



REPORT OF THE  
**COMPREHENSIVE FISHERY EVALUATION  
WORKING GROUP**

Key Largo, Florida  
14-21 January 1999

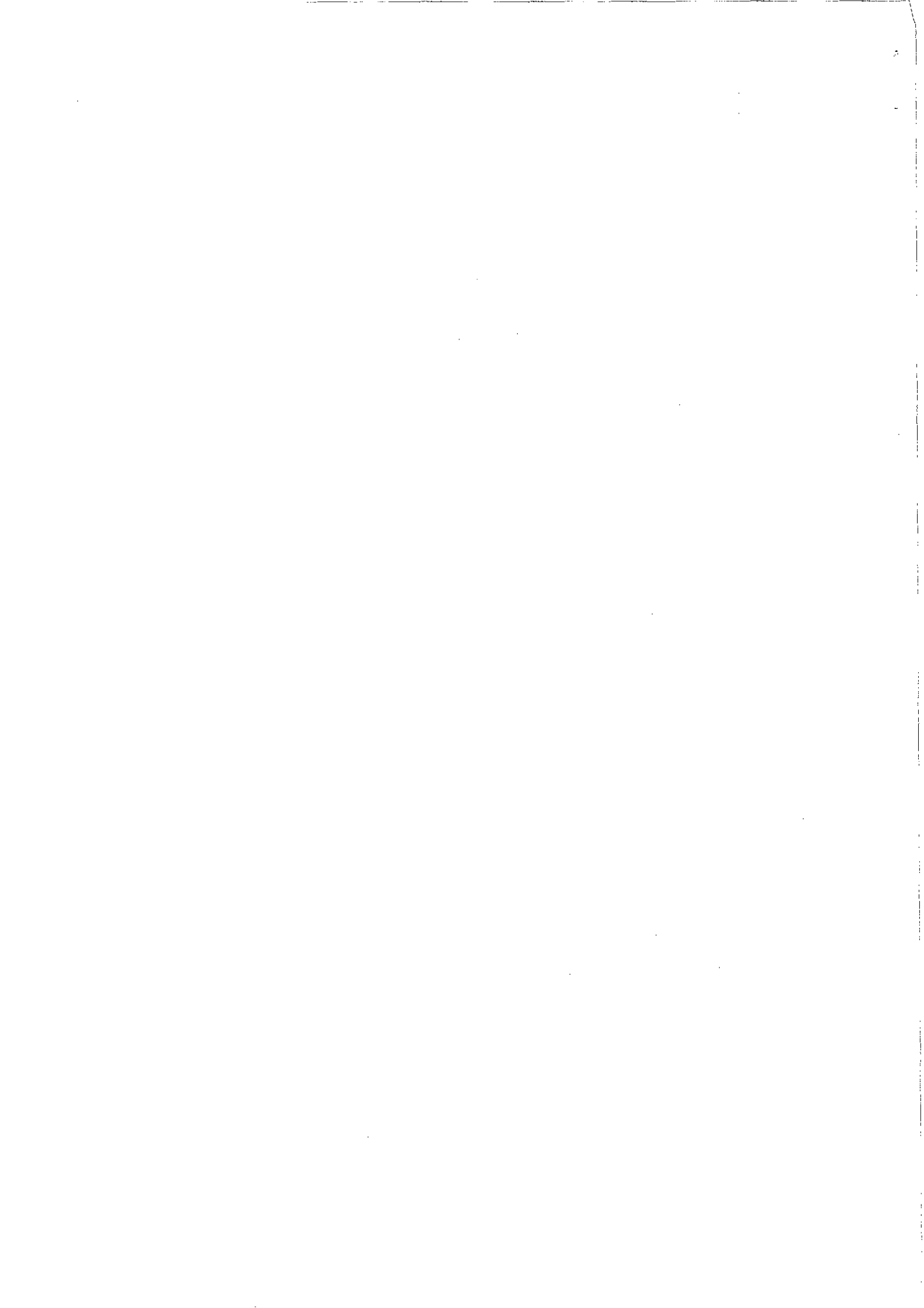
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## 1 INTRODUCTION

### 1.1 Participants

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### 1.2 Terms of reference

It was decided at the 85<sup>th</sup> Annual Science Conference in 1998 that;

The **Comprehensive Fishery Evaluation Working Group [WGCOMP]** (Chair: Dr G. Stefánsson, Iceland) will meet in Miami, FL, USA from 14–21 January 1999 to:

- a) continue the development of tools for the comprehensive evaluation of fisheries, including those taking a mixture of stocks and species;
- b) suggest and evaluate methods for medium-term projections which take into account harvest control rules, including methods capable of addressing fisheries that take a mixture of stocks and species;
- c) suggest and evaluate harvest control rules to be applied for stocks which are harvested in mixed fisheries;
- d) continue the comprehensive evaluation of the following fisheries:
  - i) North Sea flatfish,
  - ii) Norwegian spring-spawning herring,
  - iii) North Sea herring,
  - iv) Icelandic haddock,
  - v) Southern Gulf of St. Lawrence cod,
  - vi) Barents Sea cod;
- e) review the work of the MAWG on the relevance of species interactions to precautionary approaches to fisheries management and rebuilding, and where appropriate carry that work further;
- f) consider the implications of a precautionary approach and harvest control rules in relation to mixed fisheries and technical interactions;

- g) compare and evaluate methods for estimating abundance the oldest age group of "non terminal year" year classes;
- h) compare and evaluate the merits of alternative procedures for bias correction of management quantity estimates.

WGCOMP will report to the Resource Management Committee at the 1999 Annual Science Conference and to ACFM before its May 1999 meeting.

### 1.3 Structure of the report

The terms of reference for the Group are of a varied, yet intertwined nature and this is reflected in how the various items are sometimes visited in different places of the report. Relations to the Precautionary Approach including technical measures are visited in section 2. On the other hand, the effects of environmental changes, and related issues in the single-species case are dealt with in section 3.

The mixed-fisheries aspects (or technical interactions, items (a)-(c) and (e)) are dealt with in Section 4. This section contains examples of how it is possible to implement a harvest control rule which takes into account the mixed nature of some TAC-based fisheries, as well as demonstrates problems involved in not taking this into account. This has particular relevance if the fishery is to be managed in a precautionary manner.

Biological interactions can potentially be a very important issue when considering how to manage fisheries and earlier work on this topic has given mixed signals. This refers to (a)-(c) and (e), and is handled in Section 5.

Sections 6–11 describe the state of affairs and proposed development of comprehensive evaluations of the several fisheries in item (d).

The parts of items (a)-(b) and (g)-(h) which are questions on improving assessments and predictions in a single-species context are handled together as in Section 12.

### 1.4 Overview of a Comprehensive Fishery Evaluation

It was noted at the WG that the word 'comprehensive' in connection with either 'assessment' or 'evaluation' could give rise to substantial confusion about the intended meaning of the set of words.

Basically two dominant interpretations were mentioned:

- *Comprehensive assessment (CA)*: methods/process whereby all relevant data of a fishery system (biology, economy, fishery, management) are analysed within a single framework with the objective to provide general insights into the status and prospects of the fishery system.
- *Comprehensive fishery evaluation (CFE)*: predominantly model-driven approach whereby potential management scenario's are evaluated against putative management objectives under the assumption of knowledge of the underlying systems (biology, economy, fishery).

The choice between the one or the other is determined by the type of question that we want to answer: CA addresses the status (mainly) and prospects of the current fishery system, just as biological assessments attempt to provide insight into the status of stocks. Therefore, a CA does not so much address a specific question but rather a broad question like: what is the status of ...? A CFE is geared towards answering 'what-if' questions, and is therefore more specifically directed towards evaluation of management procedures that are not yet in place. The WG concluded that both concepts can be developed separately but that a comprehensive assessment should in principle precede a comprehensive fishery evaluation. One of the elements of a CFE would be to outline the elements of a CA that would be used in building the operating model in the comprehensive evaluation.

The WG agreed that a comprehensive assessment (CA) should include:

- Historical perspective
- Fisheries
  - Fleet composition and capacity
  - Fleet dynamics: effort allocation, interference, targeting, catchability and technical interactions
  - Gear and selectivity



- Catches, by-catch, catch composition and market sampling
- Discarding, high grading and unreported landings
- Economics
- Environmental impact
  
- Biology
  - Stock structure, spawning and nursery areas, and migration
  - Natural mortality
  - Growth
  - Maturation
  - Fecundity and egg quality
  - Sex ratio
  - Stock and Recruitment
  
- Management
  - International
  - National
  - Control and enforcement
  
- Evaluation of current assessments
  - Biological assessment and advice
  - Economic assessment and advice

The second step towards a Comprehensive Fishery Evaluation would be to add the scenario modelling to former, which should consist of:

- Specification of management objectives
- Specification of management performance criteria
- Definition of the underlying system model (including parameter estimates)
- Definition of the assessment method and the relationship between operating model and perceived system
- Definition of management control rules
- Evaluation of the management control rules with respect to management objectives and performance criteria

## 2 PRECAUTIONARY APPROACH

### 2.1 Recent developments in the precautionary approach within ICES

The Precautionary Approach (PA) has been an important issue for some years. ComFiE was the first ICES working group to consider how the PA could be interpreted in terms of advice on fisheries management and this work is reported in ICES (1996x, 1997x). After the formation and two meetings of the Study Group on the Precautionary Approach to Fishery Management (ICES 1997z and 1998z), this has since lead to the incorporation of the PA into ACFM advice through the use of precautionary and limit reference points.

For a given limit fishing mortality reference point,  $F_{lim}$ , there is a need to ensure that there is low probability of exceeding this reference point, i.e., to ensure that  $P[F > F_{lim}]$  is "small", where  $F$  is the overall fishing mortality in the coming year. If  $F_{lim}$  is assumed known, then this can in principle be accomplished by limiting catches so as to satisfy the probability statement. In particular, if an overall prediction (and implementation) error estimate,  $\sigma$ , is available so that  $\ln F \sim N\{\ln E[F], \sigma^2\}$ , then it should suffice to limit catches to correspond to  $F$  no greater than  $F_{lim} * e^{-1.645*s}$ , where  $s$  is an estimate of  $\sigma$ , in order to have 95% probability that  $F$  will not exceed  $F_{lim}$ . This was the intent when the formula " $F_{pa} = F_{lim} * e^{-1.645*s}$ " was set up and proposed for many stocks.

When implemented as advice for many ICES stocks the approach taken has been to obtain  $F_{lim}$  (often an estimate of  $F_{crash}$  was used) and use this to define a PA reference point,  $F_{pa}$ , designed to keep fishing mortality away from  $F_{lim}$  with a high probability,  $F_{pa} = F_{lim} * e^{-1.645*s}$  where  $s$  is a measure of the perceived uncertainty. When this is done it is clear that for each stock a close look at  $s$  is required to ensure that there really is little probability of exceeding the limit reference point for the stock in questions. For some stocks,  $s = 0.2$  was used with little justification on the grounds that it is

unlikely that the actual prediction uncertainty,  $\sigma$ , is less and hence this s-value should reflect a minimal reduction in fishing mortality needed to obtain advice in accordance with the PA.

Now that  $F_{pa}$  is available for most ICES stocks, an appropriate way forward might be to use (bias-corrected) bootstrapped estimates of the probability involved, or medium-term prediction using appropriate assumptions of uncertainty. Thus it is now possible to obtain more realistic estimates of the true probability of quantities such as  $P[F > F_{lim}]$ ,  $P[F > F_{crash}]$  etc., even taking into account the uncertainty in the estimates of the limit reference points themselves.

Given that the PA is thus already being used in ICES advice (if not in actual catch/effort allocation), there is not an immediate need to work further on the interpretations of the PA in terms of advice or its implementation. Rather, the present meeting has placed an emphasis on the next step, namely the effects of various other issues which have not been considered much before. These include technical interactions, technical measures, biological interactions and changes in biological parameters (or "regime shifts").

On the other hand, in cases where there is no feedback from management on the use of these reference points, the continued use of these in ICES advice will correspond to advising on a specific harvest control rule on a long-term basis. There is a need to try to evaluate each of these harvest control rules and compare them on a stock-by-stock basis to alternative policies (including possibly whatever policy may be implemented by management).

## 2.2 Recent questions on the precautionary approach, given interactions

The above does not take into account any interactions be they biological interactions between species, technological interactions or environmental changes e.g., in the form of "regime shifts". It is well known that if natural mortality increases, estimates of  $F_{0.1}$  may increase considerably. On the other hand, higher  $M$  implies lower surplus production and it may be necessary to reduce  $F$  in order to avoid stock depletion. It is also known that increases in predation may potentially decrease the collapse fishing mortality,  $F_{crash}$ , with corresponding implications for  $F_{pa}$ .

On the whole the PA dictates that uncertainty in these responses should be interpreted to the effect of reducing fishing mortality. The following sections give several examples of the effect of these interactions and how they relate to biological reference points as well as to the PA.

## 2.3 Technical Measures and the Precautionary Approach

Most of the work done in COMFIE when applying the precautionary approach to fisheries has been based on the applying of harvest control rules in TAC based fisheries management systems (ICES 1996x, ICES 1997x). Technical measures (minimum landing size, minimum mesh size, closed areas, closed seasons etc.) have not been in focus, and the scientific basis for the use of precautionary approach is to some extent missing. The number of fisheries which are controlled by technical measures only is, however, fairly high, even though in many cases the total catch volumes might not be very high. In contrast to this, the biodiversity value of the stocks of small scale fisheries might be very important. Due to these facts, it is important to consider precautionary management rules for technical measures.

By using the definition of spawning per recruit (SPR, e.g., Mace and Sissenwine, 1993), the objective of a target  $F$  management rule can be understood as a regulation of spawning per recruit (mortality has a direct effect on the number of times an individual spawns). In the case of technical measures, the control of spawning biomass per recruit is not based on the relationship of predicted and agreed TAC and realised biomass of the stock, but rather on the changes of exploitation pattern (e.g., mesh size and/or minimum landing size rules) or on the direct regulation of fishing mortality (closed areas, closed seasons).

In the case of TAC based fisheries management systems, technical measures can be understood as additional elements which can be used to ensure that a certain minimum SPR will take place even in such cases, where either the assessment or implementation of TAC will totally fail. One could speak about an insurance system. If there are reasons to believe that overfishing may not be detected by scientific investigations or prevented by management actions, or that unpredictable risks may realise, the technical measures may complement the performance of the TAC based management. Lauck *et al.* (1998) discuss this topic in further details in the case of closed areas.

In technical measures there are at least the following positive and negative features:

Positive features:

- 1) In general, the enforcement costs are lower than in a TAC based system. The control is usually based on remarkably lower information and implementation costs than in the case of a working TAC system. However, studies where the information and implementation costs of these two approaches would have been compared, are so far missing.
- 2) The possibility to improve the information robustness of the management system (smaller part of the regulation of SPR is based on TAC system, no need for sudden reaction on the basis of new information). This was demonstrated by a value-of-information analysis in Kuikka *et al.* (1999)

Negative features:

- 1) In a case where there is no yearly monitoring system, there are no possibilities to react on yearly level to changes in fishery or in environment. This uncertainty should be taken into account in the original evaluation of needed extension of management action (e.g., size of mesh size or closed area).
- 2) The production capacity of the stock (growth, maturity, natural mortality) might change after the implementation. On the other hand, in the case of mesh size regulation the selectivity is dependent on the size of the fish, which compensates this problem to some extent.
- 3) Poor survival of the fish escaping through cod-end can offset positive effects of mesh size management. The mortality of escaping fish is discussed in ICES (1998w) and in ICES (1997b).
- 4) Discarding might become an important problem if the selectivity pattern of the gears and the minimum landing size are not in balance. It is clear that the minimum landing size must be less than  $L_{50}$ , for example  $L_{25}$  or  $L_{10}$ . An imbalance between minimum landing size and mesh size will lead to discarding, which has an effect on the conservation efficiency of the action and on the quality of the assessment methods based on total catch estimates.
- 5) In the case of changes in exploitation pattern, the short term effects might be remarkable and the catches might be re-divided between the different groups of fishermen.

The applying of Precautionary Approach can be understood as a set of risk adverse decision rules. Both the exploitation level and the regulation of exploitation pattern can be used as ways of creating a buffer to the structure of the stock against assessment and/or implementation uncertainty. As long as the Y/R basis supports the use of lower F or higher recruitment length, there is no price on the decrease of uncertainty (= insurance fee), but as soon as Y/R basis or S/R basis analysis suggest higher F to maximise yield, there will be a price for the low F in terms of lower catch. This is basic idea of risk adverse behaviour; one is ready to pay something in terms of the expected values to obtain a lower risk. In this respect, e.g., the short term loss is not a scientific argument against the change of a mesh size.

### 2.3.1 Controlling fishing mortality by technical measures

Both closed areas and closed seasons have been used to control the overall fishing mortality of the stock. Even though the total number of papers concerning the closed areas is high, there seem to be relatively few data sets concerning the effectiveness of the closed areas or seasons. It has been suggested, that the size of the closed area should be as much as 50% of the total fishing area to be able to safeguard the recruitment of the population(s). The meta-population context is one of the basis of closed areas: the closed area can be seen as a source of recruitment for the fished areas (see e.g., Allison *et al.*, 1998).

Lauck *et al.* (1998) demonstrates the dependency of the harvesting rate and the required size of the closed area. One of their conclusions is that there is not necessarily a reduction in the catch rates even though a closed area would be implemented.

The control of total effort is another technical option to control fishing mortality. On the short term the implementation uncertainty of this control mechanism might be fairly good, but on the long term the same factors which complicate the use of CPUE data in the tuning of VPA have an effect on the implementation uncertainty of effort control (e.g., the undetectable improvement of catchability, geographical distribution of the biomass, etc.). The modelling of these changes is more closely discussed in Section 12.7.

### 2.3.2 Controlling of exploitation pattern

Minimum landing size, mesh size, hook size etc. regulations are used to control the size at which fish recruit to the fishery, and they have an effect on the exploitation pattern of the population and consequently on SPR. The effectiveness of this type of management is based on the assumption, that the fish escaping from the gear, or the released undersized fish, will survive. At least the survival must be high enough compared to the individual production parameters (growth,  $M$ , reproduction capacity) in order to enable the positive effects to realise. It is evident, that the escape mortality of herring and vendace are very high (Suuronen, 1995), whereas the survival of cod seems to be good. It might be, that mesh size regulation should not be used for such pelagic species which have easily loosed scales. More survival analyses are needed to make general conclusions.

The basic mechanism of the control of exploitation pattern is simple; in a case that recruitment length (or age) is bigger than maturity length, an increase in mesh size increases that proportion of that spawning biomass, which is safeguarded independently of the success of other management actions. A stock where recruitment length is remarkably lower than maturity length is more sensitive to systematic assessment errors than a stock where selectivity and maturity values are close to each others. In a case that recruitment length is clearly higher than maturity age, this difference can be regarded as a safety margin against assessment and management errors. However, more detailed analysis are needed to test these intuitive conclusions.

### 2.3.3 Spawning per recruit and Spawn-at-least-once policy

Mace and Sissenwine (1993) analysed the SPR replacement thresholds and apply the calculus to 83 stocks. They demonstrate the calculus and suggest, that the taxonomic affiliation and life history parameters can be used to select preliminary estimates. This kind of estimates can be considered as Bayesian priors, which will be used as long as new evidence from the stock will update them.

Myers and Mertz (1998) consider the simple management rule of "spawn-at-least-once" policy, where the recruitment age to fishery is modified so, that all individuals of the population will spawn at least once before recruiting to the fishery. The paper is based mainly on theoretical consideration, but to some extent on empiristic generalisations, as well. The paper is based on knife-edge selectivity and on an assumption, that the mature fish do not grow anymore. It presents the use of the difference between the recruitment age and maturity age as a basis for a biological limit of the fishery. Moreover, some simple equations are presented for the evaluation of exploitation limits. This type of information is simple and easily understood by the stakeholders. This rule might be a simple way to screen out the risky species in mixed fisheries with several species.

However, the practical application of spawn-at-least-once policy is not always easy. The change of mesh size, or minimum landing size (for example  $L_{10}$  to correspond with length at maturity) are the main tools for this and in some case the closed areas might have a similar effect.

In addition to the uncertainties mentioned above, the variability of the selectivity of the commercial fleet, uncertainty of growth, maturity and natural mortality should be taken into account when creating a risk averse management rule. The new data sets from the Baltic cod fishery (BACOMA project) indicate, that the selectivity of the commercial fishery is very variable, even for one boat on one area.

In this types of analysis it should be taken into account that in many cases the quality of the eggs of multi-time spawners is higher than that of first time spawners. This fact might to some extent complicate the reliability of Spawn-at-least-once policy, which is based on the data obtained from multi-time spawner data sets.

### 2.3.4 Case study: Trawl and gillnet fishery of Baltic cod

Kuikka *et al.* (1999) analysed the fisheries management of Baltic cod in a decision analysis context by using Bayesian belief network (e.g., Jensen, 1996) as a decision analysis tool. The general idea of the analysis is explained in ICES (1996x). Both the expected catch and the risk of recruitment overfishing were used as interest variables. The analysis was focused on the structural uncertainties of the problem, mainly on the choice of a recruitment model. Three models were used: 1) sampling randomly from observed values 2) Ricker model 3) Environmental Ricker model. In this case, the overall uncertainty increased remarkably when the probability distribution of the different model outcomes were combined in the Bayesian net. In fact, the behaviour of the stock was not anymore predictable with present fishing mortality when the estimates were combined. A decrease in  $F$  or an increase in mesh size made the system more predictable. However, the estimated uncertainties were still underestimates, for example  $M$  was assumed to be known.

One of the main conclusions was, that the information robustness of the management system can be improved by increasing the trawl mesh size from 120 mm to 140 mm. The models presented in the paper are used here to demonstrate the effect of mesh size change on the probability of SBB to fall below a critical threshold. Two mesh sizes (120 mm and 140 mm) and two S/R recruitment models are included to the examples. The first one is a standard Ricker model, and the second one is a model where the variability of the  $\alpha$  parameter of Ricker function was modified to mimic the inflow of saline and oxygen water to the Baltic Sea. This environmental process includes high values (year of inflow) and short decreasing trends after the inflow (decrease of salinity and oxygen in the deeps). Moreover, there is a general consensus amongst the marine biologist that the future water quality will be worse than it has been in the past. This simulation offers one possibility to demonstrate the effects of environmental uncertainty on the biological thresholds.

The details of the modelling are shortly explained here. The simulations were carried out by using a deterministic exploitation pattern, which was calculated by applying empiristic selectivity values and a length distribution model where lengths are converted to weights by a length - weight relationship. The selectivities of two gears (gillnet and trawl) were included. This part of the calculus was totally deterministic. This process resulted to weight and maturity values which were gear and F specific (F has an effect on the length distribution). These values were used in a Monte Carlo simulation model which was run over 110 years and the last 100 years were used to estimate how many times the SSB fell below the critical SSB (two alternative values were used: 240 000 and 480 000 t, the latter is used in the example here). This simulation was carried out 600 times. The probability of SSB falling below the threshold, was calculated.

The results of these simulations (two different S/R models and two different mesh sizes with five levels of exploitation) are given in Figure 2.3.1. The mesh size change has a clear effect on the probabilities. For example, by using the environmental model, the exploitation level of 0.8 for 120 mm gives a similar risk profile as the exploitation level 1.0 for 140 mm mesh size. With the bigger mesh size and present fishing pressure (= fishing mortality of 1996), there is very low probability for biomass to fall below the threshold. However, the probability increases clearly, if the environmental Ricker model is used as a yardstick. There is still a high probability to fall below the threshold, even though 140 mm mesh size would be used. Also F restrictions would be required. It must be noticed, that in this model the mean quality of the environment was also assumed to deteriorate, so these results are not dependent on the trends only. For example Lande *et al.* (1997) have also shown that the environmental variability has an effect on the biological thresholds.

In this case there was no price for the decrease of the risk when changing to the bigger mesh size (except the short term effects). In contrast, the expected yield would be 25% higher by changing the mesh size only. On the yield per recruit basis (no positive feed back from higher biomass) this increase is estimated to be 14% (with present growth). Thus, a remarkable part of the improvement is based on the improvement of SPR. The maturity length of Baltic cod was assumed to be 40 cm, which is  $L_{30}$  on the selectivity curve of diamond mesh 140 mm.

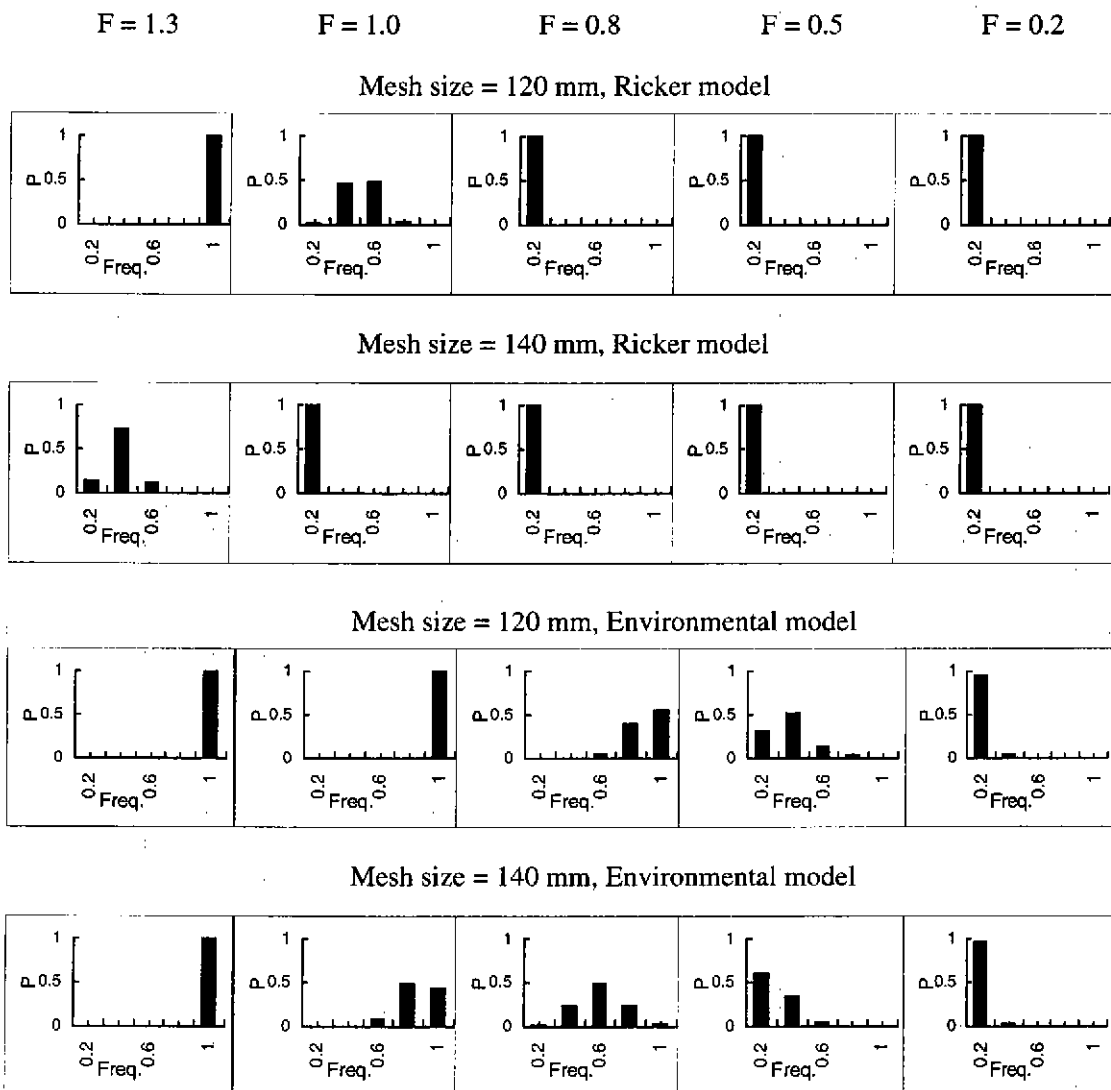
In order to demonstrate the SPR calculus on different mesh sizes, the SPR (e.g., Mace and Sissenwine, 1993) was estimated for 4 different mesh sizes (selectivities for trawl from BACOMA project, unpublished data and for gillnet of 105 mm from DIFTA) and several exploitation rates. A length distribution model was applied (from Kuikka *et al.*, 1999), in which both the maturity and retention rates of different mesh sizes can be modelled on length basis. This gives a possibility for a more exact modelling of the selectivity process compared to a standard Y/R model, and the mean weights in the population and in the catch can be calculated separately, which is important if the selection and exploitation rates are high. Moreover, the additional positive element is, that a mesh size can be implemented directly, whereas a knife recruitment age is impossible to implement in practice.

