortness of the time-series.

It is not surprising that there are differences between the north groups and the others as there are strong physical differences between these zones. The north has a relatively large and flat continental shelf. The northern and central zones are divided by the Nazaré canyon, and below this point the shelf is considerably more narrow.

These analyses strongly suggest that these metrics are not reliable indicators on their own, at least not in this system. The life history traits are changing, but the patterns of change usually are not as predicted by theory, and the reasons for the changes are not understood. For example, in the centre deep group L_{max} decreased while trophic level increased. Both data sets contain a lot of noise, but trophic level more so, suggesting that trophic level is a less sensitive metric. Whether the differences between the results of these analyses and those using the Scottish data (Section 4.4.1), where a number of life history parameters were thought to have changed in ways consistent with theory, are due to differences between the ecosystems, between the fisheries, or just due to the differences in contrast within the data sets, remains to be explored.

10.4.3.4   Spatial metrics analysis

10.4.3.4.1   Description of data

The data were selected from the Portuguese survey data. Only data for elasmobranchs (27 species) and gadids (13 species from the Gadidae family) were used in order to keep the set small, but providing some contrast. Because of their reproductive strategies, elasmobranchs *a priori* may be expected to be more susceptible to effort. The distribution of the survey is shown in Figure 10.4.3.4.1. The subset had year, longitude-latitude, depth, species name, numbers caught, biomass caught, L_{max} and trophic level. The time-series was for 1982 and 1989–2000. Preparatory to the analysis the data were aggregated to a tenth of a degree squares and a code number given to each species to aid analysis. It would have been beneficial to apply the following analyses to other data sets, but time constraints prohibited this. In the following, we use the term community to denote either the elasmobranch or the gadoid group.
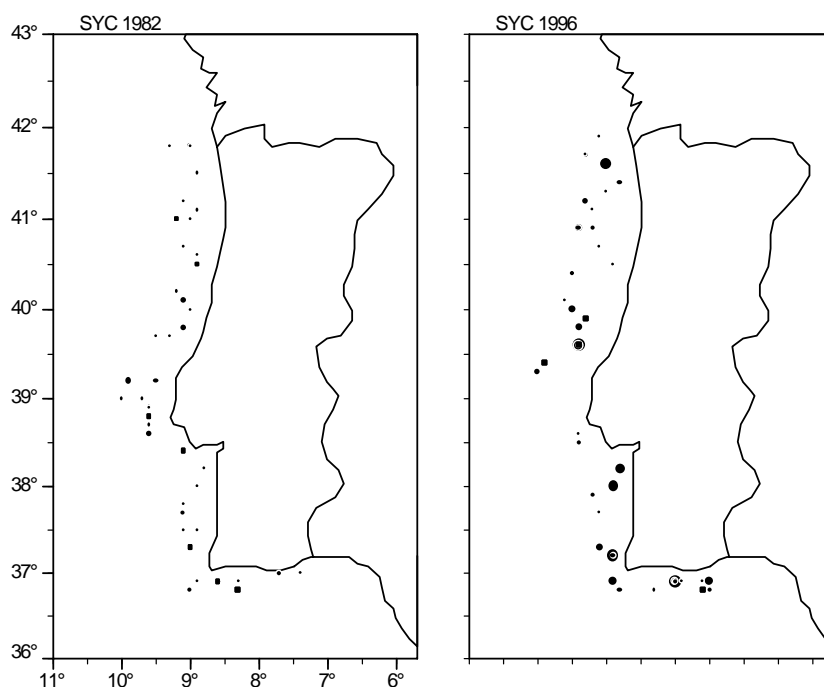

Figure 10.4.3.4.1   Sample data from the Portuguese survey series. The figure contains the aggregated biomass for *Scyliorhinus canicula* 1982 and 1996. Larger circles reflect higher abundance.

Unfortunately, the data available (a subset of the Portuguese trawl survey) did not have sufficient quality (duration and knowledge of, or dynamic range of, effort) to test the effect of fishing effort on the metrics. Instead of a hypothesis-based study, a preliminary investigation of the performance of three spatial metrics on a single data set is reported.

10.4.3.4.2 Description of metrics

Because of the unavailability of effort data, the metrics were compared to abundance trends over the time-series. Both the unweighted biomass per tow and L_{max} weighted biomass were considered.

Spatial metric 1 – Anomaly of the center of mass of the community.

Based on experience from Eastern Scotian Shelf cod, a metric was proposed which is the anomaly of the centre of mass of distribution. In that case it was noted that the centre of mass seemed to predict a subsequent rapid decrease in biomass in the late 1980s. The metric is calculated by first computing the centre of mass for each year and summing over the species or community under consideration. The average over time-series of centroids is found and then the distance (in nautical miles) from each annual point to the average is found giving an annual anomaly.

Spatial metric 2 – Index of contagion

A contagion index is proposed which is the number of neighbours within a set radius. Figure 10.4.3.4.2.1 shows three different distributions of four animals or sets of animals and a test radius of 2 units. In the upper plot, the four animals are so far apart that they have no neighbours within the test radius. In the middle plot, the four animals are in a row and there are three pairs of neighbours within the radius. Finally in the bottom plot, all four animals are in a cluster and the index is now 6. Because it is hypothesised above that contagion is probably more important on the species level, this metric is computed for each species of concern and then summed for all species under consideration. This sum could either be unweighted or weighted by abundance.
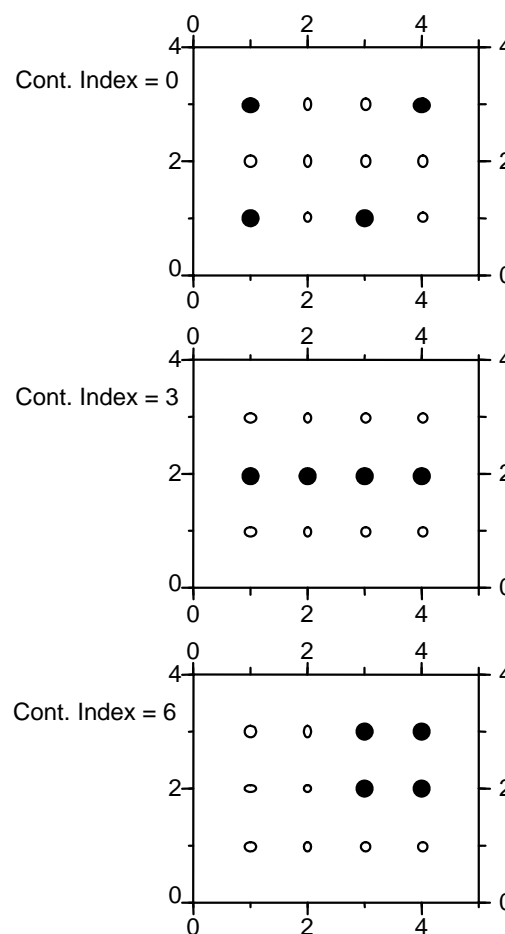