In this analysis, the relationship of the relative F rates of gillnet and trawl were fixed and the maturity length of cod was assumed to be 40 cm.

The results for two growth rates and four mesh sizes are given in Table 2.3.1. The results demonstrate, that the increase in the selectivity values increases that area of F which is above the replacement% SPR, i.e., the safety area. This makes the management more information robust against the errors in the estimation and implementation of F. This area is highly dependent on the growth, i.e., higher growth rates decrease the need for a reliable control of F. In the case of Baltic cod, the improvement of the growth rate at the same time when the biomass decreased during the last 10 years (Gislason, unpubl.) might have a connection to the fact that the CPUE is still on an economically possible level.

## Recommendations

The uncertainties in the biological parameters of spawning per recruit (growth, maturity, natural mortality), in the selectivity of commercial fleets, and in the survival of the escapees must be studied more in order to be able to create precautionary technical measurement rules in single species fisheries. Moreover, the effects of technical measures on biological parameters should be studied, as well.



**Figure 2.3.1.** The effect of mesh size and S/R model on the probability distribution of SSB being below 480 000 tons in the Eastern Baltic Sea cod stock. The figure describes the probability distribution of SSB being below the threshold. The risky area is on the right. The F is the coefficient of the age specific F values. A good growth rate (= near the present one) is assumed.

**Table 2.3.1** Application of the findings of Mace and Sissenwine (1993) to the selectivity data of the Baltic Sea cod (preliminary selectivity estimates). The %SPR value closest to the F (0.1) of Mace and Sissenwine (39.4% for Atlantic cod) is underlined and the values closest to replacement %SPR are given by bold. The highest and lowest observed growth rates are applied. The relative F 1.0 = the situation in 1996.

Relative F	Exit window 105 mm	Exit window 117 mm	Diamond 120 mm	Diamond 140 mm
<b>Good growth rate</b>				
0.0	100.0%	100.0%	100.0%	100.0%
0.2	57.8%	61.5%	58.5%	62.4%
0.4	<u>36.0%</u>	<u>40.7%</u>	<u>36.9%</u>	<u>41.8%</u>
0.6	24.1%	28.9%	25.0%	29.9%
0.8	17.3%	21.8%	18.1%	22.7%
1.0	13.1%	17.4%	13.9%	18.1%
1.2	10.4%	14.4%	11.1%	15.0%
1.4	8.6%	12.4%	9.2%	12.8%
1.6	7.2%	10.9%	7.9%	11.3%
1.8	<b>6.3%</b>	9.8%	<b>6.9%</b>	10.1%
2.0	5.5%	8.9%	6.1%	9.1%
<b>Poor growth rate</b>				
0	100.0%	100.0%	100.0%	100.0%
0.2	<u>34.1%</u>	<u>43.1%</u>	<u>35.5%</u>	<u>44.8%</u>
0.4	15.6%	24.0%	16.9%	25.1%
0.6	9.1%	16.2%	10.1%	16.7%
0.8	<b>6.1%</b>	12.4%	<b>6.9%</b>	12.5%
1	4.5%	10.2%	5.2%	10.0%
1.2	3.5%	8.8%	4.1%	8.4%
1.4	2.8%	7.8%	3.4%	7.3%
1.6	2.3%	7.0%	2.8%	<b>6.5%</b>
1.8	1.9%	<b>6.4%</b>	2.4%	5.8%
2	1.6%	5.9%	2.1%	5.3%

#### 2.4 The Relationship Between Minimum Landing Size, Minimum Mesh Size And Closed Areas In A Mixed Fishery Environment

New technical regulations have been agreed by the European Commission management of the North Sea flatfish fishery. One of the elements of the regulation is a reduction in the minimum landing-size (MLS) for plaice from 27 cm to 22 cm. The regulation is to be implemented at the start of the year 2000. The ICES North Sea Demersal WG (WGNSSK) has been asked to evaluate the likely consequences of this change in regulation.

Plaice is caught in a mixed fishery with sole where the latter is the main target species (due to higher price). There are three technical measures that interact in this fishery:

- Minimum mesh size of 80 mm (south of 55° N) and 100 mm (north of 55° N),
- Restriction on effort and gear in the 12 mile zone and a closed area off the Dutch, German and Danish coast (the 'plaice box')
- Current minimum landing sizes of 27 cm (plaice) and 24 cm (sole).

In order to evaluate the effectiveness of protecting juvenile flatfish from discarding (the stated objective of the technical measures) using the measures mentioned above, the following questions need to be answered:

- How do the minimum landing sizes relate to the theoretical gear selection curves?
- How will fishermen behave in reaction to the new technical measures?

Some possible behavioural responses by fishermen include:

- Continue to fish and discard as usual, ignoring the minimum landings size
- Market plaice of 22–27 cm but otherwise fish as usual
- Develop a directed fishery for small plaice

It is recognised that no definitive answers or methods can be supplied by the COMFIE WG at this stage. However, the WG has tried to outline an approach that could be followed in the near future to address the above questions.

There is a need to establish a theoretical approach to allow for the evaluation of the three technical management measures. Before such an analysis could be performed it would be necessary to have access to the objectives that underlie the measures which, if known, could be evaluated using the simulation tools as outlined in Section 6.7 and used in Section 4.2.

### 3 SINGLE SPECIES BIOLOGICAL REFERENCE POINTS

#### 3.1 Introduction

Stocks and fisheries experience changes through time in many of the variables that affect their productivity. These changes could be more or less without a trend (“white noise”) or could follow trends in cycles of 5–10 years, or more. In some cases, the changes will be readily apparent; in others, it will take much retrospective examination to postulate that there were changes in productivity. In this section we examine the need for re-computing biological reference points given that there are observed or perceived changes in the characteristics of the fishery and the stock.

Two classes of reference points were considered. Standard “per recruit” reference points, namely  $F_{0.1}$ ,  $F_{max}$  and  $F_{35\%}$  S/R were estimated using inputs of  $M$  natural mortality ( $M$ ), weight at age and  $F$  at age (selectivity). Age structured stock production reference points (ICES 1997x) were also estimated, including  $F_{msy}$ ,  $F_{crash}$  and  $B_{msy}$ . The production analyses used the same inputs as per recruit plus information from stock recruitment relationships. These relationships were fit with both the Ricker and Beverton/Holt models.

Data from two stocks were used as case studies, southern Gulf of St. Lawrence (SGSL) cod and Icelandic haddock. The SGSL cod stock data series extends from 1950–97. There have been large scale changes in size at age for this stock. Historical highs were observed in the late 1970s while current values are the lowest seen (see Section 10). There is also evidence that  $M$  may have changed in recent years. The Icelandic haddock time series extends from 1978–97.

The analyses that follow are based on the presumption that a given assessment is an accurate portrayal of the “true” situation, i.e., no simulations were used to generate known values. The following procedures were used for the computations:

*Resampling:* Non-parametric, based on the input observations. For the stock-recruitment data, S-R pairs from the stock assessment were re-sampled. For the selectivity, maturity and weight, age-specific vectors were resampled, keeping consistency in the values so that they corresponded to the same year. In the case that selectivity or weight was held constant, it was set equal to the median age-specific value for a fixed time period (a 5-year period ending in the “current” year, or the entire time series, depending on the analysis).

*Stock-recruitment relationships (SRR).* Beverton and Holt, parameterised as  $R = \frac{1}{a + \frac{b}{SSB}}$ , where the slope at the

origin is given by  $1/b$ , and the Ricker parameterised in the usual form of  $R = a SSB e^{-b SSB}$ , where the slope at the origin is given by  $a$ .

*Error assumption.* For convenience, most of the bootstrap results presented in this section were based on linear fits of the stock recruitment functions. These were obtained starting with linear transformations as

$$\ln(R/SSB) = \ln(a) - b SSB \text{ for the Ricker, and}$$

$$1/R = a + b/SSB \text{ for the Beverton-Holt.}$$



Using simple linear regression, the Ricker transformation used is equivalent to a nonlinear regression assuming lognormal errors on the recruits. This is not the case for the linearly-transformed BH relationship, however. For comparison, a limited number of bootstraps were conducted with nonlinear least squares fits of the BH relationship assuming lognormal errors on the recruits (see Section 3.2.2).

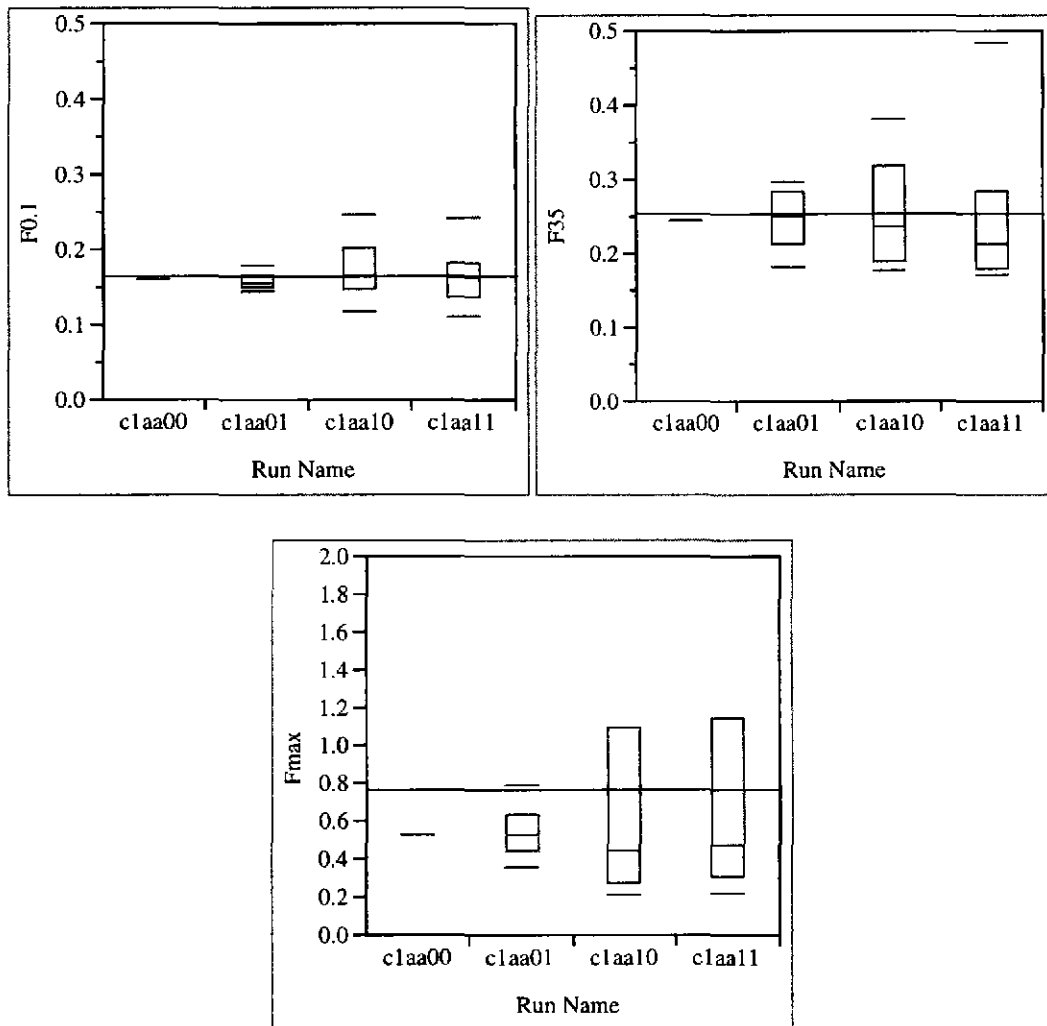
*Biological reference points.* Computed with Fishlab® using standard equations. MSY-related estimates are age-structured, assuming a given stock-recruitment relationship. In any given bootstrap, the SRR assumed was that given by the bootstrap-specific parameter estimates, i.e., no additional residual variance was superimposed.

### 3.2 Southern Gulf of St. Lawrence Cod

#### 3.2.1 Impact of Variation in Weight and F at Age on Per Recruit Reference Points

In these simulations, the entire time series of weight at age and F at age were resampled for bootstrap estimates. Years were chosen at random and the weight and F at age vectors from that year were used as inputs to the calculation of reference points.

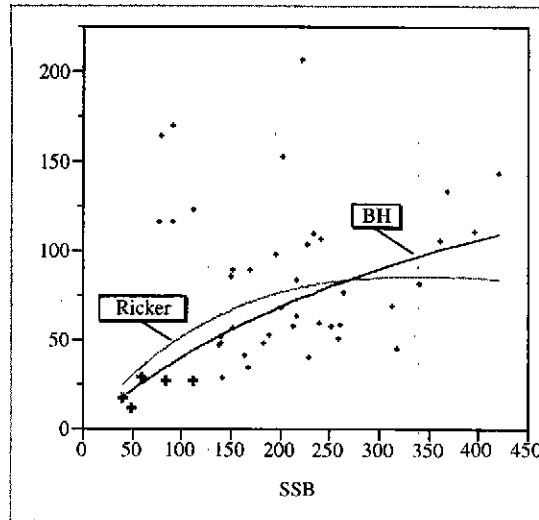
Per-recruit reference points were impacted primarily by the observed variation in weight at age (Figure 3.2.1). There was little difference in the spread of estimates between the runs where the weight at age was resampled and where both weight at age and F at age were resampled. This is likely due to the higher dynamic range in weight at age in the time series, but there may also be covariance in weight at age and F at age since vectors from the same years were chosen for both.



**Figure 3.2.1:** Bootstrap estimates of  $F_{0.1}$ ,  $F_{max}$  and  $F_{35\%}$  SSB/R for SGSL cod using the entire time series of weight at age and F at age. The 4 runs were for different combinations of error. The last 2 digits in the run labels indicate error in weight at age (0 = no bootstrap, 1 = bootstrap) and error in F at age (0 = no bootstrap, 1 = bootstrap). The box encloses the middle 50% of the observations and the horizontal lines show the middle 80%.

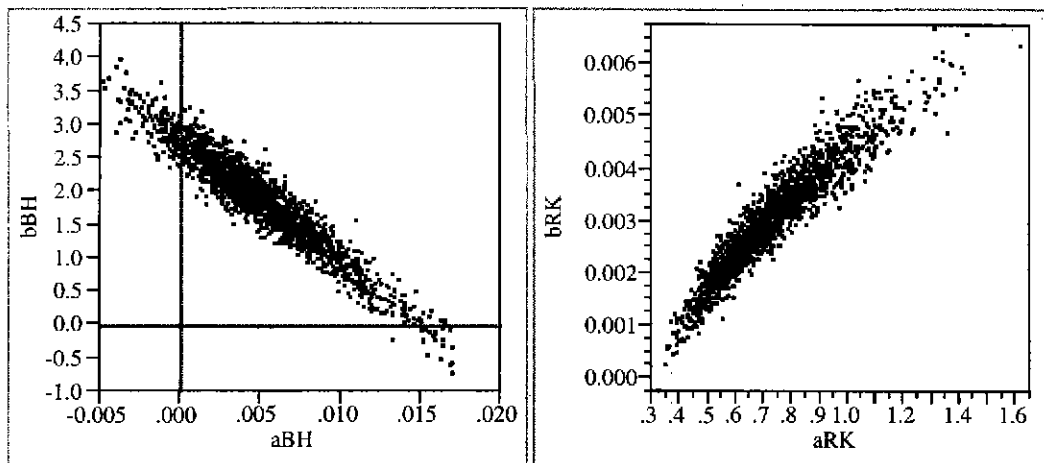
### 3.2.2 Impact of Variation in SR data, Weight at Age, and F at Age on Production Reference Points

Estimates of production-based reference points are dependent on stock/recruitment (S/R) data as well as biological variables of weight at age and F at age. In addition, the choice of analytical model (Ricker vs. B/H) and assumptions about error structure can be important. The procedure used here was to select life history data (F and weight) as described in section 3.2.1 and to bootstrap SR pairs from the entire time series. The assessment (SPA) results for SGSL cod assuming  $M = 0.2$  were used. The scatter of R/S data for SGSL cod show high variability typical of most of these relationships (Figure 3.2.2).



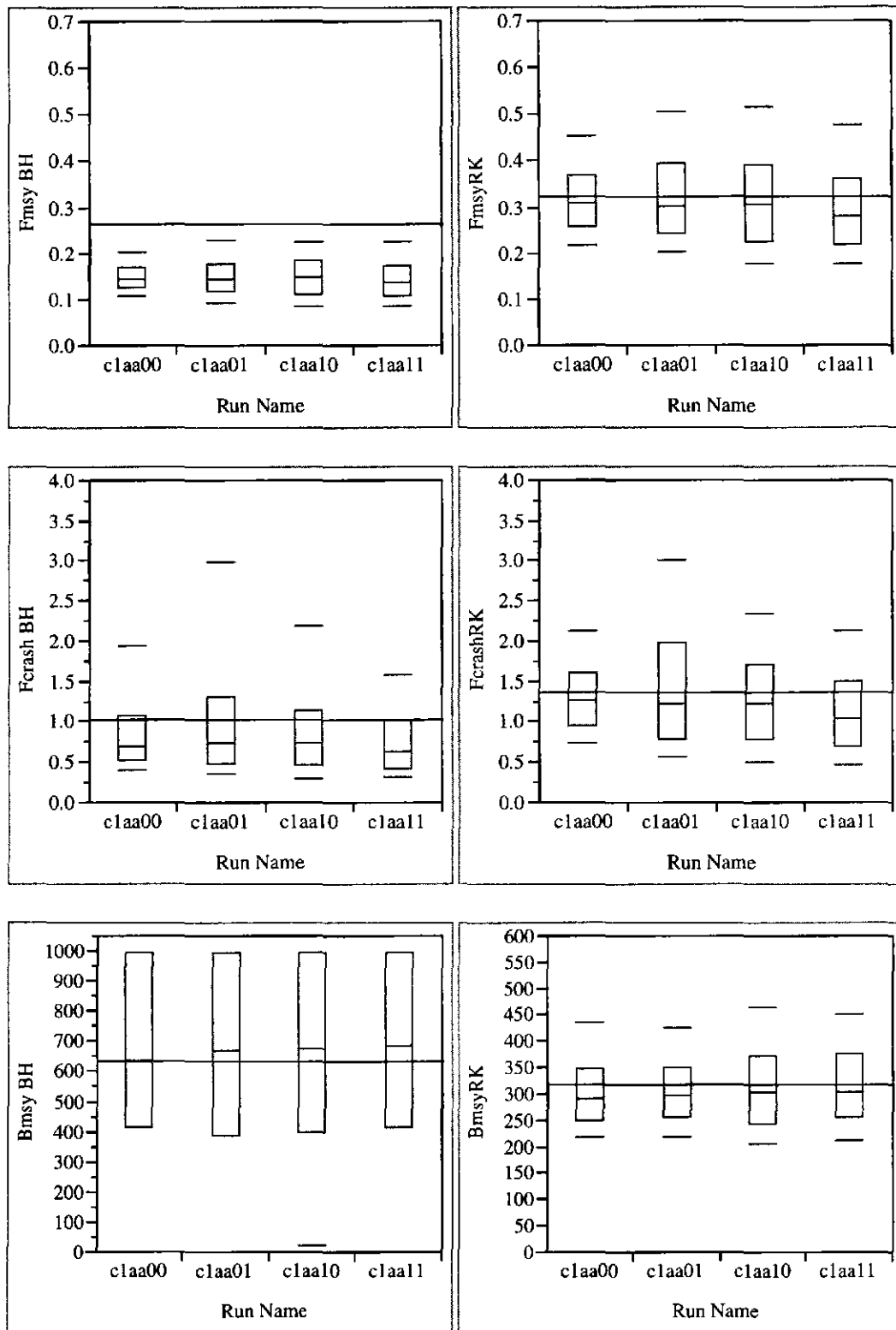
**Figure 3.2.2.** Stock recruitment data for SGSL cod from an SPA assuming  $M = 0.2$ . The fitted Ricker and Beverton/Holt (linear transformation) relationships are shown. The last 5 S/R pairs in the data are shown in bolder type and are near the origin.

The scatter of SR relationship parameter estimates for the linearly-transformed Beverton/Holt and Ricker relationships are shown in Figure 3.2.3. The B/H model produced several negative parameter estimates, and this resulted in a large number of nonsense estimates of production based reference points (no constraints were placed on the linear regression procedure to avoid negative parameter estimates). These plots also indicate the degree of correlation among the parameters.



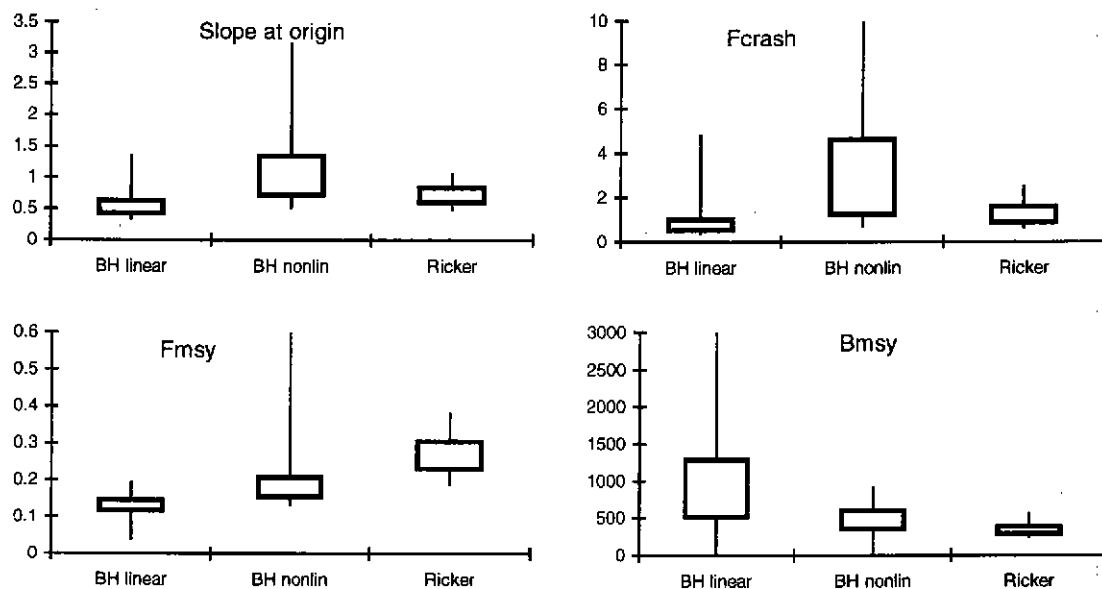
**Figure 3.2.3.** Comparison of linearly-transformed Beverton Holt (left) and Ricker (right) SR parameter estimates from 2000 bootstrap trials using the entire SGSL cod SPA time series, assuming  $M = 0.2$ . The distributions are not scattered in the same direction because the parameterizations differ between the two relationships (see Section 3.1).

When the linearly-transformed BH relationship was fit in a reasonable manner (i.e., positive parameter estimates), it produced more conservative production-based reference points (i.e., lower  $F_{msy}$  and  $F_{crash}$  estimates, higher  $B_{msy}$  estimates) than those associated with the Ricker relationship (Figure 3.2.4). This is likely the result of the lower slope at the origin of the linearly-transformed BH relationship (Figures 3.3.1 and 3.2.5). Production reference points were relatively insensitive to random variations in weight at age and  $F$  at age (Figure 3.2.4).



**Figure 3.2.4:** Comparison of bootstrap estimates of  $F_{msy}$ ,  $F_{crash}$ , and  $B_{msy}$  estimated assuming either a Beverton/Holt (left side) or Ricker (right side) stock/recruitment relationship. Data came from the entire SGSL cod SPA time series, assuming  $M = 0.2$ .

The impact of the error assumption was further investigated using the Beverton and Holt model. Five hundred bootstraps were made with resampling only the S-R data pairs, and using the median selectivity and weight vectors for the period 1980–1984. The BH model was fit with the linear transformation presented above and also by nonlinear least squares, assuming lognormal errors. For comparison, the Ricker SRR was fitted with the same linear transformation as used above, which is equivalent to a nonlinear fit assuming lognormal errors. Figure 3.2.5 shows summaries of the resulting distributions of the slope at the origin,  $F_{msy}$ ,  $F_{crash}$  and  $B_{msy}$  for the various SRRs. It is apparent that the error assumption can have a substantial impact on the estimates. For this example, the differences in the magnitude of the slope at the origin are greater between error assumptions (BH model) than between S-R models (compare distributions in upper LHS panel of Figure 3.2.5). In terms of estimates of  $F_{msy}$ , the lognormal error-assumption BH results are somewhat intermediate to those between the linearly-transformed BH and the lognormal Ricker fits.