Figure 10.4.3.4.2.1 Samples of contagion index.

Spatial metric 3 – Index of overlap

This index is proposed to indicate the displacement of a resource from its traditional, pristine or desired distribution. A reference year (or distribution) is chosen and then it is compared to the other years in a time-series. As the data are aggregated onto a 0.1 degree grid it requires only to check if the same square is occupied as in the reference distribution. The index is the fraction of occupied grids in the reference distribution that are shared. Figure 10.4.3.4.2.2 shows this index for a simple data set.
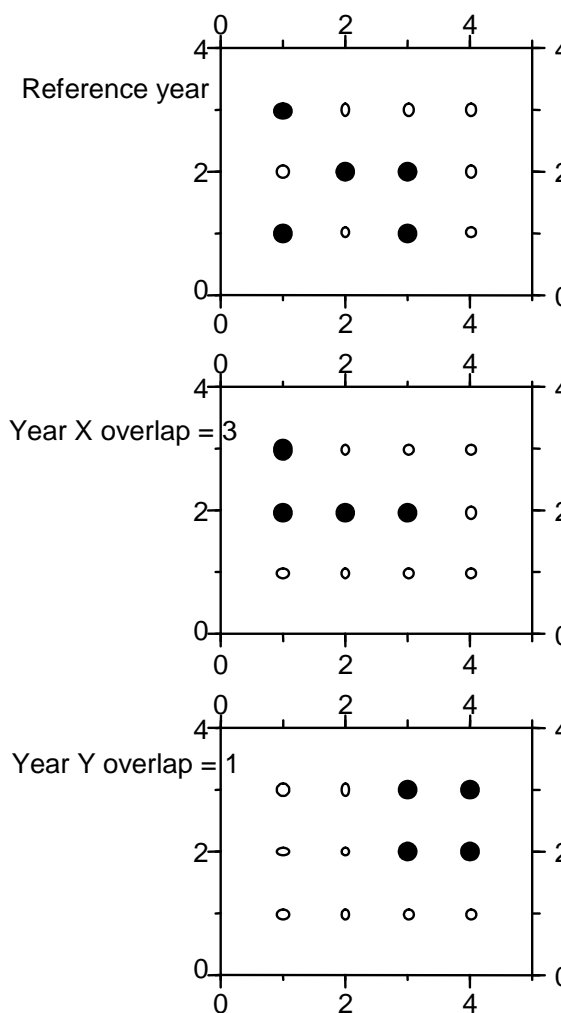
Figure 10.4.3.4.2.2 Samples of an overlay index.