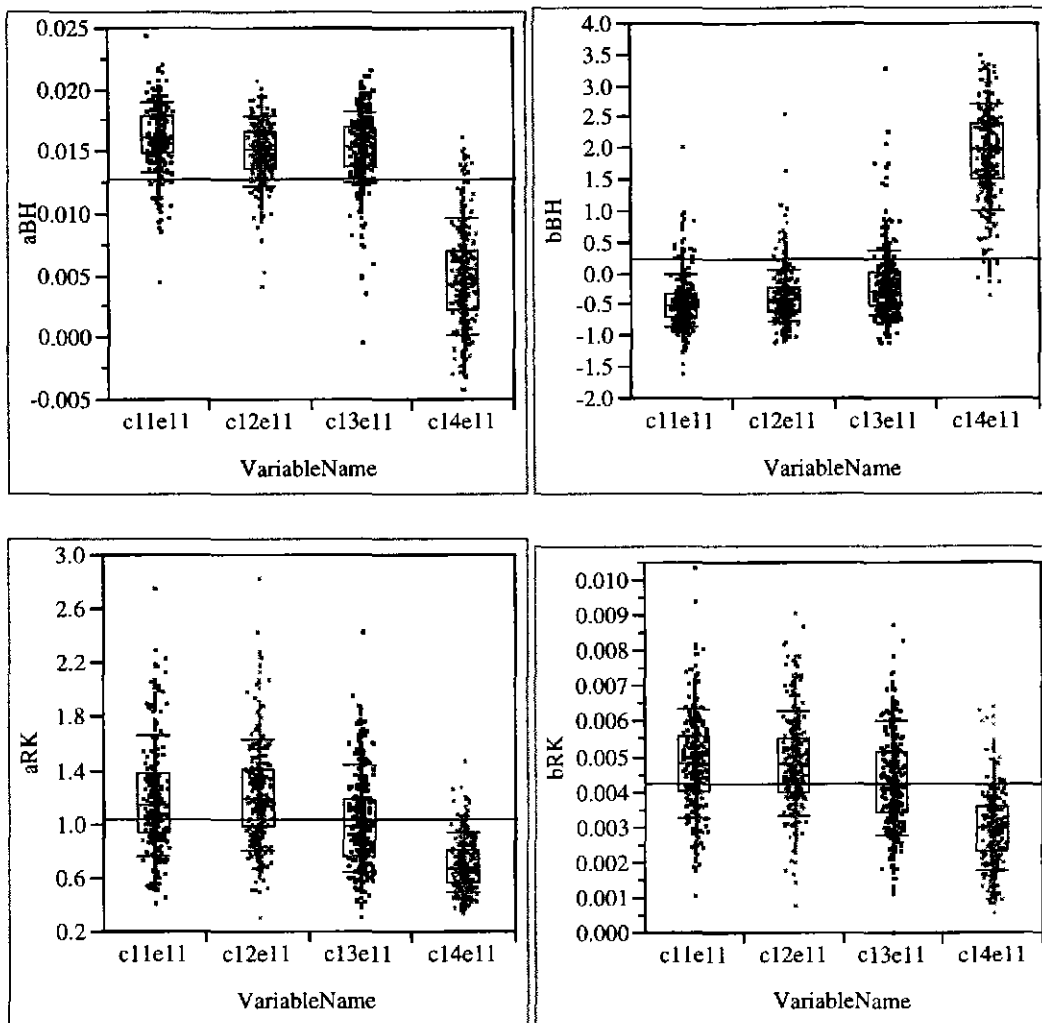
**Figure 3.2.5.** Distributions of bootstrap results for various reference points using the linearly-transformed Beverton-Holt model, a BH model estimated via non-linear least squares assuming lognormal errors, and a Ricker model assuming lognormal errors. The boxes show the range of the centermost 50% of the bootstraps, and the lines are approximate 90% limits.

### 3.2.3 Accounting for Changes in Weight and Selectivity at age over Time

Changes in life history characteristics of a stock over time may necessitate recalculating biological reference points. Changes in length and selectivity at age of SGSL cod have been described at previous meetings of COMFIE.

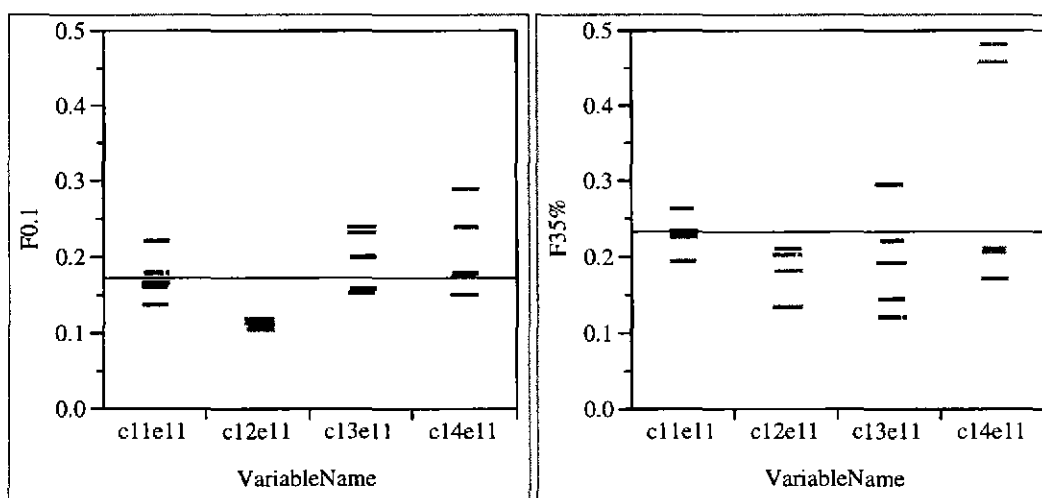
Bootstrapping was used to estimate reference points for 4, 5-year time periods. The SR data were selected from the periods 1950 to 1982, 1987, 1992, and 1997. Weight at age and  $F$  at age vectors were selected from 5 year windows, 1978–82, 1983–87, 1988–92, 1993–97. Results are shown for these time periods respectively indicated by the third digit in the run codes.

The linearly-transformed Beverton/Holt SR fits were generally unreliable for the first 3 time periods. The majority of  $b$  parameter estimates were negative (Figure 3.2.6). The addition of the last 5 years of SR data, which were near the origin (Figure 3.2.2) improved the estimations, almost all of the  $b$  estimates were positive. However, several  $a$  parameter estimates were negative in this time period. For the Ricker fits, the addition of the last 5 years of SR data resulted in reduced  $a$  and  $b$  parameter estimates.



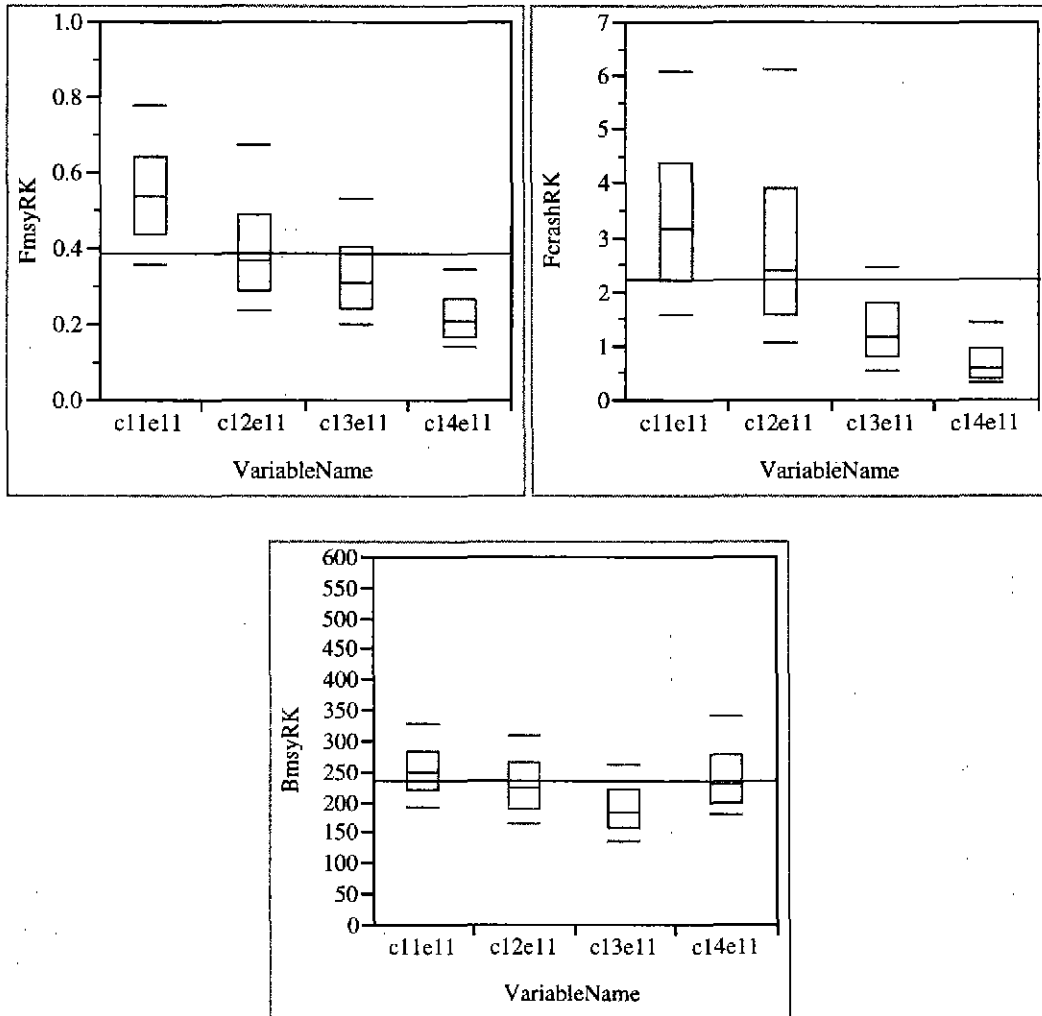
**Figure 3.2.6:** Stock recruitment parameter estimates for different time periods of the SGSL cod SPA results assuming  $M = 0.2$ . Parameters from the linearly-transformed Beverton/Holt relationship are shown in the top panels while the Ricker parameters are shown on the bottom. The periods used for bootstrapping ended in 1982, 1987, 1992, and 1997 respectively.

There were only 5 combinations of weight at age and  $F$  at age available for each time period, and thus there were only 5 possible per recruit reference point estimates from bootstrapping in these time periods. There was little variation in these estimates with the exception of increases in  $F_{35\%}$  in the last time period (Figure 3.2.7).



**Figure 3.2.7:** Bootstrap estimates of per recruit reference points from 4, 5-year time periods of the SGSL cod time series. These time periods were 1978–82, 1983–87, 1988–92, 1993–97.

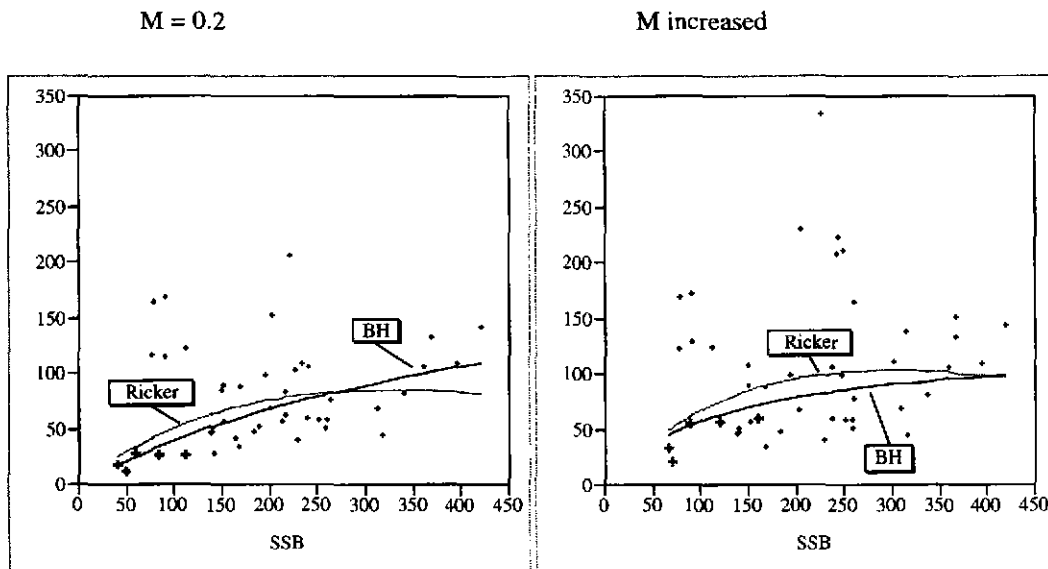
Production based  $F$  reference points declined in the 4 time periods due mainly to reduced weights at age (Figure 3.2.8). The magnitude of the decline was large relative to the standard deviation of the estimates, suggesting this was an important change. There was little change in the estimated  $B_{msy}$  reference points, suggesting some compensation between the reduced  $F_{msy}$  and the associated equilibrium biomass.



**Figure 3.2.8:** Changes in production-based reference points estimated with a Ricker SR relationship and from input data selected from 4 time periods. Input data were taken from the SGSL cod stock.

### 3.2.4 Impact of Changes in $M$ on Reference Points

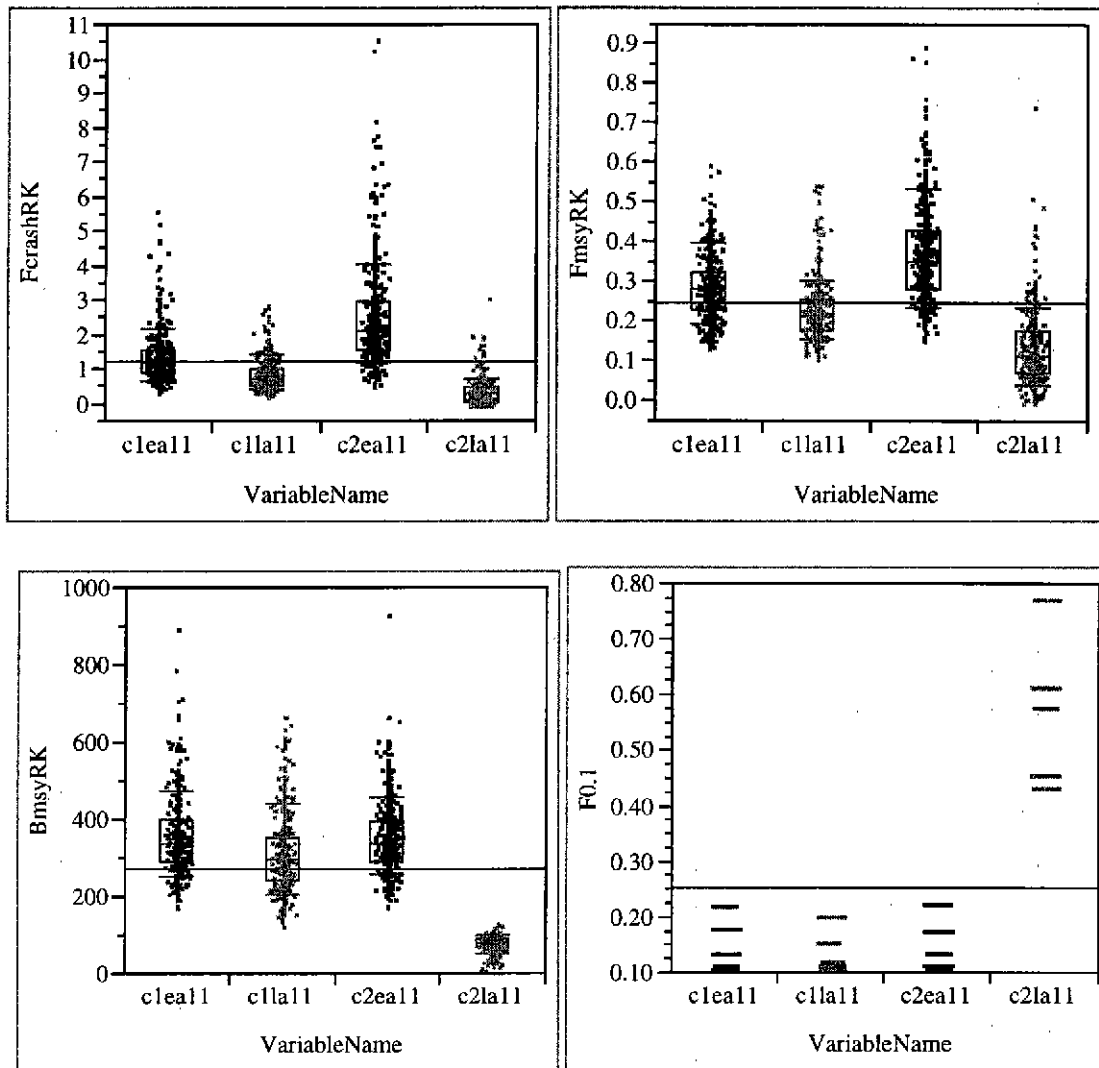
There is some evidence to suggest that  $M$  may have changed for the SGSL cod stock (see section 10). Since  $M$  is an important input in reference point estimation, the sensitivity of estimates to changes in  $M$  was investigated. Input SR data were obtained from 2 separate SPAs, one where  $M$  was assumed to be constant at 0.2 and a second where  $M$  was 0.2 from 1950–84 and 0.4 from 1985–97. The respective SR plots are shown in Figure 3.2.9.



**Figure 3.2.9:** Stock/recruitment data for SGSL cod from 2 SPAs. The frame on the left is for an SPA where  $M$  is assumed to be 0.2. The frame on the right is for an SPA where  $M = 0.2$  for 1950–84 and  $M = 0.4$  for 1985–97. The fitted Ricker and linearly-transformed Beverton/Holt curves are shown. The last 5 data pairs are shown in bolder type, and they are near the origin.

Test bootstrap runs were made as follows. Bootstrap SR data were drawn from the entire time period. Weight and  $F$  at age were taken from the last 5 years of each period (1980–84 and 1985–89 for periods e and l, respectively, designated by the third digit of the run names). Reference points were calculated assuming the  $M$  values used in the assessment (fixed at 0.2 or changing from 0.2 to 0.4 in 194; runs designated by a 1 or a 2, respectively, in the second digit of the run name).

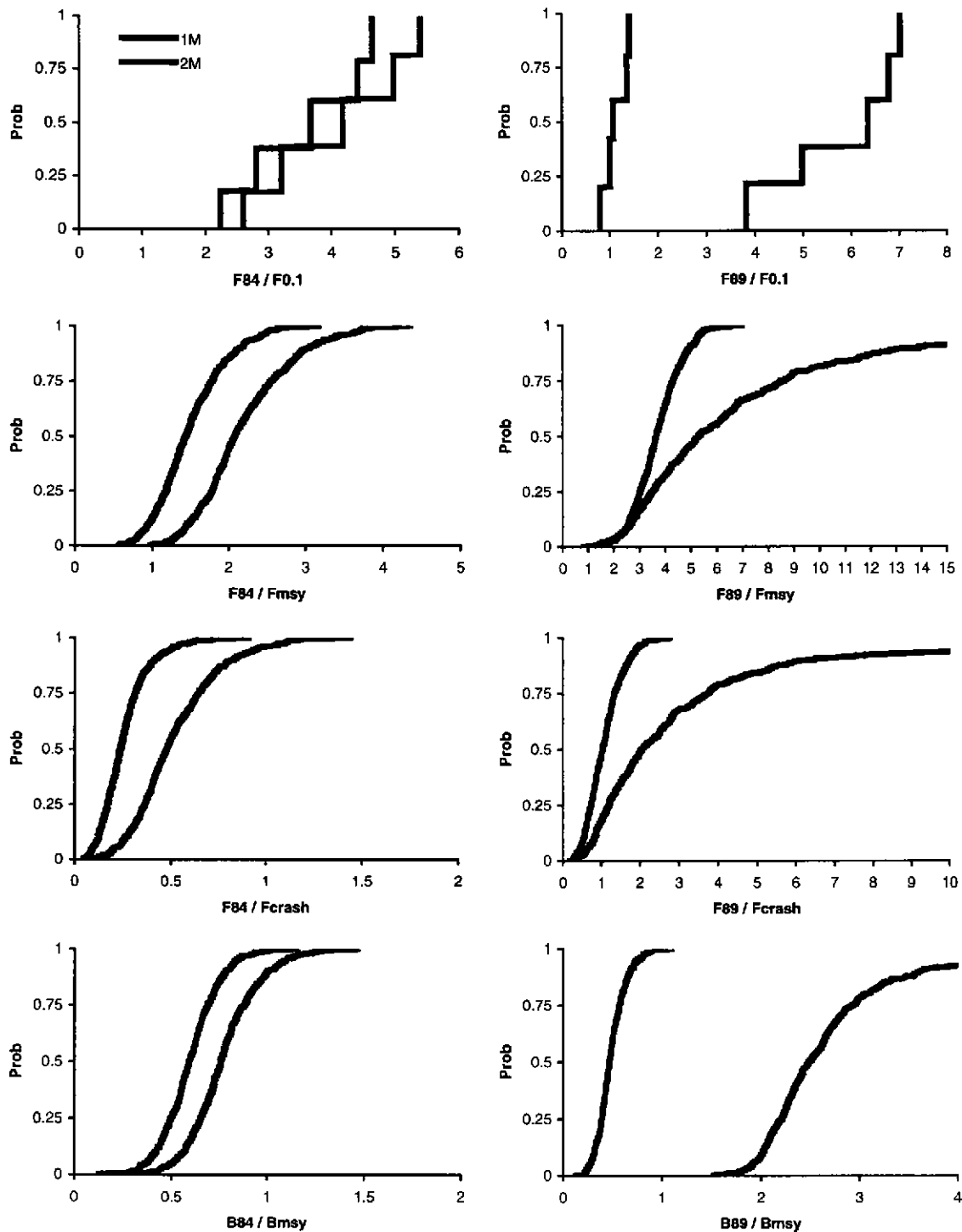
When  $M$  was assumed to be constant, there was a small decline in the estimates of  $F_{\text{crash}}$ ,  $F_{\text{msy}}$  and  $B_{\text{msy}}$  between periods 1 and 2 (Figure 3.2.10). There was very little difference in the estimated  $F_{0.1}$ . There were large declines in the production based reference points, however, when  $M$  was assumed to change. At the same time there was a large increase in the estimated  $F_{0.1}$ . Essentially, a production model would suggest to fish less if  $M$  was higher while a yield per recruit model would say fish harder to catch the fish before they die.



**Figure 3.2.10:** Reference point estimates for different time periods and assumptions about  $M$ . The key to run names is as follows: digit 1, c = cod; digit 2, 1-assumed  $M = 0.2$ , 2 = 2  $M$ 's; digit 3, e = 1980–84, l-1985–89. To summarise, the first 2 runs in each frame are for reference points where  $M$  is assumed to be fixed at 0.2 in all years. The second 2 are for the scenario where  $M$  is assumed to be 0.2 up to 1984 and 0.4 for 1985 and thereafter. The first and third are for reference points for the early time period while the second and fourth are for reference points in the recent period.

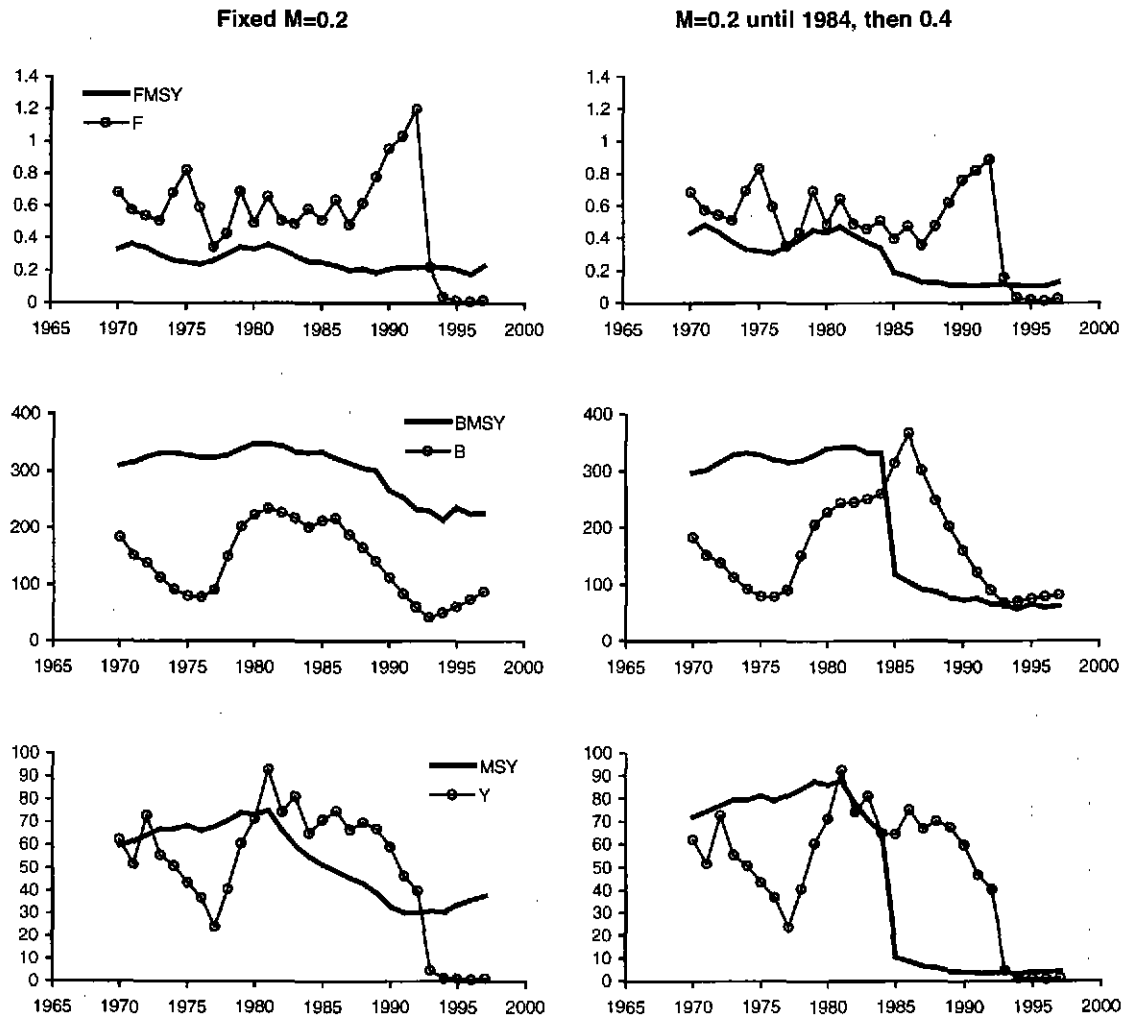
Whether MSY-related quantities should be computed using 5-year means or annual vectors for this stock seems to be a trivial question relative to the effect of the doubling of  $M$ . What happens when this change is not accounted for? Figure 3.2.11 shows the perceived probability that current  $F$  or current  $B$  ("current" in 1984 or 1989) exceeds various reference points in the year before the  $M$  change, and 5 years after. The comparisons assume that the stock recruitment relationship was not altered by the change in  $M$ , e.g., by a change in survival rates of recruits.





**Figure 3.2.11.** Probability profiles for stock status exceeding various reference points, SG cod example. Grey curves ignore a doubling of  $M$  that occurred in 1985; dark lines capture the real change.

More insight into this is gained from examining the time trajectories in various quantities (Figure 3.2.12). The perception of the trend in  $F$  relative to  $F_{msy}$  is similar whether the change in  $M$  is captured well or not, although the one capturing the change is somewhat more optimistic. The perception of SSB relative to  $B_{msy}$  is strikingly different between the two cases. It should be noted that the extreme drop in  $B_{msy}$  is an artefact of the assumed knife-edged change in  $M$ , when it is more likely that such a change could have occurred over several years. The case that ignores the change in  $M$  would lead one to conclude that SSB has been maintained below  $B_{msy}$  and that  $B_{msy}$  has decreased somewhat since 1980. In contrast, the case that captures the doubling of  $M$  would indicate that, as the stock's productivity declined by about 65%, the recent situation has been one of SSB declining rapidly towards a much lower equilibrium. Similarly, a look at Yield compared to MSY suggests that ignoring the change in  $M$  underestimates the amount of surplus production that was taken between 1985–1992.



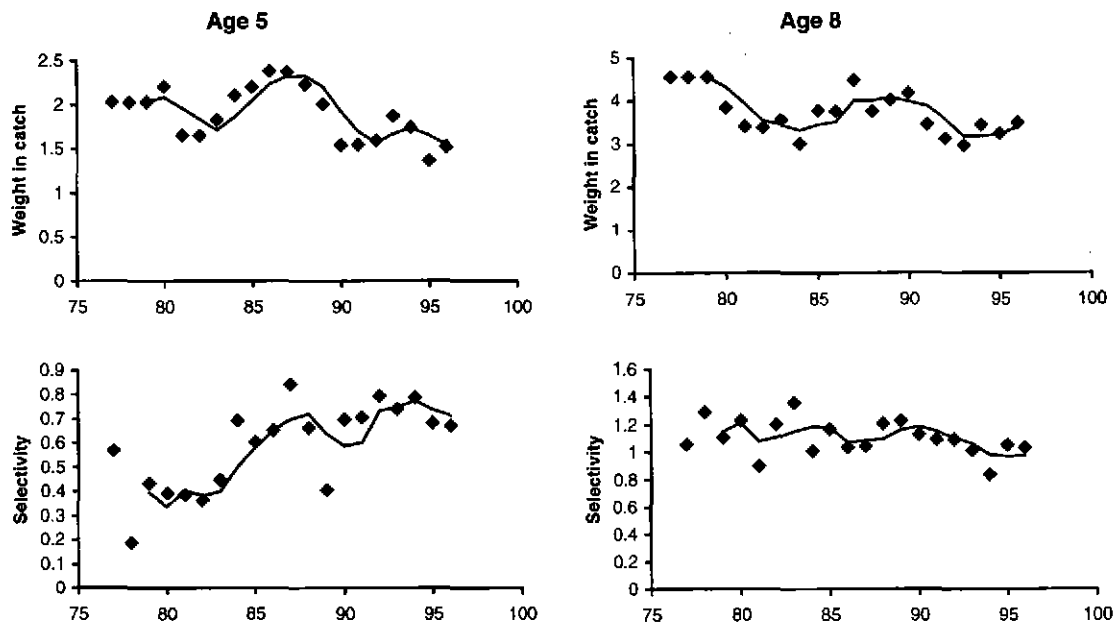
**Figure 3.2.12:** Trends in stock status and equilibrium reference points for SG cod, assuming a fixed  $M$  or an  $M$  that doubled in value in 1985.

These results suggest that:

- Changes in per-recruit reference points may be in the opposite direction to changes in SRR-based reference points: In the last example, ignoring the change in  $M$  after it occurs, leads to the perception that  $F > F_{0.1}$  when it isn't.
- Conversely, ignoring the change in  $M$  leads to overly optimistic perception of stock status.
- This stock appears to be suffering from low growth production given that weights at age are the lowest observed, and from reduced surplus production due to an increase in  $M$ . The net result is that there is very little surplus production in the stock at this time.

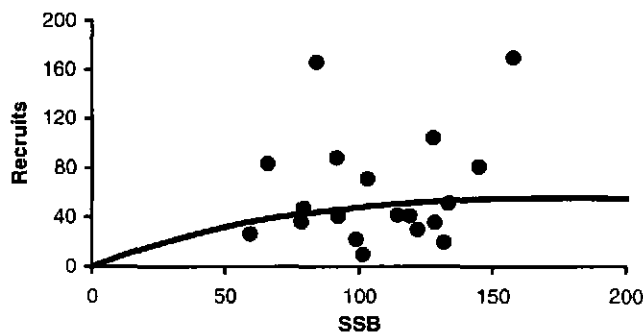
### 3.3 Icelandic Haddock

The Icelandic haddock time series extends from 1978–97. There have been some changes in selectivity and weight over time (Figure 3.3.1). With this stock, the annual variability of  $F_{msy}$  estimates was examined.



**Figure 3.3.1** Changes in weight and selectivity ( $F$  at age scaled to the mean  $F$  for ages 7 to 9) for Icelandic haddock. Lines depict 3-year moving averages.

$F_{msy}$  was computed from the age-structured method using a Ricker stock recruitment relationship (Figure 3.3.2). The Ricker function was fitted to the entire series of observations ( $n = 19$ ), with resampling, in each of 500 bootstraps.  $F_{msy}$  was computed each year using the S/R fits and resampling from the weight, selectivity and maturity vectors from the most recent 5 years. Fifteen percent of the bootstraps gave unreasonable SRR parameter estimates (negative parameters), and these were discarded from further analyses.



**Figure 3.3.2.** Stock-recruitment data for Icelandic haddock, and fitted Ricker relation.

Figure 3.3.3 depicts the variability of  $F_{msy}$  estimates over time. A slight trend would be evident from looking at the point estimates only, but examining the estimates with their associated variance suggests that there is considerably more noise than signal. Thus, for this data set, it may not be necessary to re-compute  $F_{msy}$  frequently. This conclusion cannot be carried out into the future, however, as it is quite possible that (a) the variance of the  $F_{msy}$  estimates may decrease from having more observations or from expanding the dynamic range of biomass observations, and (b) larger changes in selectivities or life history parameters could occur. The point estimates of  $F_{msy}$ ,  $B_{msy}$  (SSB) and MSY are shown in Figure 3.3.4. These suggest that  $F$  has been maintained higher than  $F_{msy}$  without causing declines in biomass (as can be explained by exceptionally large year classes in 1976, 1985 and 1990); more could be learned about the MSY-related quantities if S-R observations were available from a period where  $F < F_{msy}$ .

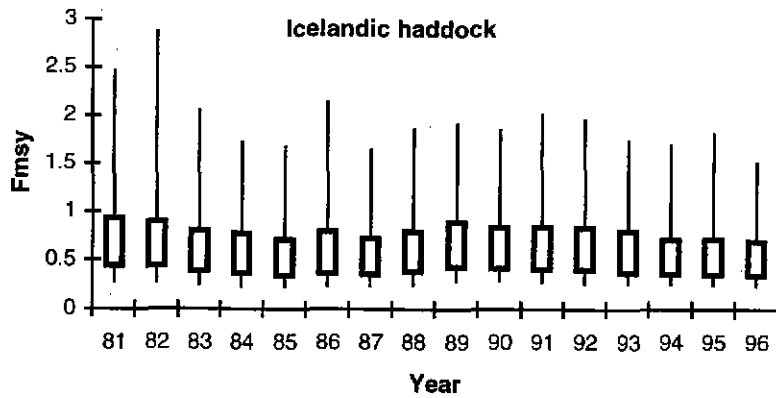


Figure 3.3.3. Annual  $F_{msy}$  estimates for Icelandic haddock. The thin lines depict the 90% range based on bootstrapping. The blocks depict the range of the central 50% of the bootstraps.

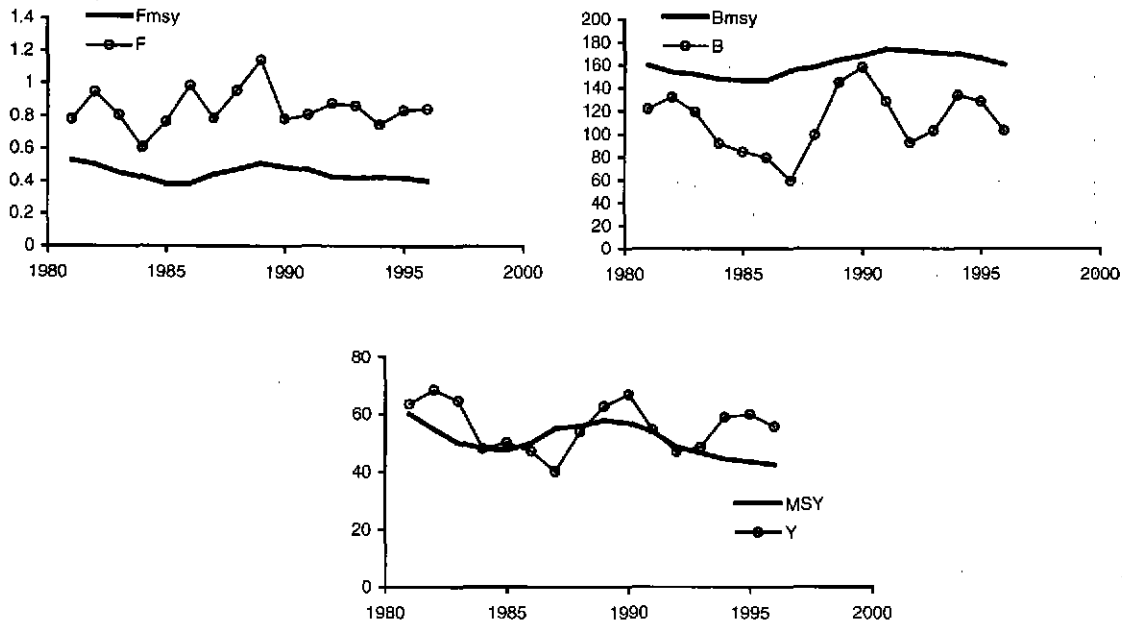


Figure 3.3.4. Point estimates of  $F_{msy}$ ,  $B_{msy}$  and MSY for Icelandic haddock, and related quantities from the assessment.

### 3.4 Conclusions

In conclusion, these analyses suggest that BRPs should be recomputed when there are “real” changes in the biological or fishery parameters. How often the quantities need to be recomputed when the underlying causes are changes in weight, fecundity and selectivity probably depends on the annual signal:noise ratio in the data. This could be investigated on a stock-by-stock basis, for example, through bootstrapping of annual estimates (see haddock example, above). When the underlying cause is more elusive, such as a systematic change in natural mortality, it may be difficult to quantify the change without considerable research resources (for example tagging experiments every few years) or before 5–10 years pass such that one could compare diagnostics in the stock assessment (e.g., residual patterns) against auxiliary data. When the stock-recruitment relationship changes, empirical evidence is even more difficult to obtain. In the cod example, it was assumed that only  $M$  changed, all of a sudden, and that the SRR was not affected. It is possible that there is a risk involved in doing this, if survival ratios actually decreased, such that the new reference points could overestimate the stock’s productivity.

## 4 BIOLOGICAL REFERENCE POINTS, HARVEST CONTROL RULES AND TECHNICAL INTERACTIONS

### 4.1 Introduction

So far Biological Reference Points (BRP) have been defined and Harvest Control Rules (HCR) evaluated mainly in a single species context. Several ToR to this group, notably f), ask for consideration of how the definition and use of BRPs and HCRs may be affected when technical interactions are taken into account. In this section we attempt to elucidate which aspects of the response require methodological developments, and which require clearer expression of objectives by managers.

In principle technical interactions refer to cases where fishing operations in a given area generate fishing mortality on groups of stocks (and ages thereof) living on the same grounds. However, many fisheries in which such interactions occur also involve a variety of vessels differing in size, gear, tactics, etc. It is thus appropriate to extend the discussion to this facet of the so-called mixed fisheries.

#### 4.1.1 On the methodological side

##### Regarding BRPs

A distinction has to be made between BRPs focused on sustainability of either reproductive potential (e.g.,  $F_{crash}$ ,  $FX\%Bv$ ) or yield (e.g.,  $F_{MSY}$ ,  $F_{0.1}$ ). The former relate to concerns that stocks may be driven to extinction by the cumulated effect of natural and fishing mortality. Whereas multispecies considerations involving biological interactions clearly change both the value of these BRPs and the perception of current state relative to these, there is apparently no reason why account of technical interactions only would cause such changes in perception. In other words, the formal definition of 'conservation' reference points and their estimates from single species assessments should remain relevant in a mixed fisheries context. Likewise,  $B_{lim}$  or  $B_{pa}$  for given exploitation pattern are unlikely to be modified in concept or value due to the inclusion of technical interactions. Where practical problems arise, this is because the various species fished by the same fleets or fleet components in a given area require different amount of change in fishing mortality for their specific target to be met. However, finding how these conflicting requirements should be reconciled does not require the definition of new BRPs; rather, it is a matter to be handled in the evaluation and implementation of the HCRs.

In contrast, reference points related to some maximisation of output from the fishery are influenced by the inclusion of technical interactions, as output is then an aggregation of contributions from the various species. Moreover, when the component species have significantly different value, it is recognised that the simple sum of yields in weight may not make much sense, and the contributions may need to be weighted by prices, at least on a relative scale. Also, whereas the usual per-recruit calculations may be appropriate to estimate yield-based reference points in a single species setting, inclusion of technical interactions requires that either the aggregation process or the stock-recruitment functions reflect the relative magnitude of recruitment of each species. Another difficulty with such 'maximisation' reference points is that they implicitly assume that maximising yield, value or profit is the objective of managers and fishers; experience indicates, however, that this is seldom the case and that managers are in fact concerned with many other criteria as well. Also note that maximising some aggregate quantity without examining the consequences for individual components may not be acceptable to some fractions of society.

It is a well known fact (although not always clearly stated in scientific advice) that both 'conservation' and 'maximisation' reference points are conditional on the underlying exploitation pattern. However, it is often overlooked that when decision rules alter the relative activity of fleet segments that have disparate exploitation patterns, this has the effect of changing the overall exploitation pattern and therefore the value of those BRPs. Thus, if the HCR aims at keeping the fishery within limit reference points but also includes a feed-back that changes the allocation of effort among fleets, the BRPs become moving targets. Likewise, if the decision rule implies changes in technical measures (e.g., mesh size, young fish boxes), it is likely to change the exploitation pattern and the value of BRPs, notably if fleet segments are not affected to the same extent.

The major problem encountered so far with assessment of and advice for mixed fisheries is that most practical methods of assessment assume a fixed linkage across species, i.e., assume that an X% change in effort translate into an X% change in the reference Fs at age of all species caught in the assemblage. Of course, this assumption is unlikely to be valid when fleets are defined as broad groups of vessels (e.g., the international fleets targeting roundfish or flatfish in the North Sea), and finer subdivisions into groups (métiers) having consistent fishing tactics on given subsets of species are required. Seasonal disaggregation also helps in this respect. But even then it is often uncertain whether, or how long, the linkage observed at some point in time will persist. Although they often use the mixed-fishery argument to blame

discarding on inconsistent regulation (namely TACs) across species, fishers would generally dispute the assertion that they have no flexibility to target individual species within an assemblage, even during a given trip. There is evidence that fishers change target and practice (e.g., spatial distribution of effort) as the relative abundance of species changes or in response to management measures (e.g., trip limits), but the nature, timing and extent of such changes is not easy to predict. To handle such cases, it is necessary to evolve from the 'metier-based' models most commonly used so far to assess technically interacting fisheries (where a metier is a given combination of gear, target species and/or area fished) to real fleet-based models in which vessels belong to only one fleet and may change metier in response to various factors. However, defining the functional form of this response and having the appropriate data at hand is far from a trivial task.

Eventually, it is appropriate to recall a well established principle, that proper assessment of technical interactions requires fishing mortality estimates based on total catches, including discards. The perception of how some fleet segments interact with others can be seriously wrong when the analysis considers landings only. This is particularly the case when fisheries are effectively sequential, with the sequential interaction taking place in young fish that are mostly discarded. Also, 'indirect' technical interactions, i.e., when some fishing practices impact species by altering their habitat, pose particular problems of observation and data.

#### Regarding HCRs

The generic forms of HCR considered by this WG seek to maintain the fishery within an acceptable domain defined in terms of spawning biomass on one axis and fishing mortality on the other, and include specified rules to adjust F depending on how far current state stands from the limits of the domain, inside or outside.

As indicated above, it is unlikely that taking only technical interactions into account modifies the SSB limits on one axis. Simply, the acceptable domain of SSB will have to be defined with consideration of all species of interest. In practice, things may not be that simple as it remains to be decided by managers whether they want the limits to be met for all species at any time (likely set by the species in the most critical state) or accept a degree of risk for some species. Note that an explicit decision on this point has never been obtained from managers so far (e.g., for cod vs. haddock or whiting in the North Sea). Nevertheless, it is the F variable in the HCR which may be most problematic in a multispecies context as it is unlikely that a given change in effort will lead F to the desired value for all species simultaneously, even if we leave aside the issues of targeting flexibility or of implementation error. It will be all the more difficult to handle F in the HCRs that the practice in ICES has been to define the F-based reference points in absolute value whereas all experiences with assessment of mixed fisheries demonstrate that using absolute Fs is unworkable in this context. Effort levels, or relative changes thereof, by fleet segment or for the overall fishery are the only common 'unit' across species for the decision variable.

In the evaluation of HCRs, the only implication of technical interactions is that the performance criteria need to be computed and presented for each species and fleet segment (where appropriate) in addition to those given for the overall fishery as it can be expected that managers want to see the consequences for each component. This may lead to large volumes of output, associated with the usual issue of summarising and displaying multi-dimensional data.

#### **4.1.2 On the management side**

Management has essentially to do with deciding upon trade-offs (e.g., short- vs. long-term, yield vs. revenue or employment, producers vs. processors, etc.), and this is all the more true when dealing with mixed fisheries as the requirements of various species and fleet segments are generally conflicting and have to be reconciled through some form of compromise.

In this context, one of the key pieces of information required from managers is not only a clear statement of objectives (this applies in any case), but also an explicit ranking of priorities regarding the various species. Scientists have often worked on the basis that the most depleted stock(s) should set the constraint for the whole fishery, but managers have seldom expressed clear views as to whether they endorse this principle or want alternative criteria to be used. In particular, if some stock is so depleted that a recovery plan is called for (e.g., West of Scotland saithe), there has been no clear policy stated regarding how other stocks fished by the same fleets in the same area should be dealt with. Likewise, there has been no agreement on the time scale and speed at which stocks outside PA limits should be returned within these. Without this information, the range of possible management options is so wide that scientists are unlikely to select for evaluation those that are effectively relevant and useful for practical decisions.

More generally, the definition and evaluation of HCRs may well remain an academic exercise if conducted by scientists in isolation. In the few instances where the concept of HCR has been pursued to the point of implementation, managers and industry representatives have been involved in the process and could directly provide feed-back and suggestions

that improved acceptability. Involvement of policy makers and stakeholders is still more critical when the management rule implies different burdens and benefits for the various fleet segments, as this has immediate bearing on the very touchy issue of wealth (re-) distribution. Such rules cannot be properly evaluated on biological bases only, nor decided by scientists only.

## 4.2 An evaluation of technical interactions in the North Sea flatfish fishery

### 4.2.1 Introduction

The importance of technical interactions in the North Sea plaice and sole fisheries were evaluated using a simulation model (Kell *et al.*, 1998). This comprised a multi-species multi-fleet operating model from which pseudo-data are sampled and then used to assess stock status (using the current single species assessment) and set catches using a simple harvest control rule to set quotas for each stock independently. See Section 6.7 for details. The intention was to demonstrate the importance of technical interactions, and the confounding between management and assessment.

### 4.2.2 Modeling of catches over quota

In the multi-species operating model it was assumed that fleets would continue fishing until both sole and plaice quotas were both exhausted. Catches over the quota would therefore be taken for the species whose quota had been met first, however, these catches went unreported and are not used by the working group in their assessment of stock status.

### 4.2.3 Experimental design

The simulation experiment (summarised in Table 4.2.3.1) was conducted with treatments corresponding to fishery, assessment method and harvest control rule. An individual treatment within an experiment is a particular choice of operating model and management procedure.

The fisheries were either single species plaice, single species sole or multi species plaice and sole to allow the effect of technical interactions to be investigated. The historical assessment was either conducted using XSA as in the working group (WG) or by simply copying the true stock data (Perfect). The assessment methodology was the same as that used by the ICES Working Group on the Assessment of Demersal Stocks in the North Sea and Skagerrak (ICES, 1999a) based on XSA, the PA Software and a medium term projection. The control rules are given in Section 6.7.5.

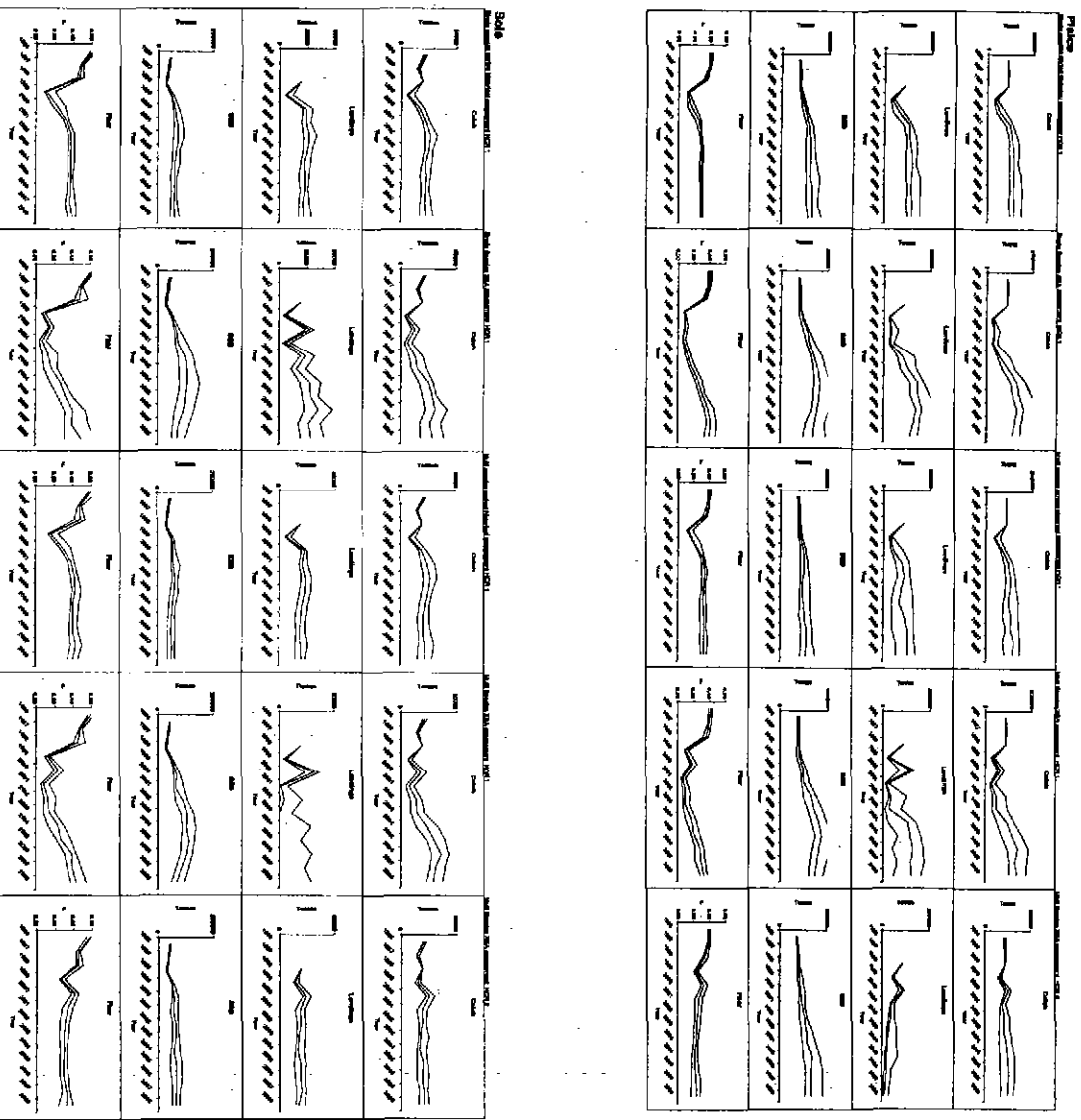
**Table 4.2.3.1** Experimental design

Experiment	Fishery	Assessment	Control Rule
1	Plaice	Perfect	1
2	Sole	Perfect	1
3	Plaice and Sole	Perfect	1
4	Plaice	WG	1
5	Sole	WG	1
6	Plaice and Sole	WG	1
7	Plaice and Sole	WG	2

### 4.2.4 Results

The results are summarised in Figure 4.2.4.1, the 25th, 50th and 70th percentiles of catch (reported and unreported), landings, SSB and Fbar are plotted for each experiments. Catches, SSB and Fbar are from the true population not as perceived by the working group, landings are those reported to the working group. Landings may differ from the true catches especially in the multi-species experiments.

Figure 4.2.4.1 Trajectories of catch, landings, SSB and Fbar in the future for the simulation experiments





The single species experiments with perfect assessment are essentially an improvement on the current historical assessment and projection used by the working group. As the combined Monte Carlo simulation and bootstrap of XSA provides better estimates of uncertainty in stock status and carries these through into the calculation of reference points and the projection. The projection also allows for multiple fleets with different selection patterns. For both plaice and sole fishing mortality is initially reduced before climbing back to the target level. SSB, catches and landings recover relatively quickly.

When the current population status is assessed using XSA the mean fishing mortality in the true population is reduced for longer resulting in lower catches in the short term and higher SSB in the long term compared to when the working group has perfect knowledge of stock status. This is because the working group underestimates  $N$  and over estimates  $F$ , probably due to the assessments inability to track fast changes in the stock. There is also greater inter-annual variation due to the lag between assessment and management.

In the multi-species experiments with perfect assessment yields and SSBs are lower and are more variable, although  $F$  is at the same expected level  $F$  it also is more variable than in the single species case.

Once historical stock status is assessed using VPA, however, the occurrence of mis-reporting in total catch causes the assessment to quickly go wrong. This is due to the feedback between the bias in the assessment and misreporting. This is exacerbated by the form of harvest control rule 1. Therefore harvest control rule 2 implemented a simple form of effort control, this is also more realistic as changes in fishing mortality are likely to be more gradual than dictated by harvest control rule 1. Therefore harvest control rule 2 set the quota by setting catches corresponding to the average of the target and last year's  $F$ . The result is that in changes in  $F$  and SSB were more gradual and the ability of the working group to assess the stock improved.

Although the scenarios examined were relatively simple they illustrate the importance of technical interactions and the link between management and our ability to assess the stock. More work needs to be done on modelling the response of fleet dynamics and hence the response of fishing mortality to management in mixed fisheries.

More work needs to be done on evaluating the response of fleets to harvest control rules in mixed fisheries.

## **5 MULTISPECIES BIOLOGICAL INTERACTIONS AND REFERENCE POINTS**

### **5.1 Introduction**

There has been increasing interest to taking an 'ecosystem' approach to resource assessment and management instead of a single species, or collection of single species assessments. The underlying question to be addressed is: "What are the consequences to biological reference points (or harvest control rules) in assuming single-species when the resource is part of a multi-species system?". The WG had a number of sources available. The first was the report of the Multispecies Assessment Working Group (MAWG, ICES 1997a) which devoted a section to the multispecies considerations in the development and provision of precautionary advice. Secondly, papers by Gislason (1999) and (WP) were presented at the WG meeting. The former is a 3 species model of the Baltic Sea while the latter is a system of linked surplus production models.

Multispecies models (Bormicon and Flexibest) were also presented and are reviewed in Section 12.

The objectives to be completed during the meeting are a review of the available papers, development of recommendations for multispecies models, and some extensions to existing models.

It must be recognised that the modelling and resultant conclusions done during the COMFIE WG are preliminary in nature. Although care was taken, time was not available for detailed model testing and review.