10.4.3.4.3   Analysis and results

Figure 10.4.3.4.3.1 shows the abundance for the gadid group from the survey data as well as the abundance-weighted L_{max}. Abundance fell after 1982 while the weighted L_{max} showed little dynamics except for a small blip in 1992. The following figure (Figure 10.4.3.4.3.2) shows the abundance trends for animals with L_{max} above and below 100 cm. This shows that the larger animals were much more impacted over the data period but that the weighted L_{max} failed to pick up this event, probably because the larger group was such a small proportion of the total. (For plotting and to ease comparison, all indices in the following have been normalised to their mean.)
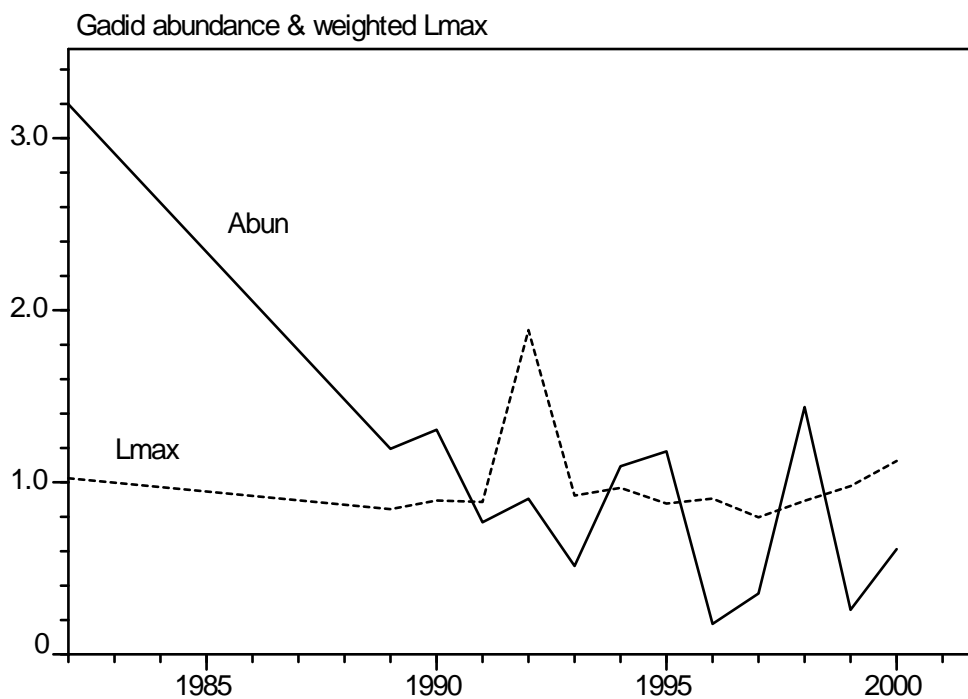
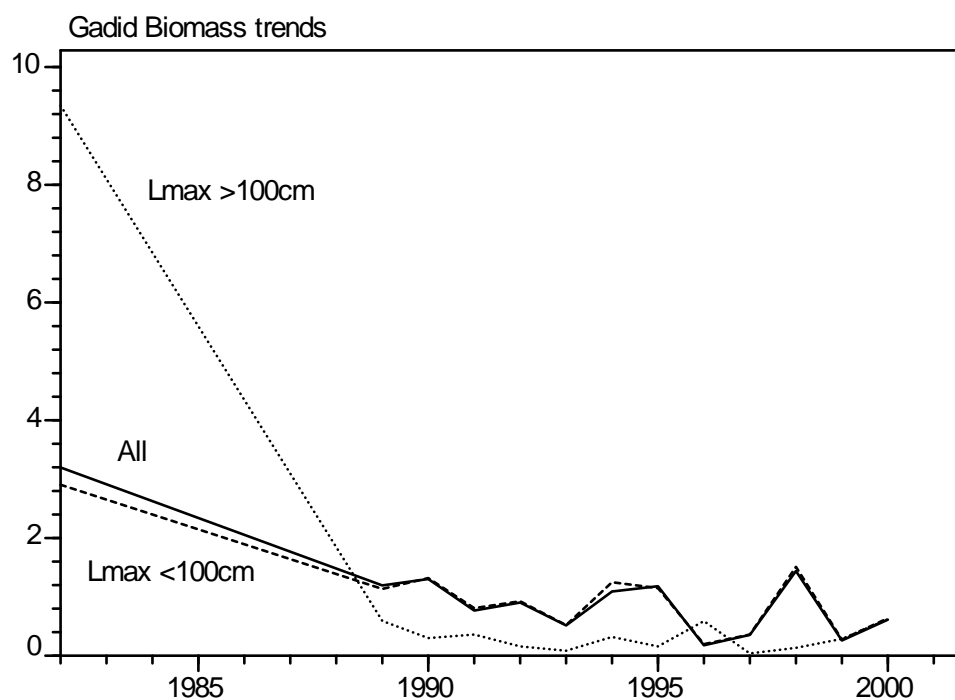**Figure 10.4.3.4.3.1** Gadid abundance L_{max} weighted by abundance.

Figure 10.4.3.4.3.2 Gadid abundance after separating animals to those with L_{max} over and under 100 cm.

An analogous pair of plots are given for elasmobranchs (Figures 10.4.3.4.3.3 and 10.4.3.4.3.4). For this group of fish, there was a rise between 1982 and the next observation in 1989 in abundance which affected the weighted L_{max}. There was a spike in recruitment also in 1996 which did not affect the group's L_{max}, presumably because it was caused by animals near the mean L_{max}. The separation into size groups shows that the first event was due to large elasmobranches, while the second was dominated by smaller fish.
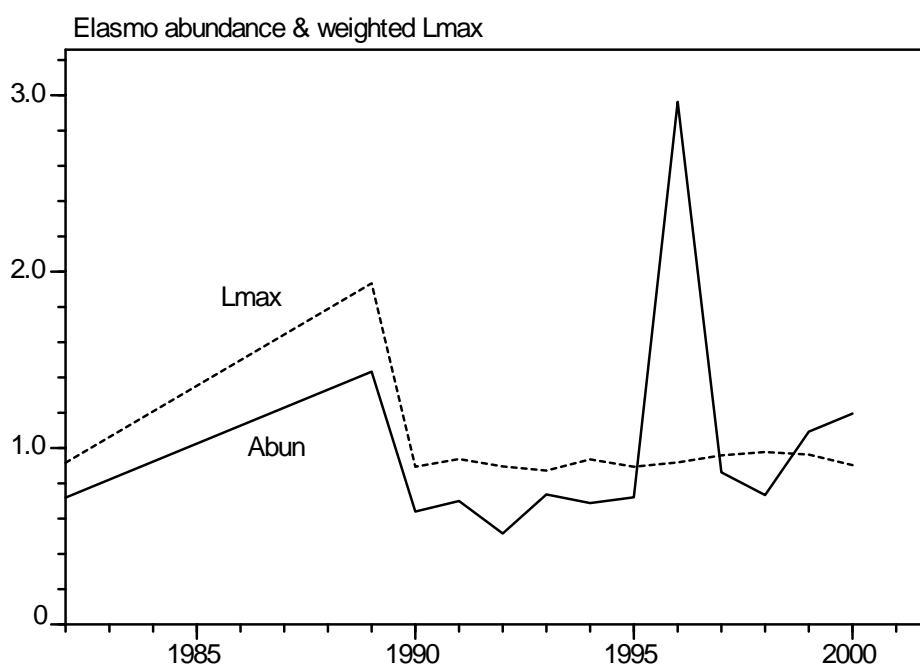
Figure 10.4.3.4.3.3 Elasmobranch abundance with and without L_{max} weighting.
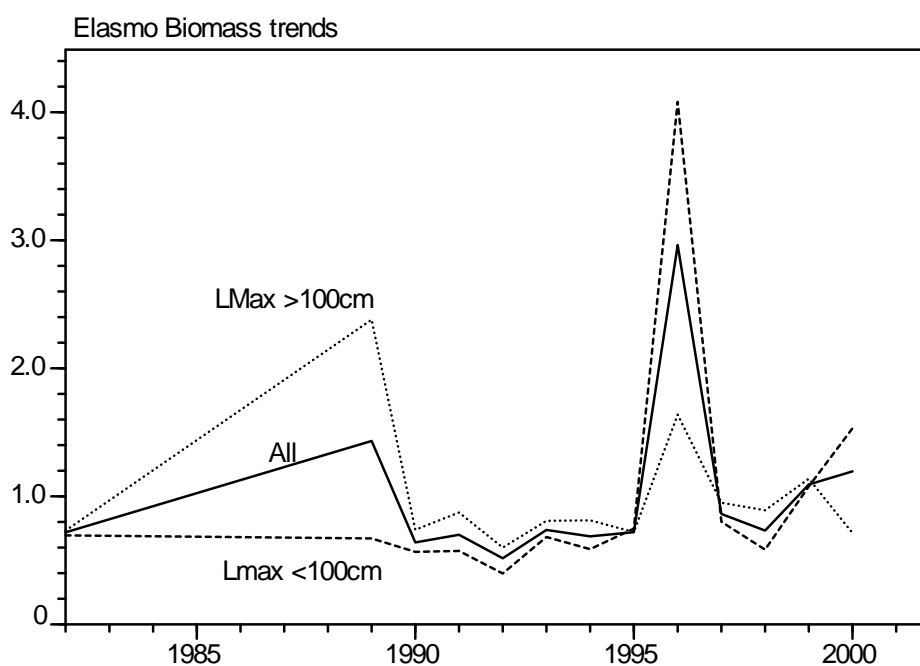