### **5.2 Review of Multispecies Assessment Working Group Report and related work**

The Multispecies Assessment Working Group (MAWG) (ICES 1997a) investigated the impact of predator-prey interactions on reference points using two conceptually different approaches. The first approach was based on single species and multispecies VPA assessment models. An analytic comparison of reference point estimates obtained from VPA assessments of a single species model assuming cannibalism or constant predation and a single species model ignoring those effects was carried out. The investigations were then extended to more complex systems using a simulation based comparison. The second approach was an analytic investigation of the interaction of fishing mortalities in a simple predator-prey system using a Lotka-Volterra model formulation which is not based on

equilibrium considerations but models the dynamic relationship. A number of traditional biological reference points are not well defined in the context of interacting species, for example, reference points derived from observed R/SSB relationships ( $F_{low}$ ,  $F_{med}$ ,  $F_{high}$ ,  $F_{loss}$  and  $B_{90\% R 90\%Surv}$ ) as they depend on historical predator and prey densities. Hence comparisons were restricted to  $F_{MSY}$ ,  $F_{0.1}$ ,  $F_{crash}$  and  $B_{MSY}$ .

### 5.2.1 MAWG results

The MAWG considered two simple cases in which the natural mortality of a single species is increased. In a single species system with cannibalism on the pre-recruit young by adults, a single species VPA will underestimate recruitment but estimates of  $F_{MSY}$ ,  $F_{crash}$  and  $B_{MSY}$  will not be affected. However, if instead of cannibalism there is assumed to be constant predation by some external predator on all age classes (and natural mortality is assumed constant) fishing mortality will be overestimated by the single species VPA. As a consequence single species VPA estimates of  $F_{MSY}$  will be lower and estimates of  $F_{crash}$  and  $B_{MSY}$  will be higher compared to estimates based on a multispecies VPA. The working group also presented some generalised results obtained for the two single species cases. However those results appear to be contradicting the above conclusions.

Using an age structured simulation model the MAWG also looked at the impact of multispecies interactions on rebuilding strategies, that is on the time of it would take to rebuild the stock. It was found that if multispecies aspects are taken into account the time to rebuild a depleted population of one of the species is much longer than when estimated from a single species model.

A more complex predator-prey relationship was analysed by simulation for which a 2 and 3 species model were used based on a system of cod. The models included Ricker stock recruitment relationships. The more complex model had cod preying on both sprat and herring and on their young and resembles the system found in the Baltic Sea. Fishing was assumed to be directed independently at cod and jointly at sprat and herring. Gislason (1999) extended the 3 species model to include a feedback of prey abundance on predator weight increase and of weight at age on predator maturity which will be described below.

The second approach taken by the MAWG was an analytic analysis of a Lotka-Volterra predator-prey model. The Lotka-Volterra model is a production model for prey-predator systems in which predator population growth depends on prey numbers and prey mortality is determined by predator numbers. Theoretical considerations based on this model lead to estimates of the range of fishing mortality rates for both predator and prey species under which neither the prey nor the predator population will collapse (in a deterministic model) or for which the maximum yield will be achieved in an equilibrium state.

The MAWG concluded from the work presented that *M2 can have a great effect on at least some of the reference points explored by COMFIE. The timing of predation relative to the timing of fishing (...) is an important consideration, as is the size of F. If F is "large", M2 has relatively little effect on yield or recruitment, and hence on the estimated reference points. When F is as low as 0.2, though, values of M2 as low as 0.2 can lead to major inaccuracies if single species approaches are used to estimate recruits per spawner, yield per recruit, or reference points derived using those population attributes.*

### 5.2.2 Review of Gislason (1999)

Gislason (1999) compared point estimates of reference points based on single species VPA assessments with those obtained from traditional MSVPA and an extended MSVPA. In the extended MSVPA model a feedback of prey abundance on predator weight increase and of weight at age on predator maturity is added. Recruitment was a linear function of spawning stock in both multispecies models. As a general conclusion Gislason noted that the concept of maximum sustainable yield in terms of weight might make little sense for a system of interacting species if the species differ substantially in market value and some species are more productive than others (in particular lower valued species). In this case, economic considerations should be taken into account for defining fishing mortalities leading to maximum yield in value or net revenue (taking account of the cost of the fishing operation).

Reference points estimates for  $F_{0.1}$  and  $F_{MSY}$  obtained with three models, single species VPA, traditional MSVPA and extended MSVPA differed quite substantially with an increase in value as model complexity increased. Spawning stock reference points  $B_{50}$  and  $B_{pa}$  were also affected by the kind of model used. For the simple case of one predator with two prey species considered by Gislason, increasing the connectivity of the species (from single species VPA to extended MSVPA) led to an increase in the number of fishing mortality rate combinations for which the predicted spawning stock biomass was simultaneously above the biomass reference points  $B_{50}$  and  $B_{pa}$  for all three species. Thus it seems that for the simulations carried out by Gislason, reference points calculated from single species VPA are conservative estimates.

Gislason concluded that *reference limits for forage fish cannot be defined without considering changes in biomass of their natural predators. Likewise, reference limits for their predators cannot be defined without considering changes in the biomass of their prey.*

## Comments

It seems, in the context of this model, that the more connections between species the more robust the system becomes to fishing in an equilibrium state but how useful is this for defining harvesting strategies which should lead towards those equilibrium states when there are dynamic reactions on the way? Hence reference points based on equilibrium models might not be that useful to guide the selection of suitable fishing mortality rates. In any case, harvesting rules for predators and their prey should consider the interactions between the two and the current status of both groups. The Lotka-Volterra modelling approach allows to take the dynamic interaction into account to some extent but has some strong assumptions. In the Lotka-Volterra model having no age structure in either the model or the fishery implies equal catchability for all ages as well as equal proportional vulnerability of animals in different age-length groups. The analytic of the Lotka-Volterra model presented by the MAWG could be useful for simple systems but for more complex systems the theoretical approach becomes complicated and requires further investigations. The question whether the deterministic approach should be abandoned in favour of a stochastic approach also deserves attention.

A number of questions remain unanswered by the investigations carried out by both the MAWG and Gislason. To put things into perspective it would be important to know what are the relative importance of unaccounted discard mortality and predation mortality. Is there a connection between discarding rates and trophic position or between trophic position and robustness of estimated reference points. How the complexity of the food web, the heterogeneity (number of dominant species) of the food web and estimated reference points are linked should be investigated. A very important question is what should be done if we know that species interactions have an important impact on reference point estimates. It should be considered whether reference points are appropriate for a multispecies system or whether they should be extended to reference lines or reference surfaces.

Because multispecies FMSY was almost twice as large as the single species estimate, Gislason concluded that it may be a dangerous reference point. Further he stressed the importance of investigating structural uncertainty which he defined as the uncertainty due to model formulation.

### 5.2.3 Random Generation of Deterministic Multi-Species Models

Preliminary work has been conducted toward development of a framework to facilitate random generation of deterministic, multi-species models. To keep the whole approach reasonably simple, the model used in this development is of the biomass dynamic (surplus production) variety. Specifically, a multi-species, multi-fishery "Schaefer" model is used (where a "fishery" is defined in terms of its target species). The basic idea is to specify the dimensions of the model (number of species and number of target fisheries) and the parameters of a set of statistical distributions, from which model parameters are drawn randomly. These model parameters are then used to form connections between the various species-species and species-fishery pairs in the system. To assure that the models thus generated are reasonable and tractable, the following constraints are imposed:

- All species must have positive levels of abundance in the unfished equilibrium state.
- The unfished equilibrium state must be stable.
- A unique solution for maximum sustainable yield must exist (see below).

To date, this framework has been used to address the relationship between ecosystem and single-species approaches to the concept of maximum sustainable yield (MSY). To this end, an "ecosystem MSY" was defined as the maximum yield that can be obtained in equilibrium from all target species combined (including bycatches of species which are targeted by other fisheries), subject to the constraint that all species retain positive levels of abundance in the MSY equilibrium state and that all fishing mortality rates associated with MSY remain non-negative. A "single-species MSY" was defined similarly, except that the yield was maximised with respect to a single species at a time rather than for all species combined. When computing MSY for a single target species, it is also necessary to adopt some sort of convention for dealing with the fishing mortality rates of the other target species in the system. So far, it has been assumed that each of the "other" target species are harvested at their respective "ecosystem MSY" rates. In other words, the fishing mortality rate for each target species is individually adjusted so as to maximise the equilibrium yield from that species, conditional on the assumption that all other target species are harvested at their respective ecosystem MSY rates. In computing both "ecosystem" and "single-species" MSYs, it has been assumed that the true values of all parameters are known.

Computer memory limitations have so far hindered analysis of systems consisting of more than a few (five or so) species. Typically, 100 randomly constructed systems are generated for a given set of distributional parameters. Although only a few sets of such randomly constructed systems have been analysed, the patterns that have emerged suggest that further investigation is warranted. Some of the patterns observed so far include the following:

- The ecosystem MSY is always less than the sum of the single-species MSYs, although the difference may be very small.
- The fishing mortality rate associated with a particular single-species MSY is often close to the ecosystem MSY rate for that species, but may also be much higher or much lower.
- Fishing all target species at their respective single-species MSY rates can result in one or more species going extinct in equilibrium, even though each single-species MSY (taken one at a time) is constrained so as to prevent equilibrium extinction of any other species.

In presenting the above preliminary conclusions, it should be emphasised that the analyses undertaken to date involve a number of strong assumptions, including:

- Ecosystem dynamics are assumed to be deterministic (i.e., no process error).
- The state of the system is assumed to be known (i.e., no measurement error).
- The values of all parameters are assumed to be known (i.e., no estimation error).

An attempt was made during this meeting of the ComFiE WG to relax some of these assumptions. The results of this attempt are described in Section 5.3.1.

#### 5.2.4 Depensatory stock-recruit and seal-cod affects on biological reference points

A brief presentation was made comparing single species Shepherd-Sissenwine plots with two species analysis. The first case used simulated data fit to a Ricker curve and compared that to a depensatory curve (Peterman, 1977). A potential cause of depensation is predation mortality. The steeper right-hand of the depensatory curve was reflected in an approximate halving of  $F_{crash}$ , although the MSY related reference points were unaffected. See Figure 5.2.4.1. Over the range, or indeed with data from most fisheries, of data it would be impossible to discriminate between the two stock-recruit relationships.

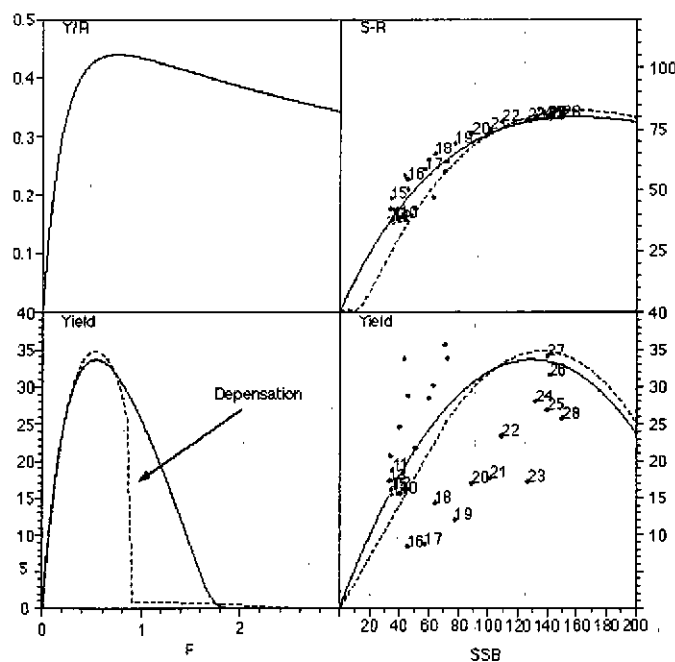


Figure 5.2.4.1 Comparison of Ricker and depensatory stock-recruit equilibrium yields.

In the second case (Figure 5.2.4.2) uses data from Eastern Scotian Shelf cod. Two assessments were performed; the first was a traditional single species VPA and the second had seal predation is modelled. Natural mortality is modelled as an age dependent constant and a term based on estimated consumption by grey seals in which the rate is proportional to cod abundance. Seal predation is modelled as an age dependent process with only cod less than age 8 being consumed and ages 3 and under as being the most suitable. The two species natural mortality is time dependent as the grey seals have experienced a population growth of approximately 12% per year. Although the higher natural mortality approximately doubles the average recruitment, the surplus production is much lower as is seen by comparing the lower panels of Figures 5.2.4.2 and 5.2.4.3. All references or limits are affected by the inclusion of seal predation suggesting that the previous harvest levels will not be attained with current levels of seal predation.

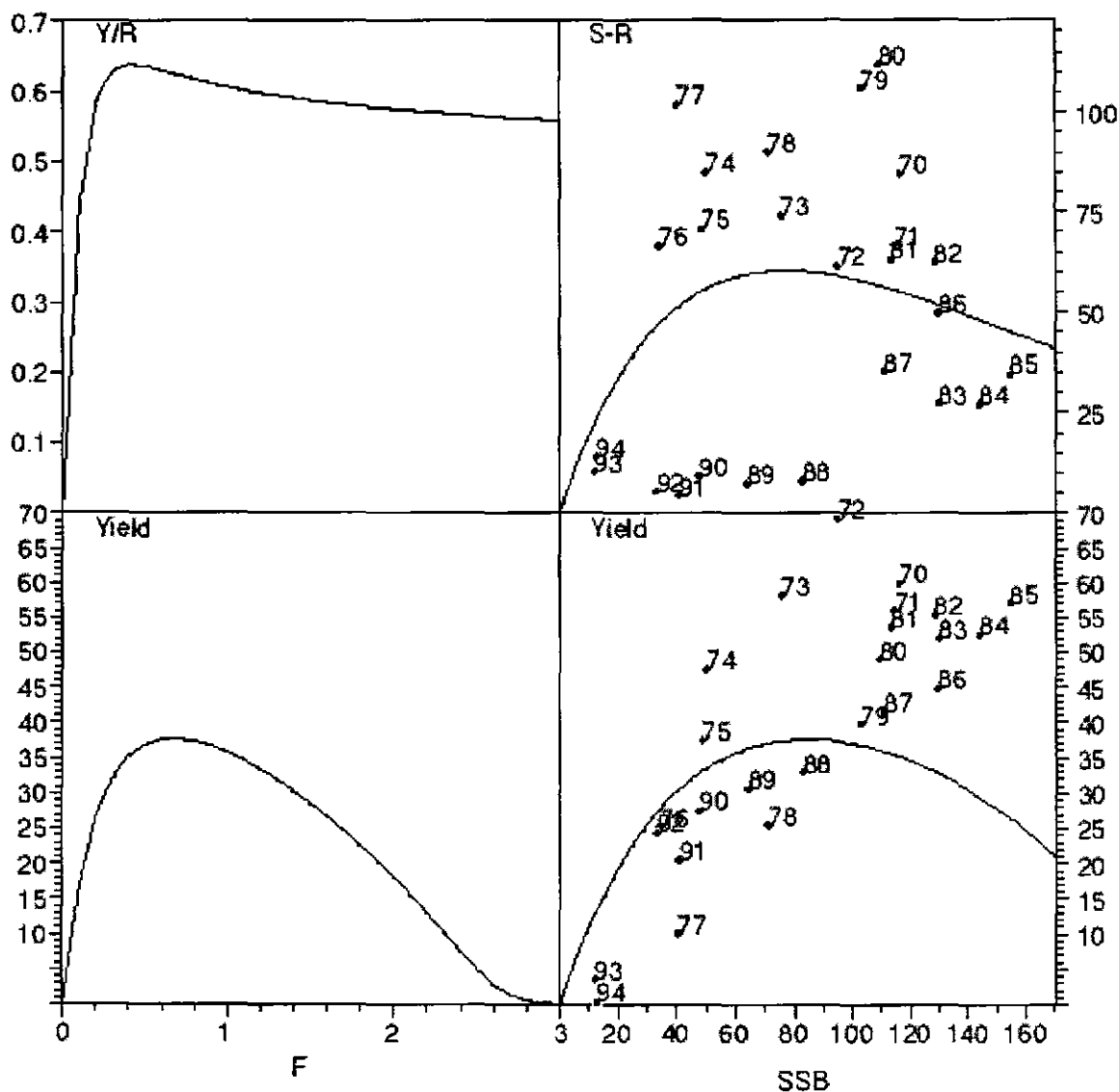


Figure 5.2.4.2. Equilibrium analysis of Eastern Scotian Shelf cod with  $M = 0.2$

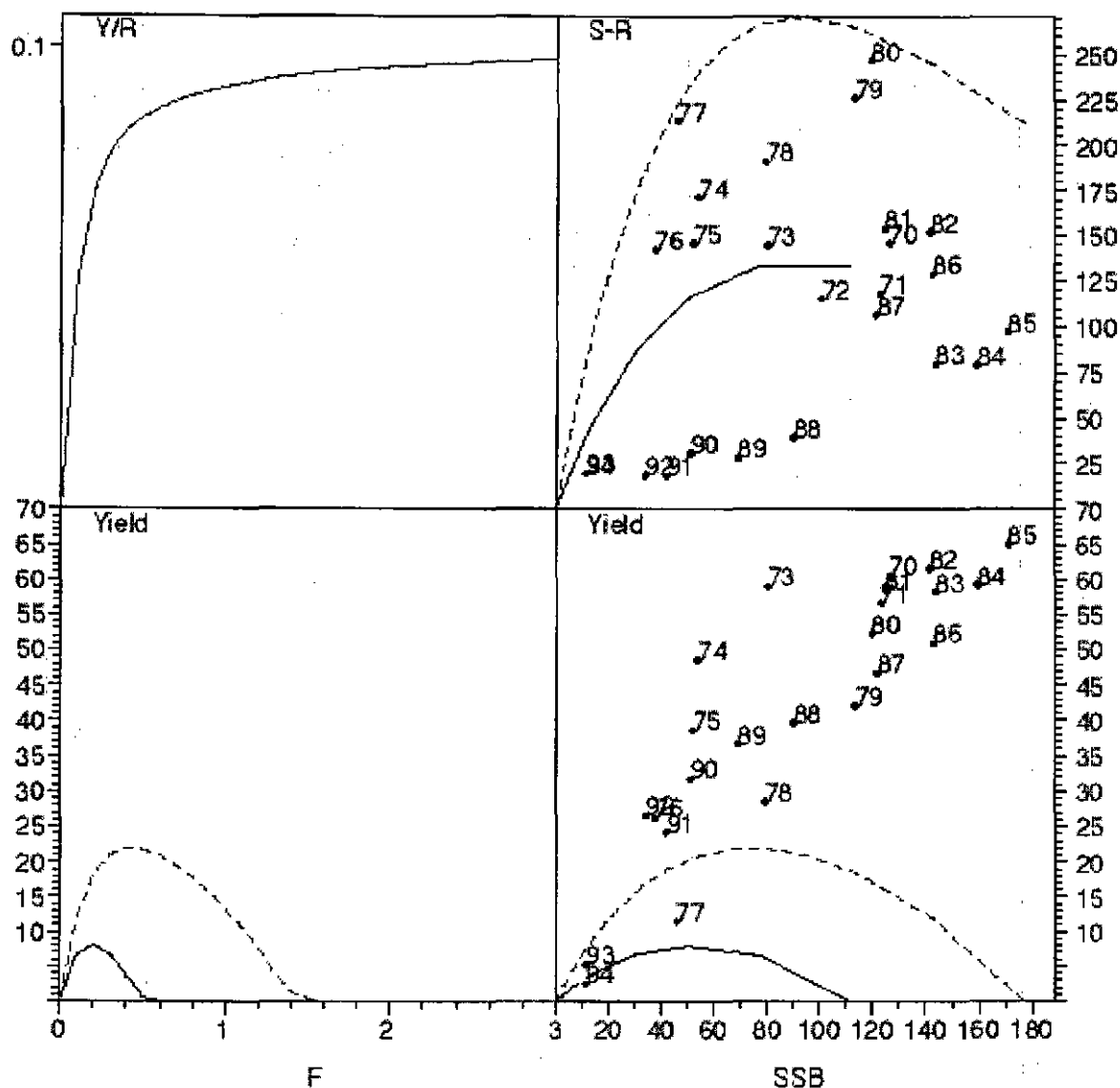


Figure 5.2.4.3. Equilibrium analysis of Eastern Scotian Shelf cod with  $M = 0.2$  and compensated seal mortality. Solid line is fit  $\tau =$  through stock-recruit data, dashed line is twice that among of recruitment per SSB

### 5.3 Further developments during meeting

Given the time available for model development and testing, limited objectives were defined. The first objective was to try and define a minimally complex model that would be sufficient to assess a wide range of multispecies effects, especially in contrast to single species analysis. Also, age structured and production simulations were initiated.

#### 5.3.1 Incorporating Process, Measurement, and Estimation Error into Multi-Species Models

During this meeting of the ComFiE WG, an attempt was made to relax some of the assumptions employed in the multi-species, multi-fishery Schaefer model described in Section 5.2.3 (where a "fishery" is defined in terms of its target species). This was done by assuming a particular set of parameter values for a three-species, two-fishery model and projecting the system forward in time for 20 years, revising the model of Section 5.2.3 as follows:

- Process error was incorporated by adding a random deviate drawn from a normal distribution to each species' dynamics.
- Measurement error was incorporated by simulating survey and catch monitoring programs, where survey abundance and catch were both taken to be normally distributed about the true relative size of each species.
- Estimation error was incorporated by using an extended Kalman filter (Harvey, 1990) to define a likelihood, which was then maximised to obtain estimates of the parameters.

More specifically, the particular model that was examined during the meeting consisted of a cyclical food chain, where Species 1 consumed Species 2, Species 2 consumed Species 3, and Species 3 consumed Species 1. In terms of technical interactions, it was assumed that Species 1 and 2 were the targets of two separate fisheries, but that both fisheries took bycatches of all non-target species (i.e., Fishery 1 targeted on Species 1 but took bycatches of Species 2 and 3, while Fishery 2 targeted on Species 2 but took bycatches of Species 1 and 3).

Two sets of parameter estimates were compared with the true parameter values: The first set was obtained by using the simulated data for all species and fisheries to estimate the full set of 24 parameters for the three-species, two-fishery model (the "multi-species assessment"). The second set was obtained by using the simulated data for one target species at a time to estimate only those 5 parameters necessary to define a one-species, one-fishery model (the "single-species assessments").

The first step in comparing the two sets of parameter estimates and the set of true parameter values was to compute the MSY fishing mortality rates corresponding to each. In the cases of the true parameter values and the parameter estimates obtained from the multi-species assessment, the "ecosystem MSY" rates were used (see Section 5.2.3); in the case of the parameter estimates obtained from the single-species assessments, the MSY fishing mortality rate was computed without considering interactions with other species. To simplify computation, in all cases the MSY fishing mortality rates were computed as though the system behaved deterministically. Given that the "real" system is known to be stochastic, this method of computation introduces an inconsistency into the logic of the analysis, but one which is fairly common in stock assessment practice.

The second step in comparing the three sets of parameter values was to apply their associated MSY fishing mortality rates to a 100-year stochastic projection. To simplify computation, it was assumed that measurement error was negligible, so that the "right" catch could be taken each year. Again, given that the "real" system is known to be measured with error, this method introduces an inconsistency into the logic of the analysis, but time was insufficient to permit a more sophisticated treatment. In each case (true parameter values, multi-species estimates, and single-species estimates), it was assumed that the system was initially in equilibrium under the MSY fishing mortality rates associated with that case. The average catch associated with each case was then calculated.

The results of this analysis are shown in Table 5.3.1.1 and Figure 5.3.1.1.

Parameter Values	Average Yield	$F_{MSY}$ (Spe. 1)	$F_{MSY}$ (Spe. 2)
True	0.896	0	0.4
Multi-Species Estimates	0.798	0.107	0.269
Single-Species Estimates	0.621	0.206	0.251

Table 5.3.1.1—Average yield and MSY fishing mortality rates associated with three sets of parameter values in a stochastic projection.

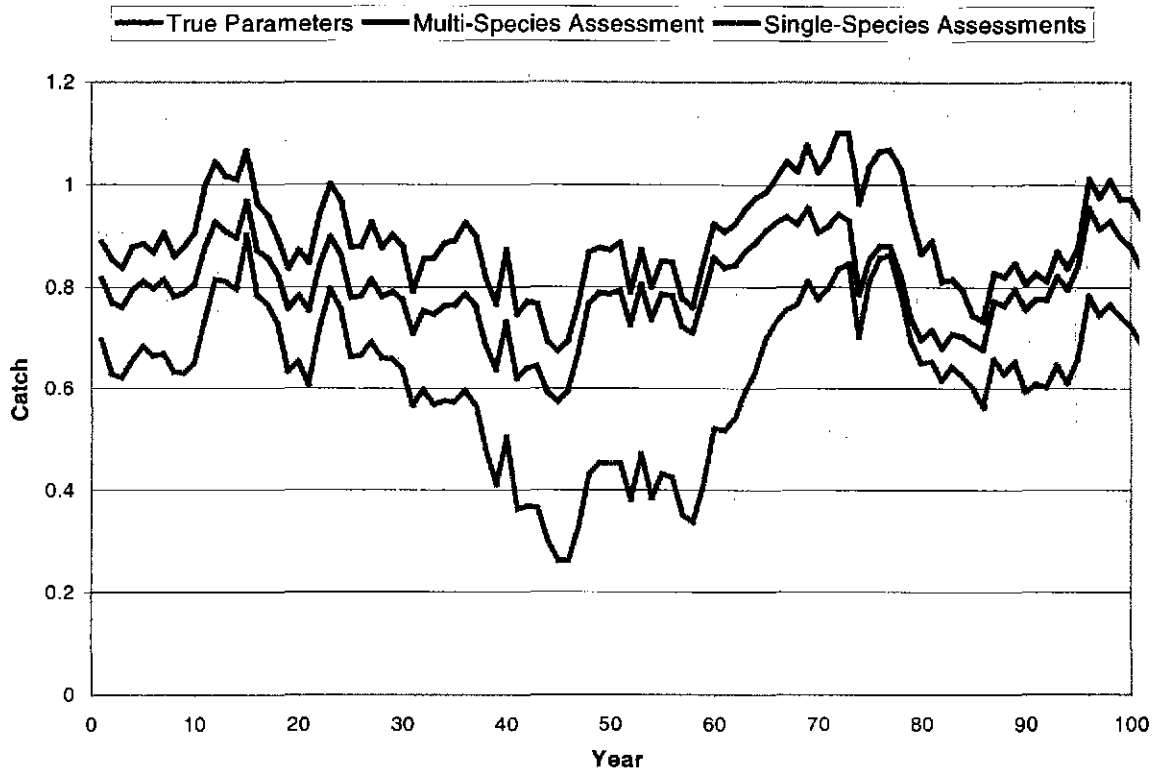


Figure 5.3.1.1—Trajectories of catch during a 100-year projection under MSY fishing mortality rates associated with three sets of parameter values: the true parameter values (upper line), the parameter estimates obtained from the multi-species assessment (middle line), and the parameter estimates obtained from the single-species assessments (lower line).

The above results indicate that neither the multi-species nor the single-species assessments yielded MSY fishing mortality rates that were particularly close to the MSY fishing mortality rates associated with the true parameter values. In the case of Species 1, for example, the MSY fishing mortality rate associated with the true parameter values was zero, whereas the multi-species assessment yielded a value of about 0.1 and the single-species assessment yielded a value of about 0.2. The actual yields generated by the various sets of parameter values were not quite as different, however. The average yield obtained under the multi-species assessment was only about 11% below the average yield obtained under the true parameter values. The average yield obtained under the single-species assessments fared somewhat worse, coming in about 31% below the average yield obtained under the true parameter values. Because time during the meeting was insufficient to explore more than this single example, however, it is currently impossible to determine the extent to which the above results might be typical. For future research, the following recommendations are offered for the purpose of evaluating the robustness of the results obtained during this meeting:

- Increase the number of systems analysed, and the number of realisations of each system.
- Increase the number of species in the system.
- Examine assessment models of intermediate complexity.