Figure 10.4.3.4.3.4 Gadid abundance after separating animals to those with L_{max} over and under 100 cm.

Figure 10.4.3.4.3.5 shows the abundance and the anomaly of the centre of biomass for the gadid group of fishes. It is difficult to infer the performance of the anomaly with this short data series although it appears to be somewhat opposite in phase to the abundance after 1989, suggested by the data from 1993, 1997, and 1998. Figure 10.4.3.4.3.6 shows a more dynamic anomaly. In 1992 there is little change in the biomass but the centre of mass moves dramatically, whereas in 1996 abundance shows a large change while the distribution metric changes very little. Further investigation is required to see if these reflect "real" events.
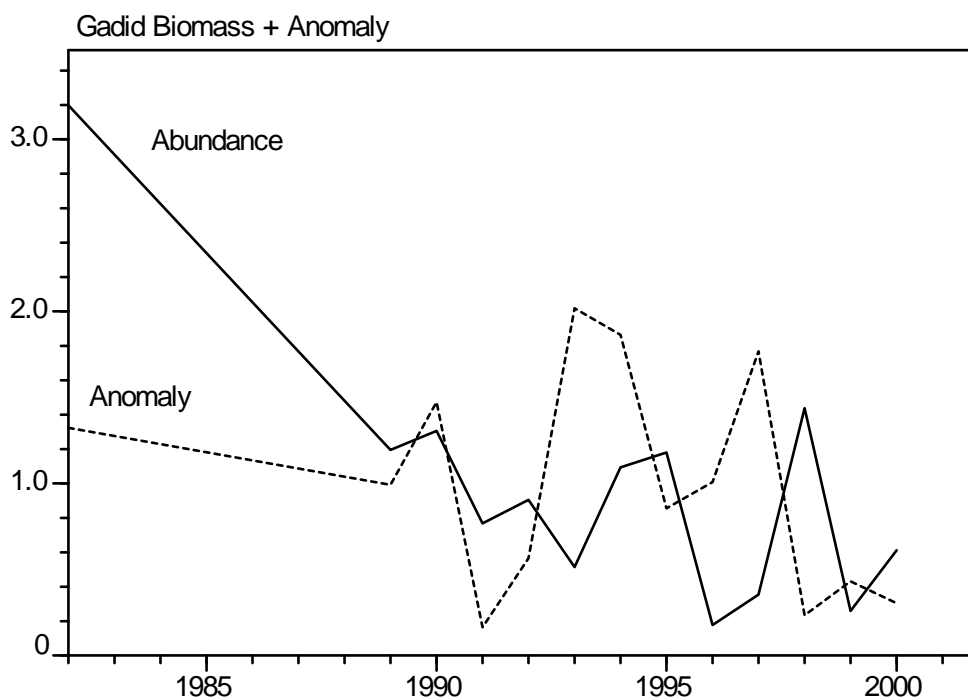
Figure 10.4.3.4.3.5   Gadid abundance and anomaly of the centre of mass.
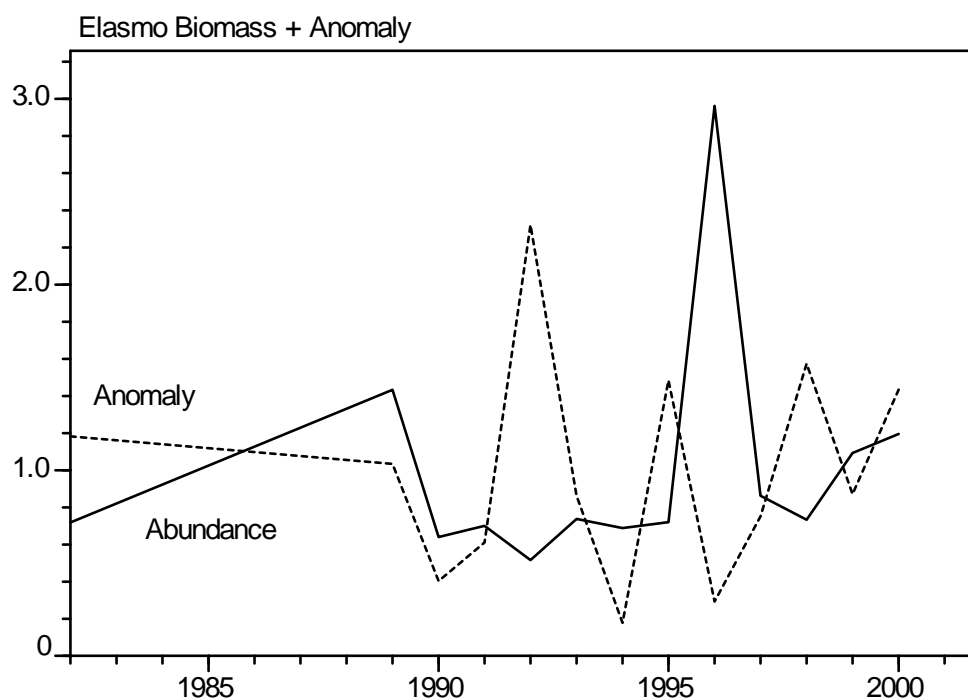

Figure 10.4.3.4.3.6   Elasmobranch abundance and anomaly of the centre of mass.

Figure 10.4.3.4.3.7 shows the abundance, contagion index, and overlap index for the gadid blue whiting (*Micromesistius poutassou*). The two spatial indices are highly correlated to the abundance. Figure 10.4.3.4.3.8 shows the abundance, contagion index, and overlap index for the elasmobranch *Scyliorhinus canicula*. These species were chosen because they were commonly seen in the survey. While the abundance has a spike in 1996, the spatial indices do not respond to the change, suggesting that the biomass distribution was not affected. Unlike the other two trends, the overlap fell after the reference year (1982), suggesting a displacement of the resource from that period.
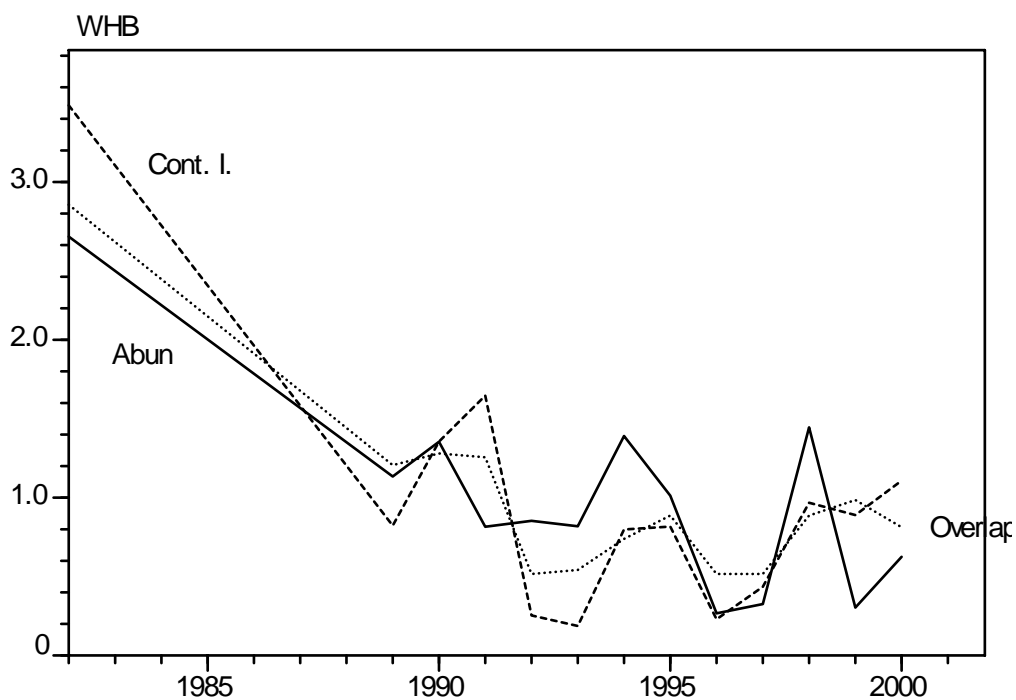
Figure 10.4.3.4.3.7 Abundance, contagion index, and overlap index for blue whiting.
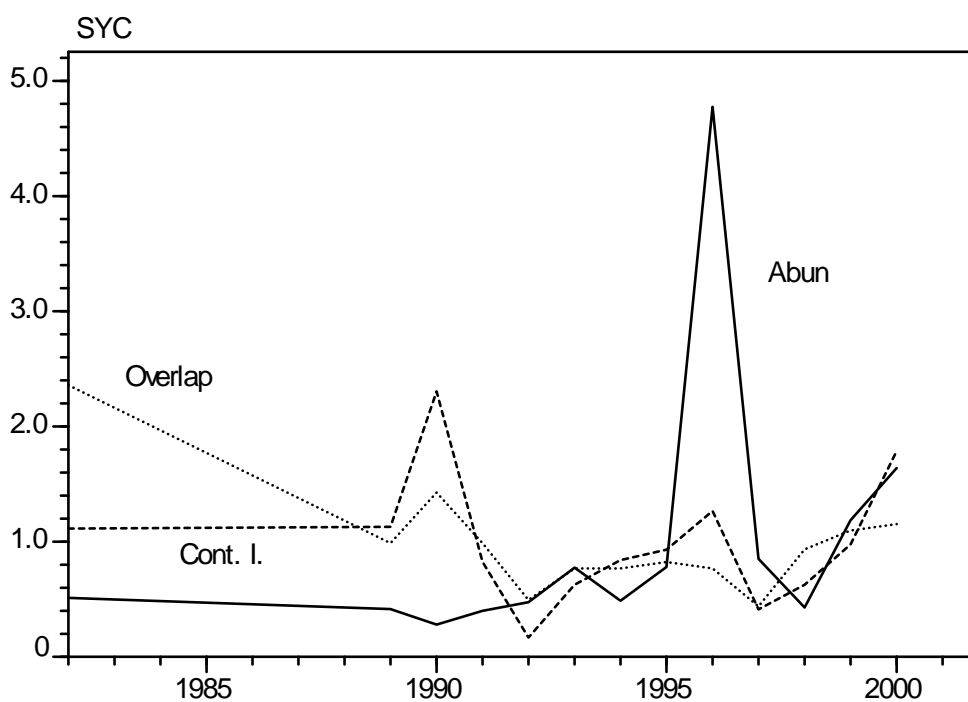

Figure 10.4.3.4.3.8 Abundance, contagion index, and overlap index for the elasmobranch species *Scyliorhinus canicula*.