- Expand the incorporation of process and measurement error beyond the dynamics and assessment of the system to encompass the evaluation of harvest recommendations as well.
- Incorporate implementation error into the evaluation of harvest recommendations.

### 5.3.2 Simulations with 3-Species Model

An earlier version of a 3 species model, which is based on the Baltic Sea model of Gislason, Sparholt and co-workers, was developed at MAWG in 1997. This model is written in Excel and was updated at this meeting. It was not possible to include all the functionality that the minimum complex model (see Section 5.4.1) requires. Nor was it possible to fully test the model during the meeting, so any results must be considered preliminary.

The modified Gislason model presented here is a forward projection, age based assessment in which three trophic levels exist with variable interaction between the levels. An apex predator feeds on both lower trophic levels and is cannibalistic on its own juveniles, while the secondary predator feeds exclusively on the primary prey species. Input data for the stocks (initial stock size and age based parameters) are based on real stocks, although the interactions are not parameterised using any known biological relationships. Recruitment in all three species is modelled using Ricker functions.

The simulations were run using four types of interactions. The most basic level switched off the feeding and growth terms thus removing any interaction between species and was therefore essentially three separate single species projections. Secondly, “top down” control was added such that feeding activities of the predators depletes the prey numbers. Thirdly the “top down” control was removed and “bottom up” control activated, in which the abundance of the prey species impacted the predator’s recruitment. This was achieved by assessing the food intake per year against a global mean intake (estimated prior to the simulations), and penalising recruitment when the intake was below the mean. Finally, both “top down” and “bottom up” controls were activated for a “full interaction” run.

Only the apex predator will be considered here. A range of 9 fishing mortalities were inflicted on the apex predator (0, 0.2,.....1.4, 1.6) while the fishing mortalities on the other two species were held constant at 0.4. Natural mortality was divided into two sections, predation based which remained constant, and non-predation based which varied according to the level of interaction inflicted on the system. In order to obtain point estimates for reference points ( $F_{crash}$ ,  $F_{msy}$  and  $B_{msy}$ ) the runs were entirely deterministic.

### Results

Figure 5.3.2.1 shows the four time series projections of SSB for the apex predator at a fishing mortality of  $F = 0.4$ . The terminal SSB is approximately 200,000 tonnes lower for the full interaction model compared to the single species (no interaction) model. The use of the top down control makes little difference to the projection. The use of the stronger “bottom up” control is seen to destabilise the time series at  $F = 0.4$ , and this occurs over a wide variety of fishing mortalities. The full interaction run, combining these two forms of control, restores some stability, higher fishing mortalities are stabilised more quickly. Relative stability is, however, achieved by the end of the projection period and thus the system equilibrium is defined as the mean of the final 5 years of the projection.

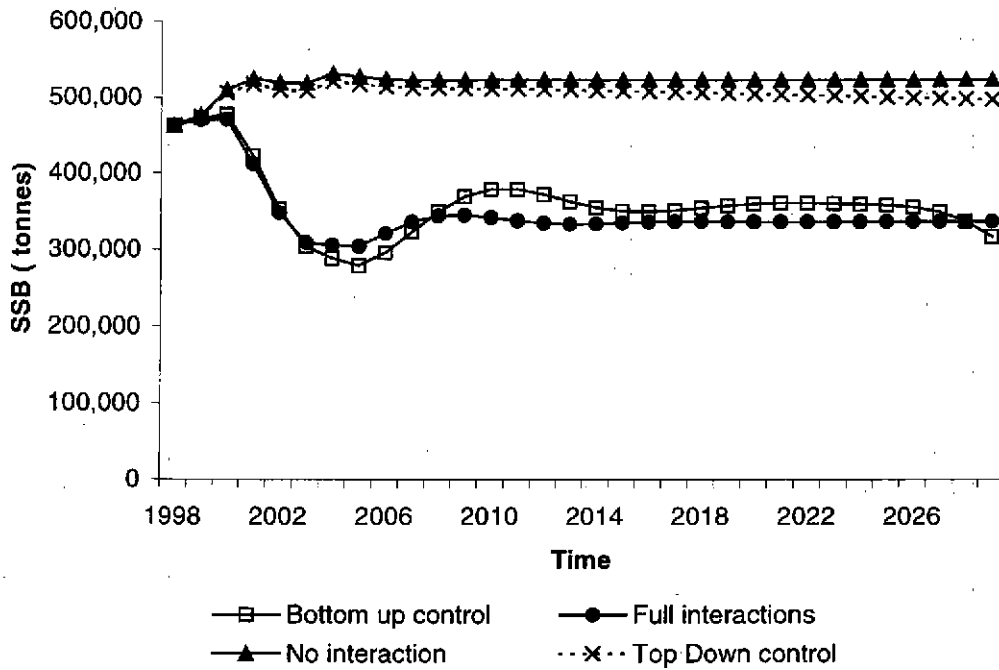


Figure 5.3.2.1 Time series projections (32 years) of SSB for  $F = 0.4$  (all 3 species) over the four interaction types.

Figures 5.3.2.2 and 5.3.2.3 show the effects of interaction type on two likely reference points,  $F_{msy}$  and  $B_{msy}$  and give an indication of effects on  $F_{crash}$ . The exclusion of multispecies interactions clearly overestimates  $B_{msy}$ . A precise value for  $F_{msy}$  is not available due to the coarse scale used, although  $F_{msy}$  appears relatively unaffected by interaction type at about 0.4.

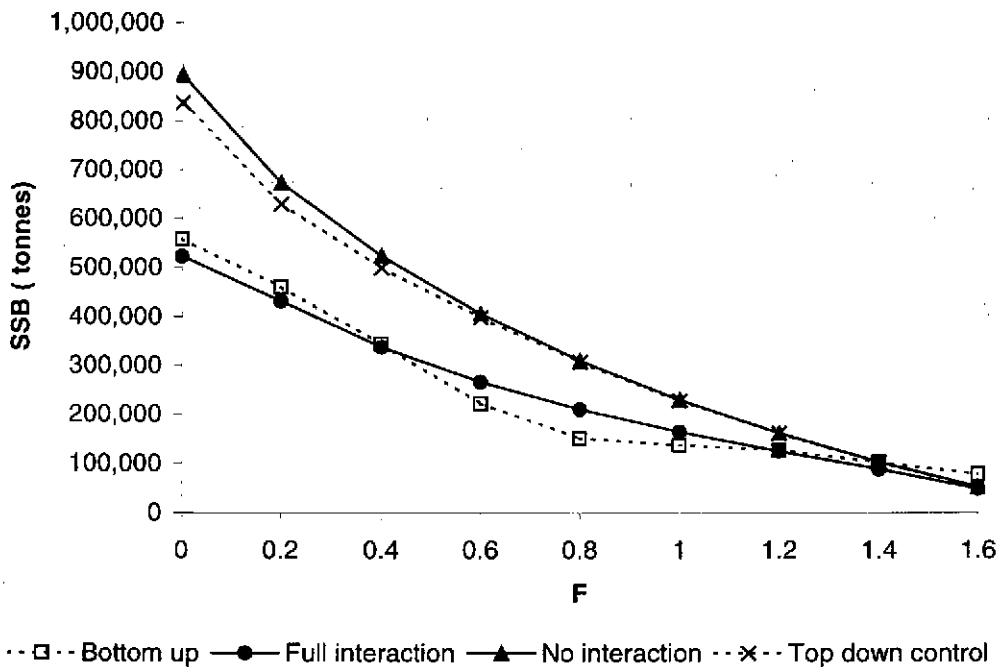
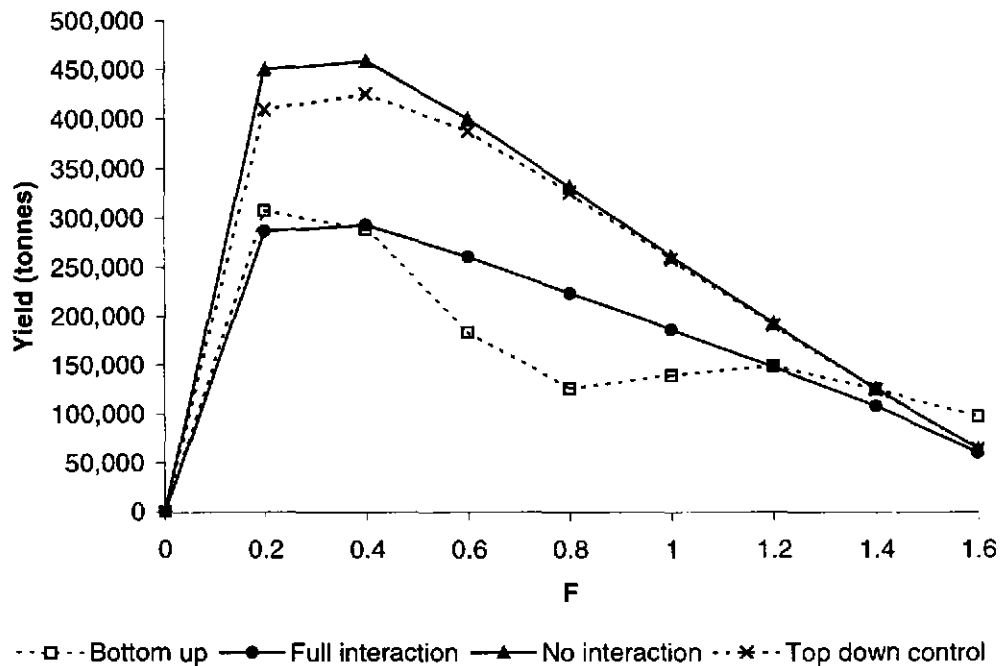


Figure 5.3.3.2. SSB of apex predator as a function of  $F$ , given  $F=0.4$  for both prey species over four different interaction types. SSB is taken as the average of the final 5 years of the projection.



**Figure 5.3.3.3.** Yield of apex predator as a function of  $F$ , given  $F = 0.4$  for both prey species over four different interaction types. Yield is taken as the average of the final 5 years of the projection.

Projection of the trajectories by eye  $F_{crash}$  gives values of approximately 1.8 for the single species and top down interactions, but actually increases when the bottom up and full interactions are considered. As fishing mortality increases on the apex predator, the two prey species increase in density, thus the cannibalism rate is reduced for the apex predator and the total mortality rate is transformed at high fishing mortalities.

## Discussion

The Figure presented here only show a single dimension transect through the potential 3 dimensions of fishing mortality and the full range of prey fishing mortalities should be explored. Higher fishing mortalities on the prey species have the potential, under full interaction, to further reduce the productivity of the apex predator due to increased cannibalism and reduced nutrition (affecting the apex recruitment).

Model complexity affects estimates of biological reference points in terms of accuracy and temporal stability, thus system parameterisation should be carefully considered.

## 5.4 Conclusion

### 5.4.1 Recommendations/considerations for modeling

The following recommendations are considered to be a minimum for a model which contains sufficient complexity to capture a wide range of ecosystem effects. Simpler models, of course, could be used to examine specific effects, e.g., 2 species interactions, deterministic effects, etc. See Section 12 for additional discussion on this topic regarding existing models.

- i) Minimum of 3 interacting populations.

Although 2 species predator-prey is beyond the scope of most current assessments, its behaviour has been well studied. Earlier theoretical work (May (1974), Levins (1966)) showed that the dynamics of a system become more complex for 3 or more interacting species.

- ii) Bottom up control

MSVPA and related models have only top down control, via predation mortality. Lower trophic levels could influence higher levels if ration influenced either survivorship or reproduction success. The Gislason paper (Gislason, 1999) included such a mechanism. Similarly, but with an opposite sign, bottom up control was incorporated by Sparholt (1996) in which prey species consumed the eggs/larvae of the predator. Density dependence across species in a shared environment may also require consideration. An example of the importance of "bottom up" control is the present high sprat biomass in the Baltic Sea, which may be large enough to inhibit cod recovery.

iii) Age structure

For many resource management questions age, or at least stage, structure is required. Concepts of recruitment, spawning stock, fishable stock and proportions susceptible to predations (suitabilities) require at least stage structure.

iv) Capability for more complex stock recruitment functions

The standard Ricker and Beverton-Holt recruitment functions could be extended to include density dependent phenomena such as depensation or refugia.

v) stochastic effects

Both stochastic parameters and stochastic process error should be included to investigate robustness.

vi) Other extensions

Spatial heterogeneity greatly increases the complexity of the model and is probably not needed in the description of a minimally complex model. Similarly, individual growth in which the population elements are functions of age, time and size also adds another dimension into the model.

#### 5.4.2 General conclusion/recommendations

The multi-species analyses undertaken during this meeting describe approaches which are in some ways complementary. However, precisely because different approaches have been taken here, it is not always clear how the results can be synthesised. It is important to remember that there are special problems involved in comparing multispecies models. This is partly because of the relatively large number of parameters involved, several of which may be quite sensitive. Although the work done during the meeting was similar to previous modelling exercises it would take considerable time to compare their results. Such comparisons would be important in the identification of critical elements, however. When the models are less similar, say the age aggregated biomass dynamic model and the age disaggregated 3 species model comparisons are more difficult. There currently exist models of Baltic, Barents and Icelandic Seas with 3 species versions. Although they treat environmental data differently they could be used in tests to compare model similarities and critical structural differences.

The WG considers it important to incorporate multi-species dynamics in the evaluation of fishery systems. The work in Section 5.3.1 is an example of preliminary steps toward this goal. Due to the early stage of development of multi-species models, this group finds it necessary to explore multi-species modelling. It would be desirable to increase the communication among the various multi-species modelling groups. Perhaps this could be accomplished by exchanging a common model description analogous to the exchange of fishery data to compare VPA programs. Alternatively, a meeting might expedite the exchange. Once the confidence in the various models was established the generation of multi-species harvest targets could proceed quickly. Several important, related questions were identified that should be considered by MAWG, COMFIE or a joint WG. First, in a fishery, what are the relative magnitudes of predation mortality, fishing mortality, environmental forcing and the uncertainty in reproduction dynamics? If multi-species effects are suspected to be significant, the practical question then arises of how to quantify them; what are the data requirements and the costs of obtaining them?

The question was posed as to the future roles of COMFIE and MAWG. Multispecies considerations are needed for the development of 'comprehensive' analysis. Coordination is needed with MAWG, or perhaps a successor, to avoid duplication of effort. Of special importance are the development of reference points that can be used in both single and multi-species systems. If, for example, COMFIE were to adopt the North Sea pelagics as a system for comprehensive analysis, there would be strong links to MAWG and the MSVPA results and an enhanced need/opportunity for coordination.

## 6 COMPREHENSIVE FISHERY EVALUATION OF THE NORTH SEA FLATFISH FISHERY

### 6.1 Introduction

A comprehensive fishery evaluation is thought to consist of a comprehensive assessment of the available data and regulations concerning a fishery and a model-based scenario analysis of different management scenarios. Both issues are addressed in this chapter for the North Sea flatfish fishery, which was chosen as one of the example fisheries because it exhibits technical interactions of a mixed fishery without substantial biological interactions.

The comprehensive assessment of the North Sea flatfish fishery aims to assess the current status and management of the fishery. The general question to be answered in this context is:

What do detailed biological, environmental, economical, managerial and behavioral data and analysis contribute to our understanding of the status and prospects of the North Sea flatfish fishery

The comprehensive assessment is presented in Section 6.2–6.6.

Next, a scenario analysis is performed on the effects of general harvest control rules when either or not technical interactions are taken into account (Section 6.7).

The major North Sea flatfish species plaice and sole are mainly exploited with beam trawls by the Netherlands, United Kingdom and Belgium. In the beam trawl fishery sole and plaice are generally caught together. There is a considerable by-catch of dab, which is usually discarded. Also valuable species like turbot and brill are caught, but in smaller quantities. There is a relatively small by-catch of roundfish (mainly cod and whiting).

Denmark is another major operator having a more directed fishery for plaice and sole using gill nets and Danish Seines. Germany has a predominant shrimp fishery off the German coast where substantial discards of juvenile plaice and sole are generated.

The fishery is mainly regulated by annual TAC's on the two target species plaice and sole and on the by-catch species cod and whiting. TAC regulations are accompanied by various technical measures, like minimum mesh sizes, minimum landing sizes and area restrictions (e.g., Plaice Box). The Netherlands and the United Kingdom have rather complex sets of national regulations for the implementation of the quota system.

The stocks, particularly sole, have allowed reasonable economic results since 1990. There is still a large technical capacity in relation to the size of the resource, in spite of decommissioning. The beam trawling fleet is vulnerable to reductions of sole catches and increases in fuel price, and these may cause serious economic problems. Combined with the rather high level of fishing mortality this is an important management problem.

Major issues in this fishery and the related advice are:

- The beam trawl fishery generates high discard rates of juvenile fish, mainly plaice and dab
- The beam trawl fishery causes bottom disturbance and hence ecosystem effects e.g., reduced survival of long-living sedentary species
- Technical interactions between fleets, gears used and target species play a predominant role causing among other things discarding of fish if quota are overshoot
- Possibly high-grading of less valuable fish occurs e.g., during spawning season
- Changes in growth and maturity of the main target species have been reported which may affect the consistency of biological advice
- Need to assimilate sensitive national data into an international database for research purposes
- Long term management intention to reduce fishing effort

### 6.2 Historical perspective

Before the introduction of steam vessels in 1884, the Belgian, German and Dutch fleet consisted only of rowing boats and sailing vessels. Towards the end of the 19<sup>th</sup> century a large increase in the fishing effort for the larger vessels was reported. By then fishing was carried out in the whole North Sea, the Dogger Bank, the Great Fisher Bank towards the coast of Norway, Iceland, the Barents Sea etc.

The number of sailing vessels reached a maximum between 1910 and 1920 with over 600 for Belgium and over 5000 (of which 500 trawlers) for the Netherlands. The Dutch vessels had an average GRT of 11 in 1910 decreasing to 3.5 in 1950. The trawling sailing vessels had an average GRT of 31 in 1910 and 16 in 1940.

Steam vessels were introduced in 1884 and caused a first boost in trawling effort, which coincided with the introduction of the otter trawl. Steam trawlers had their highest success by the end of the 1920s and were virtually non-existent by the end of the Second World War.

The first vessels equipped with diesel engines were introduced around 1901 and were to take over completely in the 1950s. This motorization caused a second increase in trawling effort. Originally, the otter trawl was the most popular gear.

In the early 1960s the beam trawl was re-introduced in the Belgian, Dutch and German fishery. The original light wooden construction (at that time still used in the German shrimp fishery) was replaced by a double rig (at both sides of the vessel) heavy steel gear often equipped with tickler chains and later sometimes chain mats. There evolved a continuing trend towards increasing engine power since the increases in number of tickler chains and towing speed were both found enhance catchability (Daan 1997). The maximum engine power of new beam trawl vessels that entered the fishery was restricted to 2000 HP (1470 kW)<sup>1</sup> in the Netherlands (from 1988 onwards) and to 833 kW for Belgium.

There is a need for further work on this topic, e.g., to work up a further review of the history of the stock based on available information such as that presented by Rijnsdorp & Millner (1996) and Millner & Whiting (1996). Other important references include Lindeboom & De Groot 1998 and Millner *et al.* 1996. Additional related papers include De Boer 1984, Tesch & De Veen 1933, Toet & Ouwehand 1967.

National fleet statistics and landings and economic information is given in Anon. 1912–26, 1927–29, 1931, 1934–38, 1950–57, 1959, 1976, 1991, 1992; LEI 1968–1992; Welvaert 1991, 1993

## **6.3 Fisheries**

### **6.3.1 Fleet composition and capacity**

#### **Data available**

##### EU

All EU countries have to keep databases of logbook data and vessel registration data for control purposes. These data are only necessary for vessels over 10 m. In most countries (part of) this information is available for research purposes. There is however a major concern with the sensitivity of this information and therefore additional restrictions apply to the use of these data, which may differ between countries. Furthermore, the data are not always comparable and access for international research purposes is often restricted.

In addition to the more recent logbook and registration data, some countries have older data available. The WG notes that in general it would be appropriate to collate these data which may be used for long-term evaluations of for example technical measures like closed areas (e.g., the Plaice box).

All EU registrations are organised by the flag a vessel is sailing. There has been substantial re-flagging of former Dutch beam trawlers to England, Scotland, Belgium, Germany and Denmark.

##### Belgium

No recent information available at this WG. Also it was uncertain what information could be made available.

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<sup>1</sup> 1 kW = 0.735 HP; 1 HP = 1.36 kW

### Denmark

No data was available at the WG. However, the following data-sources could be made available intersessionally:

- Logbook database (since 1987)
- Vessel registration database (since 1987)
- Sales-slip database (since 1987)
- Old database system containing catch and effort statistics (1974 – 1986)
- STECF database on composition of EU fishing fleets

Using the data sources listed above, it would be possible to reconstruct the catch, effort and value by trip and by market category.

### France

No data was available at the WG. However, the following data-sources could be made available intersessionally:

- Logbook database
- Vessel registry database
- Sales-slip database

It was not known to the WG since what year the data would be available. Using the data sources listed above, it would be possible to reconstruct the catch, effort and value by trip and by market category.

### Germany

No data was available at the WG. There is at least a database that contains logbook data since 1995. It is not certain what data exists from before 1995.

### The Netherlands

All data was present at the WG. It consists of the following data sources:

- Combined Logbook and vessel registry database (since 1990)
- Combined Logbook and vessel registry database for foreign vessels landing in the Netherlands (since 1995)
- Old database system with catch and effort data including market categories (1967–1982)
- Total beam trawl effort (since 1965)
- Total beam trawl capacity (since 1972)

The Dutch logbook database does not contain fishing hours (it is not obligatory to fill these out). Therefore additional assumptions are needed to convert total trip-time to actual fishing hours per rectangle. See below.

### England and Wales

All data was present at the WG. It consists of the following data source:

- Combined Logbook, vessel registry and sales database (since 1983)

### Scotland

No data was available at the WG. However, the following data-sources could be made available intersessionally:

- Combined Logbook, vessel registry and sales database (since 1992)
- Old logbook database (before 1992) – only available on magnetic tape

Landings and value by vessel and trip. Contains: fishing hours, number of tows, catch in kilo's, landings in value. Available since 1992 (approx.). Earlier years available on magnetic tape.

Contact person: Aileen Shanks

## Methods

The Dutch logbook database contains information by trip and by ICES rectangle (30x30 nm) for all vessels fishing under a Dutch registration. The database contains the number of days-at-sea (DAS) per trip. If more than one rectangle was visited during a trip, the total trip-time was subdivided across the rectangles according to the proportion of the total value of the demersal landings (plaice, sole, cod and whiting) in each rectangle.

$$E_{t,i} = E_t * \frac{V_{t,i}}{\sum_i V_{t,i}} \text{ where}$$

$E_{t,i}$  is effort in trip t and rectangle i.

$V_{t,i}$  is value of the landings from trip t and rectangle i.

Effort is expressed as days-at-sea or as HP days. Days at sea were not corrected for the number of hours fishing per day.

## Results

Some of the results presented below are taken from a bio-economic evaluation of multi-annual and multi-fleet measures (LEI-DLO *et al.* 1996) and from STECF working document (STECF 1998).

All EU-member countries surrounding the North Sea are participating in its flatfish fishery.

The main fishing gear used for exploiting plaice and sole in the North Sea is the twin beam trawl. This gear is used by virtually all Dutch and Belgian flatfish vessels and a large part of the British flatfish fleet. It contributes approximately 80% to the total fishing mortality of flatfish (70% of plaice and 90% of sole).

The total North Sea beam trawler fleet is estimated to comprise nearly 600 vessels, having an aggregate gross tonnage of around 120 000 G(R)T and a combined engine power of about 470 000 kW. The total beam trawling effort of this fleet in the North Sea was around 75 mln kW-days in 1993. The fishery has a labour input of around 2500 man-years.

Apart from beam trawls, plaice and sole are caught by otter trawls, Danish and Scottish seines and set nets (gill nets). Denmark and England still have a small otter trawl fishery directed at plaice, but generally plaice is just a by-catch of otter trawling. Very little sole is caught by otter trawls. With Danish and Scottish seining, a directed fishery for plaice is carried out, mainly by Danish and British vessels where by-catches of sole are practically zero. Also, mainly Danish fishermen have a seasonal directed fishery for sole with a small by-catch of plaice using gillnets with mesh sizes between 90 and 120 mm. There is also a Danish directed fishery for plaice using gillnets with meshes of 120–150 mm.

### Beam trawl fleets

The North Sea beam trawl fleet is composed of the following segments by nationality:

#### Belgium

In 1997 a fleet of 93 beam trawlers (larger than 70 GRT?), with an aggregate engine power of 54 000 kW operated under the Belgian flag. The total beam trawling effort was not specified. In 1993, approximately 70 percent of the effort was spent in the North Sea, but no recent Figures were available to the WG. The total labour of the beam trawler fleet is estimated at 450 man-years (STECF 1998). Some of the Belgian beam trawlers are owned by combined Belgian and Dutch interests.

#### Denmark

Around 5 smaller beam trawl vessels (less than 500 HP) are operated from Denmark of which 1 is fishing in the Skagerrak.



## France

In 1993, France had a small fleet of relatively small beam trawlers: 25 to 30 boats of 10 to 29 m length over all. Less than a quarter of their effort was directed at the North Sea. No recent data were available to the WG.

## Germany

At the end of 1993, Niedersachsen and Bremen had 9 beam trawlers fishing outside the coastal zone and 28 coastal fishing vessels partly engaged in beam trawling for flatfish. Schleswig-Holstein had 17 vessels fishing for white fish in the North Sea; part of this activity was beam trawling for flatfish. Part of the German beam trawlers is owned by combined German and Dutch interests and are generally operated by Dutch crews and from Dutch ports. No recent data were available to the WG.

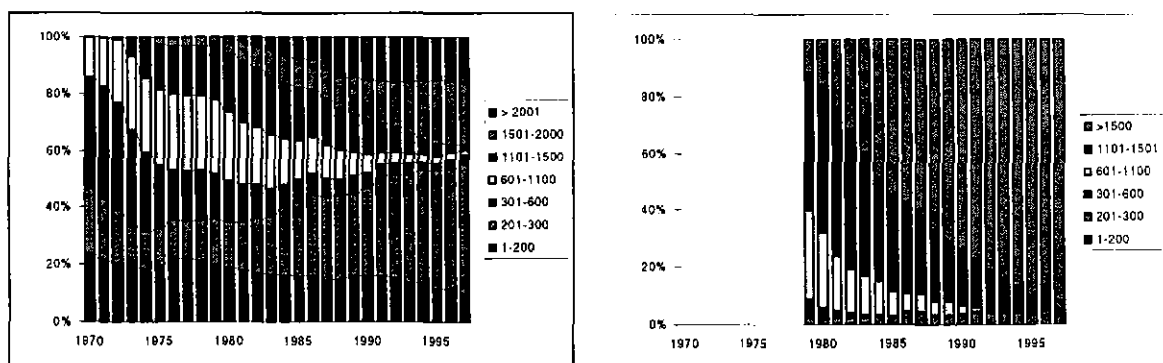
## The Netherlands

The Dutch fleet of beam trawlers consists of 416 vessels (in 1997), having a combined engine power of 455 000 HP. Out of this fleet, around 100 vessels (in 1993) also participated in other fisheries, like those for roundfish, herring or shrimp. The total beam trawling effort amounted to 72 mln. HP-days in 1997. The employment on board of the vessels has decreased to just over 1900 persons in 1997.