The gadid group trends for abundance, contagion, and overlap are shown in Figure 10.4.3.4.3.9 and the three trends are quite similar. The indices for the elasmobranch group show more divergence (Figure 10.4.3.4.3.10). The overlap index fell even more than was seen in the single elasombranch species shown. The contagion increases with the increase in abundance in 1996, suggesting that for a number of elasmobranch species, the increase was localised.
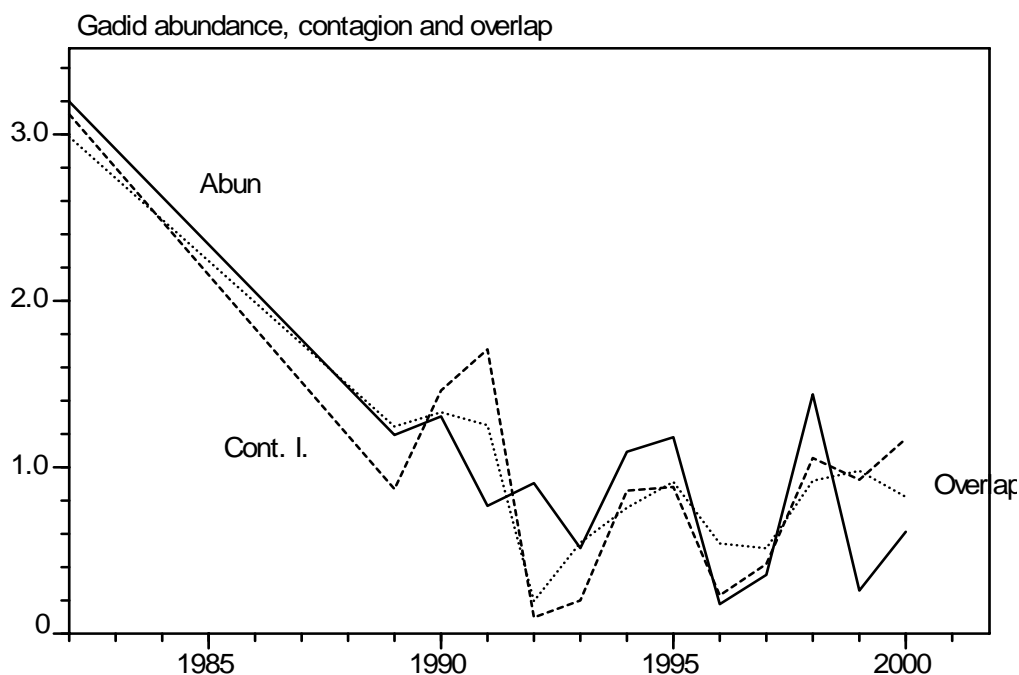
Gadid abundance, contagion and overlap

Figure 10.4.3.4.3.9 Abundance, contagion index, and overlap index for the gadid group of species.
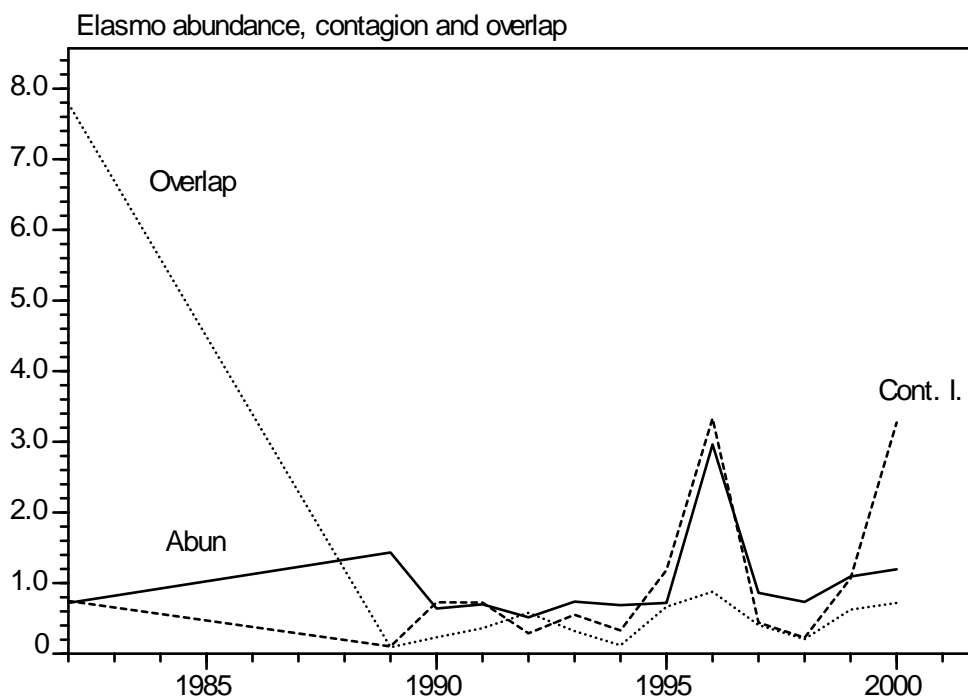
Elasmo abundance, contagion and overlap

Figure 10.4.3.4.3.10 Abundance, contagion index, and overlap index for the elasmobranch group of species.

Thus, these results should be considered only as illustrations of the metrics. Even if all the analytical tools are working properly, the data are not sufficient to select among the proposed indices; none failed conspicuously, nor did any excel. Further research should be conducted in three areas: expansion to other sets of data; refinement of the metrics; and the development of a more methodical screening procedure.