Figure 6.3.1 shows that the number of vessels was stable between 1970 and 1974 but horsepower increased reflecting a period of investment in new vessels that replaced old ones. Then a period of stable horsepower followed while the fleet diminished because many of the smaller of vessels were out-competed. A new investment wave until the late 1980s shows up in the total horsepower of the fleet but less so in the number of vessels. During the last 10 years the number of vessels and the total horsepower have come down in about the same rate (Daan, 1997).

Figure 6.3.2 shows in more detail the fleet composition by percentage of the Dutch fleet. The size category 200–300 HP increased probably because the 12-mile limit and later the Plaice box offered special fishing rights to this fleet component. Also, the fleet component above 1500 HP has expanded. The total fishing effort exerted by the Dutch fleet in HP days in recent years consisted for a large majority of vessels over 1500 HP.

**Figure 6.3.1** Number of vessels and total capacity (1000 HP) of the Dutch beam trawl fleet. Source: LEI-DLO and Smit *et al.* 1997



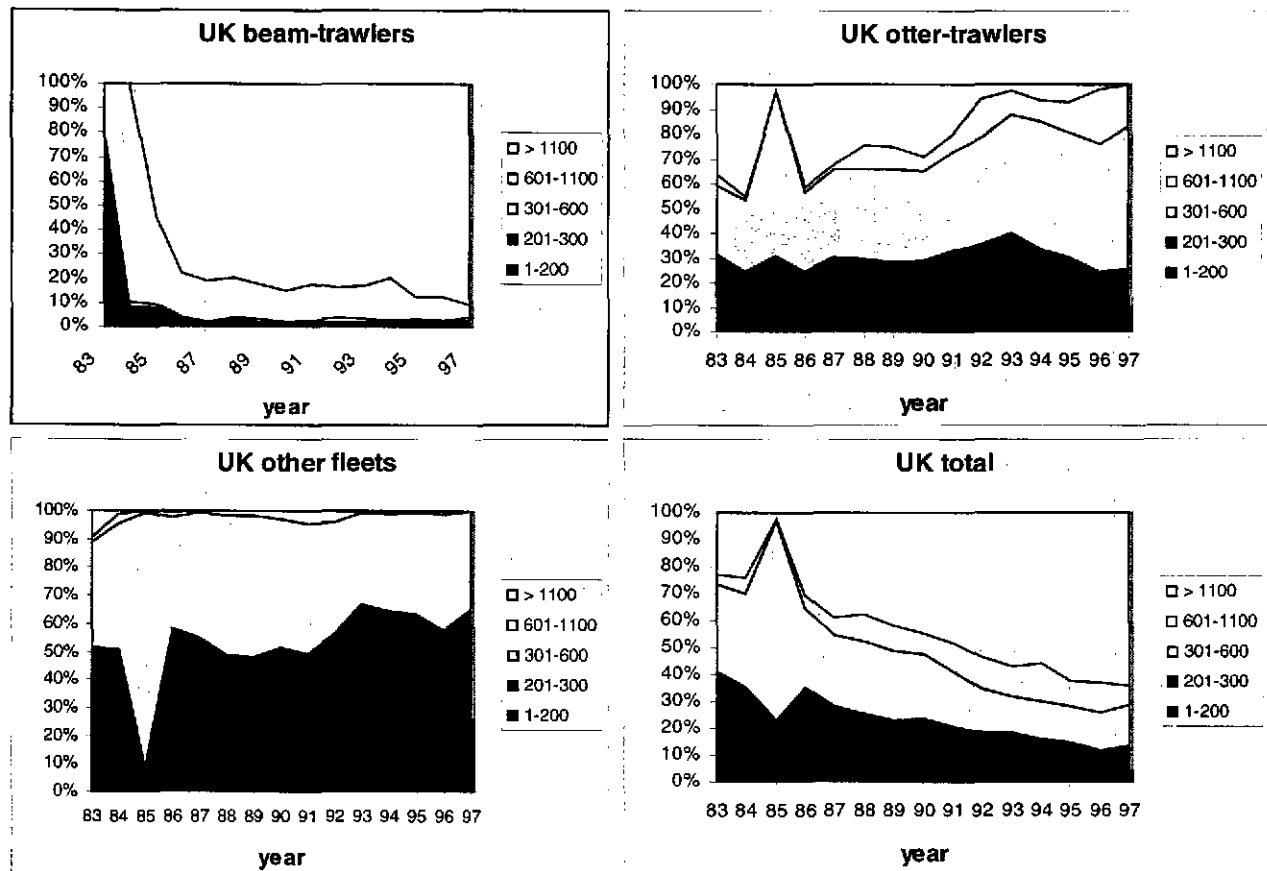
**Figure 6.3.2** Relative composition of the Dutch beam trawl fleet (left) and contribution to the total fleet effort expressed in HP days (right). Source: LEI-DLO and Smit *et al.* 1997.

## UK (England, Wales and Scotland)

The British beam trawler fleet fishing in the North Sea numbered 187 vessels in 1993, spending about 0.25 mln. fishing hours. Part of those are relatively small coastal fishing vessels, and part of the fleet is based in ports at the South and West coasts, only fishing in the North Sea occasionally. It is estimated that around 70 vessels ranging between 24 and 40 m over all length have their main fishery in the North Sea. About 60 percent of these boats are owned by combined

British and Dutch interests. They are generally manned by Dutch crews and operate from Dutch ports. (Amongst those are 20 vessels based in Scotland, having a combined engine power of 24 000 kW and an aggregate gross tonnage of 5 200 GRT).

In Figure 6.3.3 it can be seen that the total effort exerted by the English beam trawlers is predominantly by vessels larger than 1100 HP. There seems to be an anomaly in the data with regards to the gear classification in 1985.



**Figure 6.3.3** Proportion of effort (HP days) in different fleet segments for the UK fleet allocated to the different HP categories. Source: UK logbook data.

#### Non-beam trawl fleets

The non-beam trawl fisheries cannot be described with a similar precision, as those fisheries are generally only seasonally directed at flatfish, if at all. The main directed fisheries are:

- otter trawling for plaice in Denmark;
- seining for plaice in Denmark and the United Kingdom;
- netting for sole, plaice or turbot in Denmark and Germany.

Only very general data on the fleets involved are available.

#### Denmark

Nearly 40% of Danish plaice landings from the North Sea in 1992 were the result of seining. The seiner fleet of the North Sea fishing districts counted 148 vessels, having a total gross tonnage of 5 500 G(R)T, a total engine power of 21 600 kW and a total crew of 455 men. The effort is more or less concentrated in the period from March to September, when nearly 80% of the North Sea plaice landings are made.

The Danish fleet of netters in the North Sea districts counted 408 boats in 1992, having an aggregate gross tonnage of 5 600 G(R)T, a combined engine power of 34 800 kW and a total crew of 920 men. According to the landing value, over 40% of the effort of these vessels concerned fishing for flatfish. Netters account for 80% of the Danish North Sea sole catch (75% of that of turbot) and 30% of that of plaice. The sole fishery is highly seasonal, the season running from March until August. It peaks in April, when nearly half the total catch is made.

In 1992 about 15% of North Sea plaice landings in Denmark was made by (otter) trawlers under 100 GRT. Most of this was the result of directed fishing. Another 15% was landed by trawlers over 100 tons, mostly as a by-catch, or as the result of a mixed fishery. Data from 1988 show that about 40 trawlers of 10–70 G(R)T were fishing specially for plaice in that year, making about 1000 trips in total. The size of this fleet has decreased since.

### Germany

A small fleet of about a dozen German Baltic netters also profit from the sole season in the North Sea.

### UK (England and Wales)

The UK North Sea seiner fleet counts only about 40 boats. They are generally of a similar size and operate in a similar way as the Danish seiners.

The total non-beam trawling North Sea flatfish fleet was estimated to consist of nearly 650 boats in 1993, having an aggregate gross tonnage of around 14 000 G(R)T, a combined engine power of around 70 000 kW and a total crew of some 1700 men. (At a 60% dependence on fishing for flatfish, this amounts to a total employment of around 1000 man-years.)

The total fleet of vessels engaged in directed fishing for flatfish in the North Sea, both beam trawl and non-beam trawl together, was estimated at 1250 boats, with an aggregate gross tonnage of around 134 000 G(R)T and a combined horse power of around 540 000 kW. The North Sea flatfish fishery provides an employment equivalent to around 3 500 man years, but the number of crew members engaged in it for at least part of the year was estimated over 4 500.

## **6.3.2 Fleet dynamics: effort allocation, interference, targetting, catchability and technical interactions**

In general the data available on fleet dynamics is the same as the data-sources mentioned in Section 6.3.1. In addition to these data-sources, the following additional sources are available:

- Observer on board program in Denmark (since 1996). It is not yet certain, however, how available the results will be for publication.
- Micro-distribution data. These detailed data were assembled from a sample of the Dutch beam trawl fleet in the period 1993–1998 and data collection is still being continued (Rijnsdorp *et al.* 1998). In this project, the spatial distribution of Dutch beam trawlers was recorded using automatic position recording equipment (APR) that was connected to the navigator (Decca, GPS, DGPS). The APR device recorded the position information with fixed small time intervals and a spatial resolution of around 180 meters. The speed of the vessel (S) was calculated from the distance covered between two recordings. The speed during fishing (FS) was related to the engine power ( $\pm 6$  knots) but clearly distinct from speed during steaming ( $\pm 12$  knots) and during floating ( $\pm 2-4$  knots) when handling the catch. The sample of the fleet comprises 25 beam trawlers (larger than 300 HP) and stratified by harbor and HP category.

The most recent combined international effort index for North Sea flatfish fisheries was compiled in 1994 (ICES 1994). It is recognised that this index needs updating, which should be reasonably easy given the data sources presented in 6.3.1.

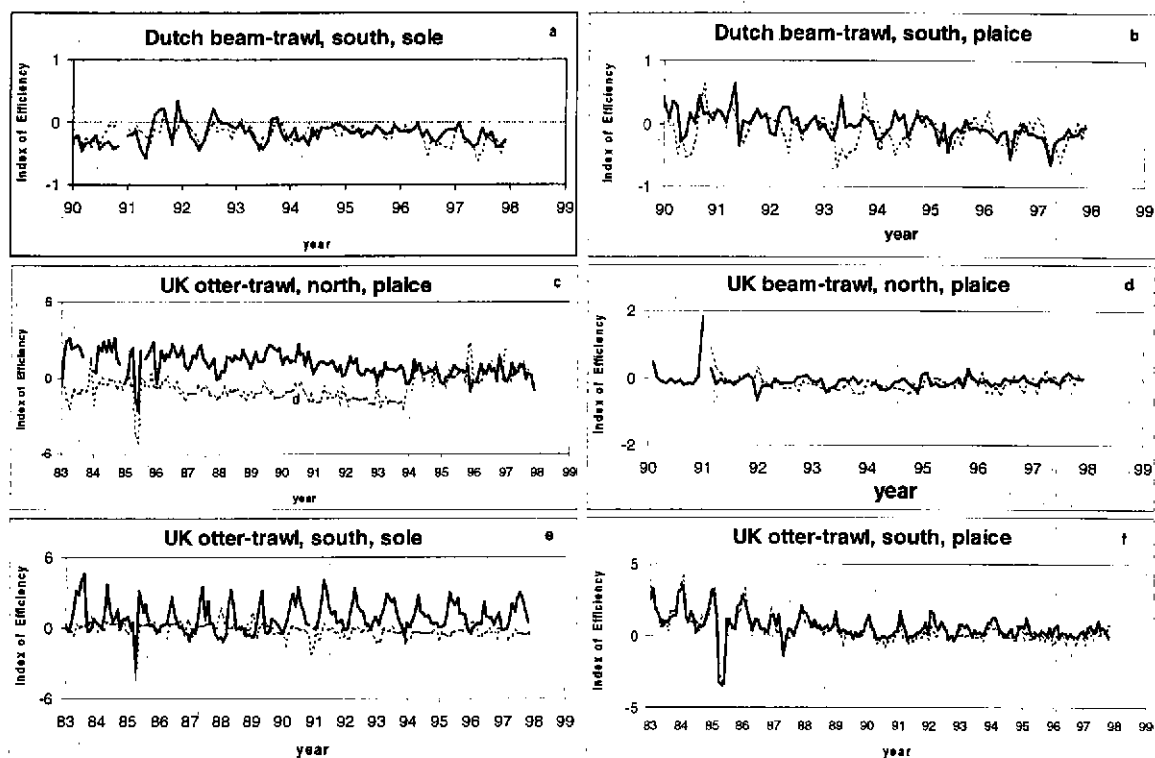
An index of technical efficiency has been calculated for the Dutch and UK flatfish trawlers. This index has been derived, from the methodology presented in section 12.6, as the log-ratio between the CPUE of each fleet (F) and the CPUE of one reference sub-fleet made up of vessels belonging to fleet F. The vessels of each fleet have been ranked in relation to the variance of their log-CPUE, and the vessels with the lowest variance were included in the reference sub-fleet. In practice, 10 vessels were retained for each fleet; vessels ranked 1–5 were allocated to one reference sub-fleet, and vessels ranked 6–10 were allocated to another one. The rationale behind the choice of 2 reference sub-fleets was to check whether the variability in the index of technical efficiency was significantly dependent on the choice of a

particular sub-fleet. The fleets used in the analysis are the Dutch beam-trawlers and the UK beam- and otter-trawlers. The selected trawling fleets caught both sole and plaice. The North Sea has been split into a northern and a southern areas. Temporal variations in the index of efficiency have been presented in Figure 6.3.3a-f for a selection of fleets, species and areas.

Variations in the index hardly depend on the choice of the reference sub-fleet for Dutch beam-trawlers catching sole (Figure 6.3.3a), UK beam-trawlers catching plaice (Figure 6.3.3d) and UK otter-trawlers catching plaice in the southern area (Figure 6.3.3f). Variations are also consistent for part of the time series for UK otter-trawlers catching plaice (period 1983–1993, Figure 6.3.3c). In particular, peaks (Figure 6.3.3c) and seasonal patterns (Figure 6.3.3f) are generally observed simultaneously with both reference sub-fleets.

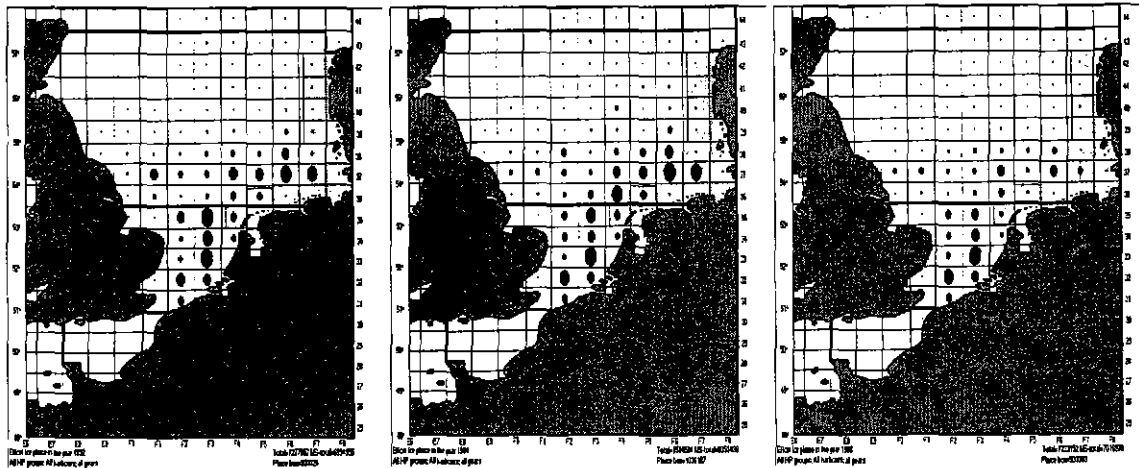
The temporal dynamics of the efficiency index were dependent on the choice of the reference sub-fleet in the other cases (Figure 6.3.3b, Figure 6.3.3c – period 1994–1996, Figure 6.3.3e). In particular, in Figure 6.3.3c, the dramatic peak observed with reference sub-fleet 1 could not be detected with reference sub-fleet 2. In Figure 6.3.3e, a negative trend was detected in both time series, but the marked seasonal pattern observed with reference sub-fleet 1 did not appear with reference sub-fleet 2.

The results derived from this analysis should be interpreted cautiously, since some of them depend on the choice of the reference sub-fleet. However, it may be anticipated that the efficiency of the North Sea flatfish fisheries has generally been decreasing in the past 10–15 years. More accurate analysis of efficiency could be provided by including external factors such as socio-economics in this analysis.



**Figures 6.3.3a-f.** Temporal dynamics of the index of technical efficiency calculated from a selection of North Sea flatfish fisheries. The plain lines refer to indices derived from sub-fleet 1 (vessels ranked 1–5); the dashed lines refer to indices derived from sub-fleet 2 (vessels ranked 6–10).

The spatial distribution of the Dutch beam trawl effort is shown in Figure 6.3.4. The 1996 distribution patterns seems to have changed as compared to 1992 and 1994. The concentration of fishing effort in 1996 is more southerly than before. The southerly distribution could be a behavioural response of fishermen to the tight plaice quota in 1996, which entailed an increased targeting on sole.



**Figure 6.3.4** (Relative) effort distribution of the Dutch beam trawl fleet in 1992 (left), 1994 (middle) and 1996 (right). The grey zone is the plaice box. The size of the dots refers to the proportion of effort exerted in one year in each rectangle. Source: NL Logbook data.

The relationship between effort, targeting, catchability and CPUE for North Sea flatfish stocks is addressed in working documents and reports presented at several occasions (Frost *et al.* 1995; Pastoors *et al.* 1997, Pastoors *et al.* 1998a, Pastoors 1998). Information for roundfish stocks can be found in Cook & Armstrong (1985) and Cook (1997).

An GLM model was estimated to evaluate the impacts of different variables on the log CPUE (Pastoors *et al.* 1997):

$$\log CPUE = constant + b_1 \log HP + b_{2,i,j} month_i * area_j + b_{3,i,j,k} flag_i * month_j * year_k$$

Where:

CPUE	Catch of plaice or sole divided by the days-at-sea
HP	Engine power of the vessel (continuous variable)
Month	January, February, ..., December
Area	The North Sea was split up into 7 areas
Flag	Boolean; has value 0 if it is a Dutch beam trawler and 1 if it is a flag vessel
Month*area	The model is estimated for all months and all areas (12*7 = 84 coefficients)
flag*month*year	All groups, all months, and all years (gives 84 coefficients (2*12*4))

CPUE was expressed as catch per day-at-sea where days-at-sea were based on the total time away from port.

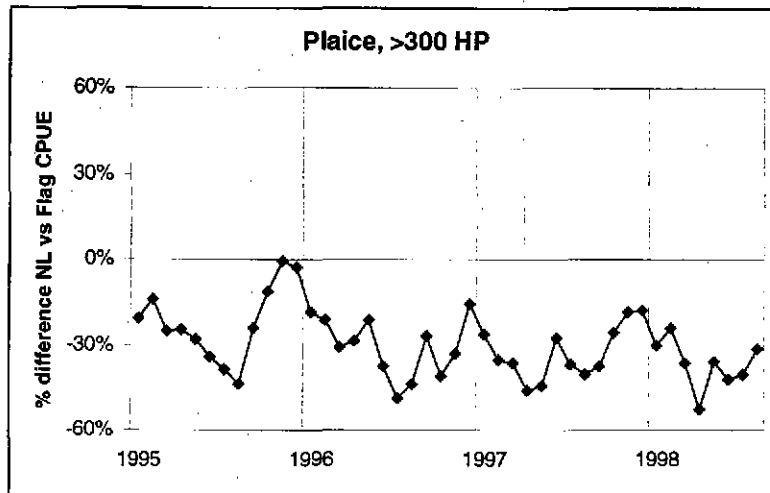
The variable *month\*area* represents the period- and location effect. The coefficients of the *flag\*month\*year* variable indicate the differences in CPUE between Dutch beam trawl vessels and flag vessels over time after filtering out the vessel category, period, and location effect. The relative (percentage) difference in CPUE can be calculated using the differences of the coefficients of *flag\*month\*year*:

$$\%diff = \frac{e^{b_{NL}}}{e^{b_{Flag}}} - 1$$

A negative percentual difference indicates a higher CPUE for flag-vessels, a positive difference a higher CPUE for Dutch beam trawl vessels.

It was shown that CPUE of plaice is dependent on the flag of the vessels involved. The analysis was based on the data available in the Dutch logbook database which consists since 1995 of all Dutch landings and landings in the Netherlands by foreign vessels. By selecting those vessels that were originally Dutch and re-flagged to another flag (e.g., German, Danish, Belgian or British), and comparing their trips with vessels that still fly the Dutch flag, the authors were able to show that the larger flag-vessels (> 300 HP) had consistently higher CPUE's as compared to the

Dutch vessels (Pastoors *et al.* 1997; Figure 6.3.5). These results may indicate that CPUE is dependent on the flag a vessel is flying and by that the available quota for the fishermen (the assumption being that vessels are re-flagged in order to be able to exploit a larger individual quotum). If larger quota enable higher CPUE's then CPUE is not only an indicator of stock size but also of applicable management.

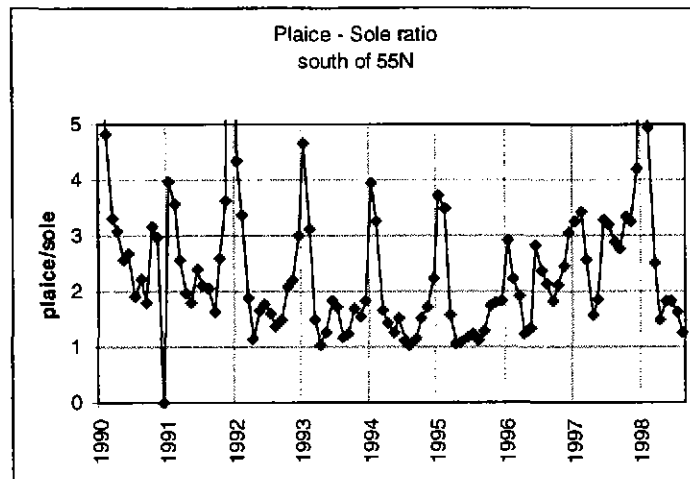


**Figure 6.3.5** Relative differences in plaice CPUE of NL vessels and flag-vessels in the period 1995 to mid 1998. Results from GLM model: A negative percentual difference indicates a lower CPUE for NL vessels.

A comparison of the methods outlined above (both GLM and technical efficiency) using all available international data is essential to make progress in understanding the developments in fishing power and efficiency and is critical in understanding the role of management in the fishery.

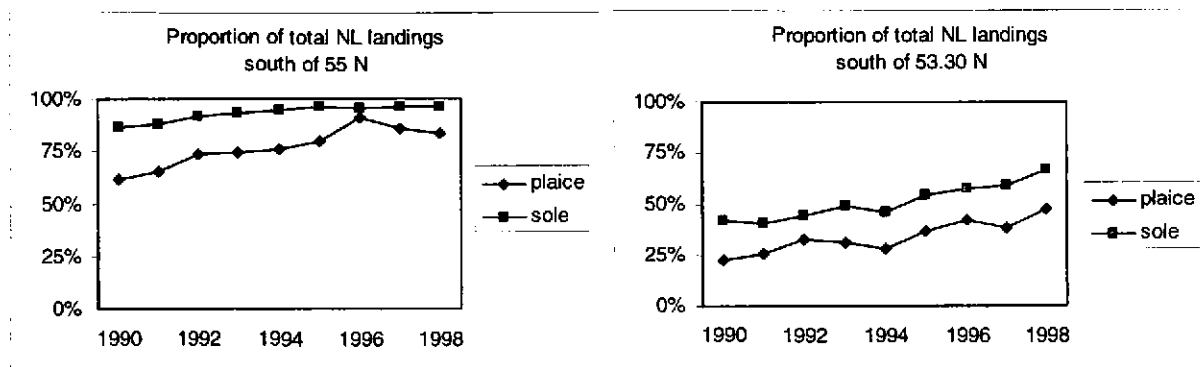
Competitive interactions among vessels have been analysed by Rijnsdorp *et al.* (WP 8, WP 9; see also Section 12.6) for the Dutch beam trawl fleet. It was shown that competitive interactions may well influence the relationships between effort and fishing mortality and thus have important implications for fish stock assessment. First, they may bias the index of abundance of commercial fish stock sizes estimated by the catch per unit of effort of commercial fisheries (Gillis *et al.*, 1993; Gillis and Peterman, 1998) something which is not always realised (Sampson, 1991). Second, they may affect the catchability coefficient as defined in the equation  $F = q * effort$ . In the present situation of North Sea flatfish where fish stock are overexploited and fishing mortality has to be reduced (ACFM, 1999), a disproportional reduction in fishing effort may be necessary to achieve fishing mortality targets. For example, a reduction in the number of vessels, resulting in a reduction in competitive interactions and an increase in  $q$ , may have a different impact on the fishing mortality than a reduction of effort in which competitive interactions are not reduced or even increased (closed seasons, closed areas).

The linkage between plaice and sole in the Dutch beam trawl catches was analysed using the Dutch logbook data. There is a strong seasonality in the ratio between plaice and sole in the area where they are caught in a mixed fishery (i.e., south of 55 N where a mesh size of 80 mm is used; Figure 6.3.6). The high plaice over sole ratio in the winter indicates the winter fishery for spawning plaice. The average ratio over all years and months was 2.2 which means that for each kilo of marketable sole, 2.2 kilo's of marketable plaice are caught.



**Figure 6.3.6** Ratio between plaice and sole landings (in weight) for the Dutch beam trawl fleet fishing south of 55 N. Source: Dutch logbook data.

When all catches south of 55 N are selected, around 95% of the total Dutch sole landings are included and between 60% and 90% of the plaice landings (Figure 6.3.7). The proportion of the plaice landings that is taken south of 55 N is shown to increase over time which indicates that the linkage between plaice and sole is getting stronger. Furthermore, as the linkage between plaice and sole south of 53.30 seems to be rather constant, the difference will most likely be caused by changes in either availability of fishing patterns between 53.30 N and 55 N.



**Figure 6.3.7** Proportion of the total NL landings of plaice and sole south of 55 N (left) and south of 53.30 N (right). Source: Dutch logbook data.

### 6.3.3 Gear and selectivity

No work was undertaken on the issue of gear and selectivity. However, it was recognised that this issue should be taken up in a comprehensive assessment of this fishery. Special attention should be paid to the relationship between fishing speed, net configuration and selection as traditional selection curves that are available seem to have been collected mainly in the early 1980s and therefore miss the technological development in the fishery towards larger engines and increases in fishing speed.

### 6.3.4 Catches, by-catch, catch composition and market sampling

#### Data available

##### Belgium

No data available to the WG.

### Denmark

No data available to the WG.

### France

No data available to the WG. It is noted that the market categories in France do not show any stability and are also regionally different, which makes generic sampling schemes difficult to maintain.

### Germany

No market sampling for plaice or sole.