10.5　　　Concluding thoughts and way forward

Work remains to carry many of these analyses further. Although every effort was made to assure error-free analyses, the Working Group set-up did not allow thorough review of all analyses by all WGECO members. Furthermore, not all the hypotheses proposed could be tested, without reformatting or revisions to the data that were not possible for the Working Group. Hence, conclusions in these sections are preliminary, although it is important to highlight that many results are consistent with the predictions made about changes in life history characteristics. Nevertheless, at this stage the analyses were not detailed enough to be used by themselves to justify strong conclusions about the sensitivity and information content of life history traits relative to fishing effort. In particular, a better understanding is needed of how

cases where predictions from life history theory were supported differed from the cases where either the predicted patterns were not found or where significant patterns actually were the opposite of the predictions from theory. WGECO feels that such analyses are important, because, along with impacts on physical habitat features, the effects of fishing on life history properties of species are some of the most lasting effects of fishing (ICES, 2000), and the work should be continued within ICES.

References

Branton, R., and Black, J. 2000. 2000 summer groundfish survey update for selected Scotia-Fundy groundfish stocks. DFO Canadian Stock Assess. Res. Doc. 2000/129. 61 pp.

Cardador, F., Sanchéz, F., Pereiro, F.J., Borges, M.F., Caramelo, A.M., Azevedo, M., Silva, A., Pérez, N., Martins, M.M., Olaso, I., Pestana, G., Trujillo, V., and Fernandez, A. 1997. Groundfish surveys in the Atlantic Iberian waters (ICES Divisions VIIIC and IXa): history and perspectives. ICES CM 1997/Y: 08.

Dann, N. 2001. A spatial and temporal diversity index taking into account species rarity, with an application to the North Sea fish community. ICES CM 2001/T:04.

Gomes, M.C., Serrão, E., and Borges. M.F. 2001. Spatial patterns of groundfish assemblages on the continental shelf of Portugal. *ICES Journal of Marine Science*, 58(3):633–647.

Greenstreet, S.P.R., and Hall, S.J. 1996. Fishing and the ground-fish assemblage structure in the north-western North Sea: an analysis of long-term and spatial trends. Journal of Animal Ecology, 65: 577–598.

Greenstreet, S.P.R., and Rogers, S.I. 1999. The effects of fishing on non-target fish species. *In* Effects of fishing on non-target species and habitats: biological conservation and socio-economic issues. Ed. by M.J. Kaiser and S.J. de Groot. Blackwell Science, Oxford. 416 pp.

Greenstreet, S.P.R., Spence, F.E., and McMillan, J.A. 1999a. Fishing effects in northeast Atlantic shelf seas: patterns in fishing effort, diversity and community structure. V. Changes in structure of the North Sea groundfish assemblage between 1925 and 1996. Fisheries Research, 40: 153–183.

Greenstreet, S.P.R., Spence, F.E., Shanks, A.M., and McMillan, J.A. 1999b. 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 U.K. registered vessels landing in Scotland. Fisheries Research, 40: 107–124.

Hill, M.O. 1973. Diversity and evenness: a unifying notation and its consequences. Ecology, 54: 427-432.

Huston, A.H. 1994. Biological Diversity: The Coexistence of Species on Changing Landscapes. Cambridge University Press. 681 pp.

ICES. 1999. Report of the International Bottom Trawl Survey Working Group. ICES CM 1999/D:2.

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

Jennings, S., Reynolds, J.D., and Mills, S.C. 1998. Life history correlates of responses to fisheries exploitation. Proceedings of the Royal Society of London, 265: 1–7.

Jennings, S., Greenstreet, S.P.R., and Reynolds, J. 1999a. Structural change in an exploited fish community: a consequence of differential fishing effects on species with contrasting life histories. Journal of Animal Ecology, 68: 617–627.

Jennings, S., Alvsvag, J., Cotter, A.J., Ehrich, S., Greenstreet, S.P.R., Jarre-Teichmann, A., Mergardt, N., Rijnsdorp, A.D., and Smedstad, O. 1999b. Fishing effects in northeast Atlantic shelf seas: patterns in fishing effort, diversity and community structure. III. International fishing 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., and Cotter, A.J.R. 2000. Spatial and temporal patterns in North Sea fishing effort. *In* Effects of Fishing on Non-Target Species and Habitats: Biological, Conservation and Socio-economic Issues. Ed. by M.J. Kaiser and B. de Groot. Blackwell Science, Oxford.

Lindeboom, H.J., and de Groot, S.J. (Eds.) 1998. IMPACT II. The effects of different types of fisheries on the North Sea and Irish Sea benthic ecosystem. NIOZ-Rapport 1998-1, RIVO-DLO Report C003/98. 404 pp.

Paloheimo, J.E., and Dickie, L.M. 1964. Abundance and fishing success. Rapp. P.-V. Reun. Cons. Int. Explor. Mer. 155: 152–163.

Pauly, D., Palomares, M.L., Froese, R., Sa-a, P., Vakily, M., Preikshot, D., and Wallace S. 2001. Fishing down Canadian aquatic food webs. Can. J. Fish. Aquat. Sci. 58: 51–62.

Piet, G.J. 2001. Development of ecological quality objectives for the North Sea fish community. ICES 2001 working document.

Whitehead, P.J.P., Bauchot, M.-L., Hureau, J.-C., Nielsen, J., and Tortonese, E. (Eds.). 1984. Fishes of the North-eastern Atlantic and the Mediterranean. UNESCO, Paris. Vols I, II and III. 1473 pp.

Titles Published in the *ICES Cooperative Research Report* Series

No.	Title	Price (Danish Kroner)
273	Guidance on the application of the Ecosystem Approach to Management of human activities in the European marine environment	*In press*
272	Ecosystem Effects of Fishing: Impacts, Metrics and Management Strategies	*In press*
271	Vector Pathways and the Spread of Exotic Species in the Sea. 25 pp.	60
270	The *Nephrops* fisheries of the Northeast Atlantic and Mediterranean – A review and assessment of fishing gear design. 38 pp.	50
269	The Annual ICES Ocean Climate Status Summary 2003/2004. 32 pp.	60
268	The DEPM Estimation of Spawning-stock Biomass for Sardine and Anchovy. 87 pp.	90
267	Report of the Thirteenth ICES Dialogue Meeting: Advancing scientific advice for an ecosystem approach to management: collaboration amongst managers, scientists, and other stakeholders. 59 pp.	50
266	Mesh Size Measurement Revisited. 56 pp.	80
265	Trends in important diseases affecting the culture of fish and molluscs in the ICES area 1998–2002. 26 pp.	40
264	Alien Species Alert: *Rapana venosa* (veined whelk). 14 pp.	50
263	Report of the ICES Advisory Committee on the Marine Environment, 2003. 227 pp.	150
262	Report of the ICES Advisory Committee on Ecosystems, 2003. 229 pp.	170
261	Report of the ICES Advisory Committee on Fishery Management, 2004 (Parts 1–3). 975 pp.	430
260	Stockholm 1999 Centenary Lectures. 48 pp.	170
259	The 2002/2003 ICES Annual Ocean Climate Status Summary. 29 pp.	150
258	Seabirds as Monitors of the Marine Environment. 73 pp.	200
257	Proceedings of the Baltic Marine Science Conference. 334 pp.	420
256	Report of the ICES Advisory Committee on the Marine Environment, 2002. 155 pp.	270
255	Report of the ICES Advisory Committee on Fishery Management, 2002. (Parts 1–3). 948 pp.	1200
254	Report of the ICES Advisory Committee on Ecosystems, 2002. 129 pp.	250
253	ICES Science 1979–1999: The View from a Younger Generation. 39 pp.	170
252	Report of the ICES/GLOBEC Workshop on the Dynamics of Growth in Cod. 97 pp. (including CD-Rom from ICES ASC 2001)	220
251	The Annual ICES Ocean Climate Status Summary 2001/2002. 27 pp.	180
250	ICES/GLOBEC Sea-going Workshop for Intercalibration of Plankton Samplers. A compilation of data, metadata and visual material, 2002. 34 pp.	190
249	Report of the ICES Advisory Committee on Ecosystems, 2001. 75 pp.	200
248	Report of the ICES Advisory Committee on the Marine Environment, 2001. 203 pp.	310
247	Effects of Extraction of Marine Sediments on the Marine Ecosystem. 80 pp.	200
246	Report of the ICES Advisory Committee on Fishery Management, 2001. (Parts 1–3). 895 pp.	1170
245	The Annual ICES Ocean Climate Status Summary 2000/2001. 21 pp.	150
244	Workshop on Gadoid Stocks in the North Sea during the 1960s and 1970s. The Fourth ICES/GLOBEC Backward-Facing Workshop. 55 pp.	160
243	Report of the 12th ICES Dialogue Meeting (First Environmental Dialogue Meeting). 28 pp.	130
242	Report of the ICES Advisory Committee on Fishery Management, 2000. (Parts 1–3). 940 pp.	1100

No.	Title	Price (Danish Kroner)
241	Report of the ICES Advisory Committee on the Marine Environment, 2000. 263 pp.	370
240	Report of the Young Scientists Conference on Marine Ecosystem Perspectives. 73 pp.	170
239	Report of the ICES Advisory Committee on the Marine Environment, 1999. 277 pp.	350
238	Report on Echo Trace Classification. 107 pp.	200
237	Seventh Intercomparison Exercise on Trace Metals in Sea Water. 95 pp.	190
236	Report of the ICES Advisory Committee on Fishery Management, 1999. (Part 1 and Part 2). 821 pp.	920
235	Methodology for Target Strength Measurements (With special reference to *in situ* techniques for fish and mikronekton). 59 pp.	160
234	Report of the Workshop on Ocean Climate of the NW Atlantic during the 1960s and 1970s and Consequences for Gadoid Populations. 81 pp.	180
233	Report of the ICES Advisory Committee on the Marine Environment, 1998. 375 pp.	440
232	Diets of Seabirds and Consequences of Changes in Food Supply. 66 pp.	170
231	Status of Introductions of Non-Indigenous Marine Species to North Atlantic Waters 1981–1991. 91 pp.	190
230	Working Group on Methods of Fish Stock Assessment. Reports of Meetings in 1993 and 1995. 259 pp.	330
229	Report of the ICES Advisory Committee on Fishery Management, 1998. (Part 1 and Part 2). 776 pp.	900
228	Report of the 11th ICES Dialogue Meeting on the Relationship Between Scientific Advice and Fisheries Management. 37 pp.	140
227	Tenth ICES Dialogue Meeting (Fisheries and Environment in the Bay of Biscay and Iberian Region: Can the Living Resources Be Better Utilized). 30 pp.	130
226	Report on the Results of the ICES/IOC/OSPARCOM Intercomparison Programme on the Determination of Chlorobiphenyl Congeners in Marine Media – Steps 3a, 3b, 4 and Assessment. 159 pp.	250
225	North Atlantic – Norwegian Sea Exchanges: The ICES NANSEN Project. 246 pp.	320
224	Ballast Water: Ecological and Fisheries Implications. 146 pp.	230
223	Report of the ICES Advisory Committee on Fishery Management, 1997. (Part 1 and Part 2). 780 pp.	760
222	Report of the ICES Advisory Committee on the Marine Environment, 1997. 210 pp.	250
221	Report of the ICES Advisory Committee on Fishery Management, 1996. (Part 1 and Part 2). 642 pp.	660
220	Guide to the Identification of North Sea Fish Using Premaxillae and Vertebrae. 231 pp (including 300 photographs)	560
219	Database Report of the Stomach Sampling Project, 1991. 422 pp.	410

These publications may be ordered from: ICES Secretariat, H. C. Andersens Boulevard 44–46, DK-1553 Copenhagen V, Denmark, fax: +45 33 93 42 15, e-mail: info@ices.dk. An invoice including the cost of postage and handling will be sent. Publications are usually dispatched within one week of receipt of payment. Further information about ICES publications, including ordering and payment by credit card, cheques and bank transfer, may be found at: www.ices.dk/pubs.htm.