### The Netherlands

Official catch statistics are available from the Ministry of Agriculture, Nature and Fishery. In the 1980s official catch statistics were uncertain due to unreported landings (see: section on high grading, discarding and unreported landings). Since 1990 catches are available considered more reliable and are available from the logbook database (van Beek *et al.* 1998). Cod and whiting are often caught as a by-catch in the beam trawl fishery and data on these species is available in the same format as plaice and sole. Information of by-catches of other species – which in some instances may be target species as well – like dab, turbot and brill is not readily available.

Market sampling is restricted to the Dutch beam trawl fleet and the results are raised to total national landings. Stratified samples of the landings are bought directly from vessels in the fish auctions. There are three levels of stratification: by harbour, quarter and market size category. Fish are sampled for age, length, weight, sex and maturation

Sampling is restricted to 4 major landing ports, which account for about 80% of the national landings. The number of samples are taken approximately in proportion to the expected landings in these ports and take account of differences in effort of various fleet components in different fishing areas.

The stratification by quarter allows the maintenance of a quarterly data base. For assessment purposes the quarters are combined to annual Figures. Sampling during spawning time (plaice: first quarter; sole: second quarter) is intensified to obtain detailed biological parameters (maturity data and weight at age of the stock).

Plaice landings are graded in 4 and sole in 5 size categories which are consistent over time. Samples consist of 15 (plaice) or 10 (sole) fish in each size category bought from the vessel selected. In addition, the total landings of the vessel, the amount landed in each size category, gear, fishing position, vessel name and vessel characteristics are recorded. The total number of samples taken per year is around 80 for each species.

### UK – England and Wales

Total market sampling database was available to the WG. Analysis based on this database are presented in section 6.4.

### Total international:

Total international catch for plaice and sole is calculated from the reportings of the countries that catch the species involved. Two sets of numbers are available:

- official landings as reported to the EC and ICES by the different member states
- estimated landings or catches by the different research institute.

The latter estimates can contain differences with the official landings due to misreporting, discarding, different conversion factors etc. The catches as estimated by the research institutes are used in the stock assessment procedures. The difference between the total official landings and the total estimated catch is presented as 'unallocated landings' in the working group report.

An overview of total market sampling effort is presented in ICES (1999a).



## Results

Table 6.3.1 gives an overview of quota and catches by country of North Sea plaice for the years 1994 and 1997. Tables 6.3.2 gives the same data for North Sea sole.

	1994			1997		
	Quotum	Catch	%uptake	Quotum	Catch	%uptake
B	9,440	7,951	84%	5,389	5,223	97%
DK	30,680	17,056	56%	17,515	13,940	80%
DE	8,850	5,697	64%	5,058	4,159	82%
FR	1,770	438	25%	1,016	587	58%
NL	59,000	50,289	85%	33,682	34,143	101%
UK	43,660	27,749	64%	24,925	22,134	89%
non EEC	11,600	530	5%	3,415	1,779	52%
Unallocated		682			1,212	
<b>TOTAL</b>	<b>165,000</b>	<b>110,392</b>	<b>67%</b>	<b>91,000</b>	<b>83,177</b>	<b>91%</b>

**Table 6.3.1** North Sea plaice quota, official landings and % uptake for 1994 and 1997. The 1997 TAC has been revised during the year; originally set at 77,000 it was first raised to 81,000 tonnes and later in September of that year it was raised to 91,000 tonnes. Quota as originally set, therefore without transfers and exchanges. Sources: TAC Regulation (EC) and ICES (1999a).

SOLE	1994			1997		
	Quotum	Catch	%uptake	Quotum	Catch	%uptake
B	2,665	2,935	110%	1,500	1,519	101%
DK	1,220	1,804	148%	685	689	101%
DE	2,135	1,744	82%	1,200	510	43%
FR	535	498	93%	300	315	105%
NL	24,075	22,874	95%	13,545	10,241	76%
UK	1,370	1,137	83%	770	479	62%
non EC		298			205	
Unallocated		1,712			1,023	
<b>TOTAL</b>	<b>32,000</b>	<b>33,002</b>	<b>103%</b>	<b>18,000</b>	<b>14,981</b>	<b>83%</b>

**Table 6.3.2** North Sea sole quota, official landings and uptake for 1994 and 1997. Quota as originally set, therefore without transfers and exchanges. Sources: TAC Regulation (EC) and ICES 1999.

The above tables give the basic quotas before transfers and exchanges ('the swap'). Annually around 15 000 t of plaice (in 1993) are transferred from the UK to the Netherlands in exchange for its small quotas of flatfish and roundfish outside the North Sea, which are abandoned for control reasons. In this arrangement also a transfer of about 500 t of North Sea sole from the Netherlands to the UK is included. Table 6.3.3 gives a comparison of Dutch plaice quota before and after transfers and exchanges.

PLAICE	NL quatum before exch.	NL quatum after exch.	% incr.
1986	68,230	79,600	17%
1987	57,500	66,870	16%
1988	66,430	80,570	21%
1989	70,270	94,520	35%
1990	68,040	88,754	30%
1991	66,120	85,420	29%
1992	64,900	79,488	22%
1993	64,920	75,150	16%
1994	59,000	70,335	19%
1995	41,140	50,860	24%
1996	30,130	34,120	13%
1997	33,682		
1998	32,280	34,855	8%

**Table 6.3.3** Comparison of NL plaice quota before and after transfers and exchanges. Source: Smit *et al.* 1998.

The Netherlands are the main operator in the North Sea flatfish fishery, having a share of about 45% of the plaice TAC (after the swap) and 75% of the sole TAC. Denmark and the UK follow at some distance with shares of 18% and 17% (after the swap) respectively of the plaice and just under 4% and 6% (after the swap) respectively of the sole. Belgium and Germany have slightly larger shares of the sole TAC, with 8% and 7% respectively, but their plaice quota's are rather small: 6% and 5% respectively. France has only a small share in the North Sea flatfish fishery.

North Sea sole is exclusively caught by EU member countries. Norway is the only third country participating in the North Sea plaice fishery, but its landings are much smaller than the available quota.

- To do: By-catch in the beam trawl fishery; overview of turbot, brill, dab, cod, haddock and whiting catches in the beam trawl fishery.

Biological advice on commercially exploited stocks relies to a large extent on market sampling information. To investigate the variability and adequacy of international market sampling, and EU research project (CFP 98/075) will be implemented in 1999 and 2000 with the aim to evaluate the adequacy of a limited number of international market sampling programs and to enhance the optimal allocation of resources for these sampling programs. Measures of the uncertainty in the estimated catch in numbers and mean weights at age will be obtained which will improve the quantified estimation of uncertainty in stock assessments and catch forecasts. The following countries will participate in the project: The Netherlands, England, Scotland, Denmark and Belgium.

The main objectives of the project are:

1. To evaluate the adequacy of the international market sampling effort for a limited number of commercially exploited fish stocks in the ICES area.
2. To quantify the uncertainties in estimated catch in numbers and mean weight at age.
3. To advise on appropriate sampling levels and methods.
4. To define protocols and develop prototype software to store and aggregate national sampling data taking into account the results of 1 and 2.

The proposed project will focus on three types of stocks that have different biological properties, different fisheries and different sampling methods: North Sea herring, North Sea flatfish (e.g., plaice) and North Sea gadoids (e.g., cod).

To increase participation within the ICES community, an ICES study-group (by correspondence) has been initiated at the ASC 1998 with the same goals as defined in the EU proposal. ICES countries that are not participating in the project will be invited to participate in the project workshops.

### 6.3.5 Discarding, high grading and unreported landings

Discarding of undersized fish is a serious problem in the flatfish fisheries due to the relatively small meshed gear used. The problem mainly affects plaice (ICES, 1987). Discarding was estimated to be about 50% in numbers in the 1980s (van Beek 1990, 1998). It is likely, however, that the level of discarding has changed over time due to changes in market conditions, growth rate and effort distribution (technical measures such as the 12 mile zone and the 'plaice box'). Therefore, there is an urgent need to reconstruct the historical discarding pattern in order to adjust the time-series of VPA recruitment estimates. Such an exercise is feasible using the available data on the growth rate by cohort, the selection ogives estimated during discard trips made on-board of commercial vessels at different historic periods, length composition of the landings, and data on the distribution of fishing effort. The interpretation can be enhanced if information could be made available on the market condition of fish landed (prices by market category of plaice, sole and other by-catch species).

Stringent quota regulation may lead to high-grading by fishermen selectively discarding low-priced size classes. For plaice this may occur during the spawning period when large spent fish are discarded because of their low price. Alternatively, high grading may occur in autumn when a strong year class recruits to the fishery. High grading will cause a substantial bias in the perceived state of the stock due to error in the estimated catches; as well as biasing knowledge about their age structure. This problem requires further study and may become important in the future if restrictive TAC's are implemented (Gillis *et al.* 1995). To date, no empirical information is available on the level of high-grading in the North Sea flatfish fishery. It is suggested to apply economical analysis to determine the conditions in which high-grading is suspected to occur.

In the late 1970s and 1980s, the estimated total landings are uncertain due to unreported landings. In 1990 this situation has improved due to a stricter enforcement of the legislation, but a critical re-appraisal of the estimates of unreported landings and their age-composition is still required.

#### Data available

- Size distributions in the landing (CEFAS, RIVO-DLO, CLO, DIFRES)
- selection ogives from discard trips around 1970, 1980 and 1990 (RIVO-DLO)
- price information (LEI-DLO, CEFAS, DIFRES)
- growth data from surveys (various sources) and otolith studies (RIVO)
- effort distribution (CEFAS, RIVO, others)
- cost and earnings data (STECF 1998)

### 6.3.6 Economics

Prices are determined by demand and supply factors. On the demand side it is important to distinguish between the fresh market (restaurants, fish shops) and the filleting industry. The fresh market is mostly in need of large sized plaice and sole and prepared to pay a higher price as compared to the filleting industry. The filleting industry looks for substitution of other fish species when prices of plaice and sole are too high. It is a global market and not merely an EU market.

On the supply side the landings have a highly seasonal pattern which will determine the price to some extent and higher landings will generally result in lower prices.

Cost and earnings data are important to evaluate the economic performance of fishing fleets and to be able to understand the individual behaviour of fishermen. Earnings are determined by prices and the amount of landings (target species as well as by-catch species) but there are also other types of earnings such as the selling or hiring of quota to third parties.

Costs can be subdivided in:

- costs depending on effort (fuel costs, fishing gear, maintenance);
- costs depending on landings (crew share, auction costs); and
- depreciation and interest costs (for hull, engine and other equipment).
- overheads and insurance

Profits are the difference between costs and earnings and may result in investments in newer and more efficient vessels. There is as yet no functional relationship available between investment behaviour and explanatory variables.

## Results

Davidse *et al.* (1993) described methods and results of cost and earnings calculations that were based on a uniform basis for four EU countries (Denmark, France, the Netherlands, and United Kingdom). They evaluated tax regulations, social security payments, subsidies, remuneration systems and regulations to limit fishing effort. Regular cost and earnings studies are going on in Denmark, the Netherlands and the United Kingdom. However, net profit or loss per vessel is calculated differently in these countries. Therefore, harmonisation was set up both in the presentation of Figures and in their calculation.

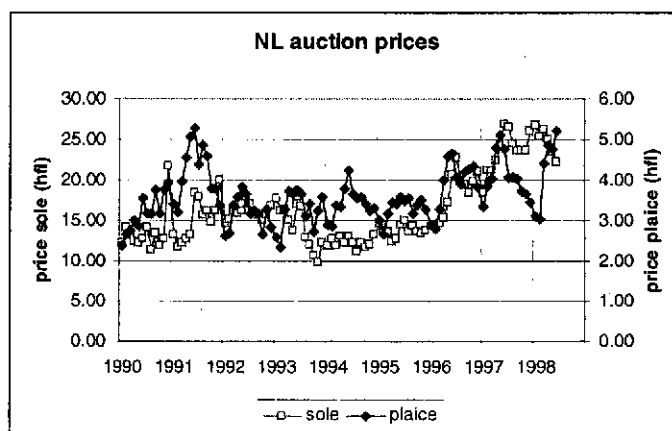
The common calculation method mainly referred to harmonisation of depreciation and interest costs and was applied to 1990 data. Average returns per vessel differed widely in the four countries. In Denmark, eight of the fifteen studied vessel groups showed net profits and these were mainly the smaller vessels. The Dutch cutter fleet showed to be profitable for seven out of ten vessel groups and here negative returns were mainly of smaller vessels due to declining catches in the shrimp fishery. The UK vessels showed positive results for three out of eight vessel groups but negative results were generally smaller than for e.g., Danish trawlers.

Economic performances of similar size vessels in the three countries showed large differences mainly due to different earnings per vessel. UK vessels were characterised by both high earnings and high costs.

Gross added value (total of profits/losses, interests, wages and depreciation) was stable compared to earnings at around 50–60%. Profitability as related to invested capital (measured by insurance value) was low for most vessel groups. Only nine out of forty vessel groups showed a rate of return on investments (ROI) exceeding or equal to the interest level, which is not uncommon for family owned enterprises.

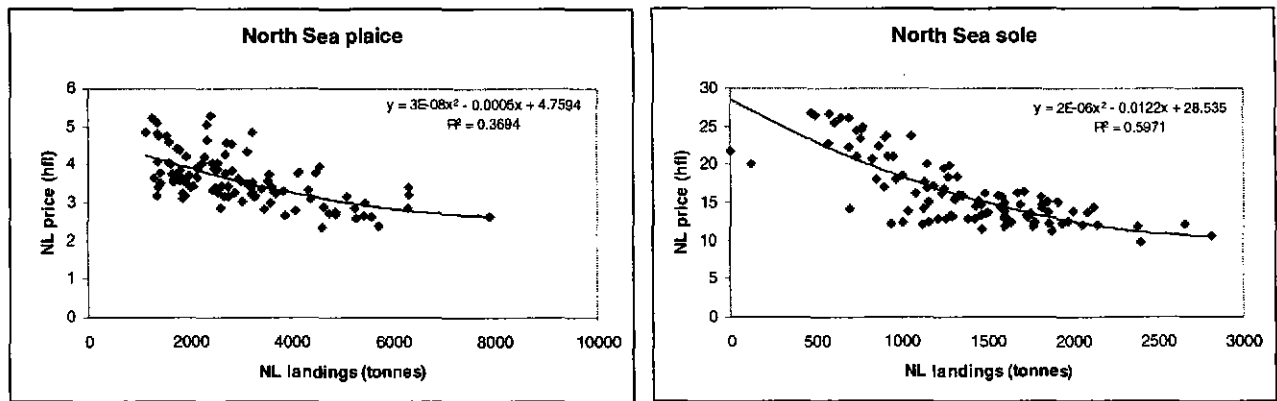
- To do: evaluate recent economic report (STECF 1998).

Average Dutch prices for plaice and sole are shown in Figure 6.3.8. It is shown that the relative low landings since 1996 have partly been compensated by the higher prices.



**Figure 6.3.8** Average Dutch prices for plaice and sole 1990–1998. Source: LEI-DLO.

The relationship between landings and price (in the Netherlands) for plaice and sole is shown in Figure 6.3.9. A simple polynomial function was fitted through the data. It is shown that sole has a higher price flexibility which is due to a larger demand from the fresh market compared to plaice which is predominantly filleted and exported.



**Figure 6.3.9** Relationship between landing and price for plaice (left) and sole (right) in the Netherlands. Source: LEI-DLO and RIVO-DLO.

To do:

- Evaluate costs. E.g. low oil price in recent years has considerably reduced costs in the energy demanding beam trawl fishery.
- Present overview of import / export by country.
- Evaluate revenues by fleet segments.
- Evaluate processing sector
- Evaluate markets

### 6.3.7 Environmental impact

North Sea flatfish are mainly exploited by beam trawl gear and concern has been raised that this gear, like other heavy trawls, may have a detrimental effect on the ecosystem, in particular on the benthos (ICES, 1988). The impact of (beam-) trawling has been studied extensively in the recent years (Lindeboom and de Groot 1998) and is still the subject of ongoing research:

- experimental studies of the direct impact on the sea-bed in trawled and un-trawled areas
- micro-distribution of the fishery and its relationship with benthos and sea bed characteristics (Rijnsdorp *et al.*, 1998).

The topic is on the agenda of the ICES Working Group on Ecosystem Effects of Fishing and results will be reported to this group.

Beam-trawling is a highly energy-intensive fishery and this aspect could be included in an assessment of the environmental impact of (beam-) trawling.

However, at this WG, no additional work was undertaken on this subject, due to a lack of time and therefore no new results can be presented.

### 6.3.8 Conclusions

It is shown in the above sections that although there is a large amount of data available at the different institutes around the North Sea, there has not been much progress in actually bringing the data together. The WG recommends that this issue should be taken up intersessionally.

It is also recognised that the economic part of the analysis is too weak so far, and collaboration with economists from different countries in joint projects is strongly encouraged.

It is shown that technical interactions play a predominant role in this fishery and should be taken into account when evaluating the management of this fishery (see also Section 4.x).

The analysis of technical efficiency in the Dutch and English fleets has shown that the efficiency of the North Sea flatfish fisheries has generally been decreasing in the past 10–15 years. However, more accurate insights in the analysis of efficiency could be provided by including external factors such as socio-economics in this analysis.

## 6.4 Biology

### 6.4.1 Stock structure, spawning and nursery areas, and migration

No new data or analysis are available since the last COMFIE WG (ICES 1997x).

### 6.4.2 Natural mortality

Estimates of natural mortality of juvenile plaice due to predation are available from Leopold *et al.* (1998). The authors concluded that in the years 1992 and 1993 the total summer consumption of 0-group plaice was around 12.55 million which accounted for 50% (1992) and 27% (1993) of the total stage 0-mortality in the Wadden Sea area.

### 6.4.3 Growth

No new data or analysis are available since the last meeting of the COMFIE WG (ICES 1997x).

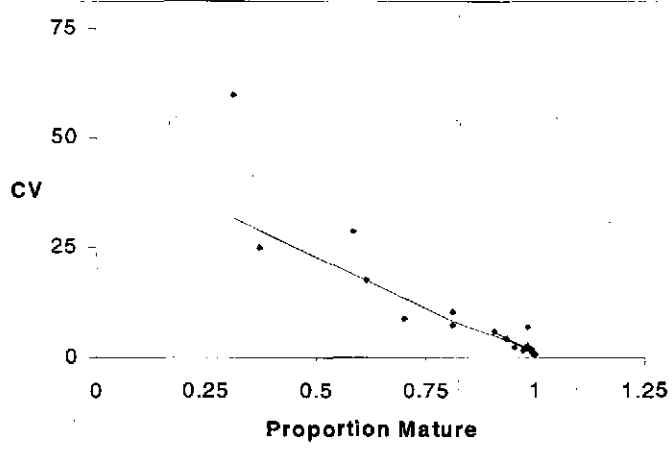
### 6.4.4 Maturation

Calculations have been performed of cv's on maturation using English market sampling. The coefficient of variation appears to be a function of the proportion mature rather than constant as is currently used (Figure 6.4.1).

Uncertainty in the proportion mature was modelled as a normal random deviate after a logistic transformation.

$$\text{i.e. } \text{logit } P \sim N \left( \log \left( \frac{\mu}{1-\mu} \right), \left( \frac{cv}{1-\mu} \right)^2 \right)$$

and the CV was user specified and where P was on the open interval (1,0). The maturity in the English market database was analysed to provide better estimates for the CV, current working group practice is just to specify a CV of 10%.



**Figure 6.4.1** Analysis of English market sampling data for relationships between proportion mature at age and the CV of the estimated maturity. Source: CEFAS.

#### 6.4.5 Fecundity and egg quality

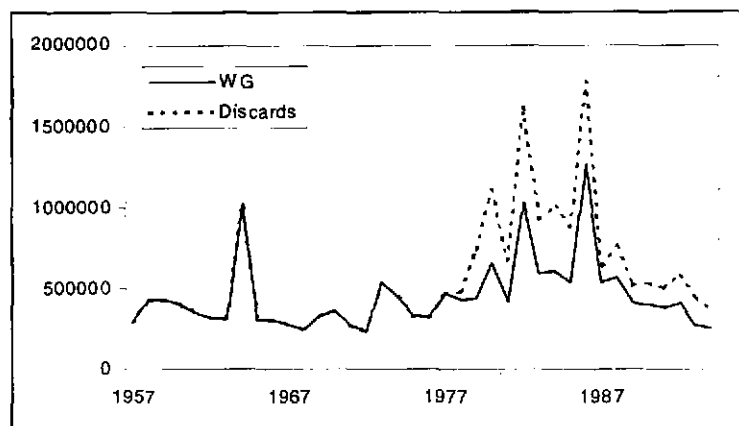
No new data or analysis are available since the last COMFIE WG (ICES 1997x).

#### 6.4.6 Sex ratio

No new data or analysis are available since the last COMFIE WG (ICES 1997x).

#### 6.4.7 Stock and Recruitment

Time series modelling of plaice recruits using ARIMA models is detailed in Kell *et al* (1998). Correcting recruitment of plaice using average discard rates as in ICES (1994) gives estimates of discards as in Figure 6.4.2.



**Figure 6.4.2** Estimates of recruitment from VPA when the catch at age matrix is adjusted for discarding using the average discard rates reported in the ICES study-group on the evaluation of the Plaice box (ICES 1994) compared to working group estimates (ICES 1999).

### 6.5 Management

The following text is largely derived from LEI-DLO *et al.* (1996).

#### 6.5.1 EU

The North Sea flatfish fishery is extensively regulated by the European Union: rules and regulations are in force in all four fields of the Common Fisheries Policy. These concern TAC's and quotas, technical measures, structural measures and marketing measures.

##### 6.5.1.1 TAC's and Quota's

Since 1975 the North East Atlantic Fisheries Commission has regulated the fisheries for plaice and sole in the North Sea by a system of T(otal) A(allowable) C(atch)es and quota's. This system was adopted by the European Community and integrated into its Common Fisheries Policy. In fact the EU is now the dominant partner in the fisheries regulation of the North Sea. The EU distributes its share of the TAC's over the member states according to a fixed key.

More details on TAC's and quota are given in Section 2.4.

##### 6.5.1.2 Technical measures

Technical measures have been laid down in Council Regulation (EC) No 3094/86 and successive amendments (latest: 850/98).

### Minimum landing size

The following minimum landing sizes are in force for 'protected' flatfish species in the North Sea [in cm]:

plaice	<i>Pleuronectes platessa</i>	27
witch	<i>Glyptocephalus cynoglossus</i>	28
lemon sole	<i>Microstomus kitt</i>	25
sole	<i>Solea vulgaris</i>	24
turbot	<i>Psetta maxima</i>	30
brill	<i>Scophthalmus rhombus</i>	30
megrim	<i>Lepidorhombus spp.</i>	25
dab	<i>Limanda limanda</i>	23
flounder	<i>Platichthys flesus</i>	25

### Minimum mesh size

The minimum mesh size for fishing for demersal species in the North Sea is 100 mm. Of the few derogations to this general rule, one is particularly relevant to the flatfish fishery: fishing for sole South of 55° N is allowed with 80 mm meshes in the cod end, provided that at least 5% of the catch is sole, and no more than 10% of the catch is composed of cod, haddock and saithe.

The 80 mm mesh used in most of the flatfish fishery results in substantial discards of undersized plaice. Before 1987 the minimum mesh size in the sole fishery was 75 mm.

### Maximum beam length

An additional technical measure concerning the fishing gear is the restriction of the aggregate beam length of beam trawlers to 24 m. This limits the fishing power of beamers over about 1300 kW to a certain extent, by reducing the swept area, but this may be partly compensated by increasing the fishing speed.

### Twelve mile zone

Beam trawling is not allowed in a 12 nautical mile (22.2 km) wide zone along the British coast, except for vessels having an engine power not exceeding 221 kW and an overall length of 24 m maximum.

In the 12 mile zone extending from the French coast at 51° N to Hirtshals in Denmark no trawling is allowed to vessels over 8 m overall length. However, otter trawling is allowed for vessels of maximum 221 kW and 24 m overall length, provided that catches of plaice and sole exceeding 5% of the total amount of fish on board are discarded. Beam trawling is only allowed to vessels included in a list that has been drawn up for the purpose. The number of vessels on this list is bound to a maximum, but the vessels on it may be replaced by other ones, provided their engine power does not exceed 221 kW and their overall length is 24 m maximum. Vessels on the list are allowed to fish within the twelve mile zone with beam trawls having an aggregate width of 9 m maximum. To this rule there is a further derogation for vessels having shrimping as their main occupation. Such vessels may be included in an annually revised second list and are allowed to use beam trawls exceeding 9 m total width.

### Plaice box

In addition to the twelve mile zone a 'plaice box' was established in 1989 (Council Regulation EEC No. 4193/88), extending from 53° N to 57° N along the coasts of The Netherlands, Germany and Denmark and about 30 nm (55 km) wide, where the same restrictions are in force as in the twelve mile zone (in effect no beam trawling by vessels over 24 m overall length with engines over 221 kW). Initially the restriction applied only to the second and third quarter, in 1994 it was extended to the fourth quarter and since 1995 restrictions applied to the whole year (see Figure 6.3.4).

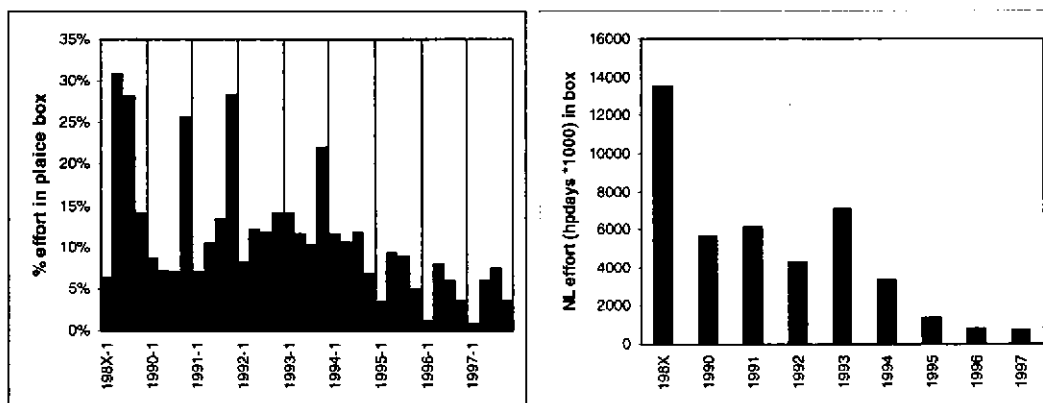
The plaice box was established in order to protect the nurseries and up-growing areas of young plaice, which should result in increased yields of plaice, possibly to an extent of 25%, compared to a situation without the box. An evaluation by an ad hoc study group in 1994 has confirmed the impression of the fishermen that the predicted effects have not been realised. In the summer of 1999 another ad-hoc study group will attempt to evaluate the Plaice box using total international effort data and data from dedicated surveys in 1996, 1998 and 1999.



Apart from the fact that the box was closed to the larger beamers only during half of the year, an important factor in not realising the expected effects has been the expansion of the fishery within the box. The protection offered by the box, and earlier by the twelve mile zone, has led to substantial investment in powerful 221 kW - 24 m vessels, mostly replacing smaller, obsolete boats on List I. In The Netherlands e.g., the number of boats with engines from 192 to 221 kW increased from 107 to 124 between the end of 1989 and the end of 1993, while the total fleet size decreased by nearly 100, from 573 to 474 boats.

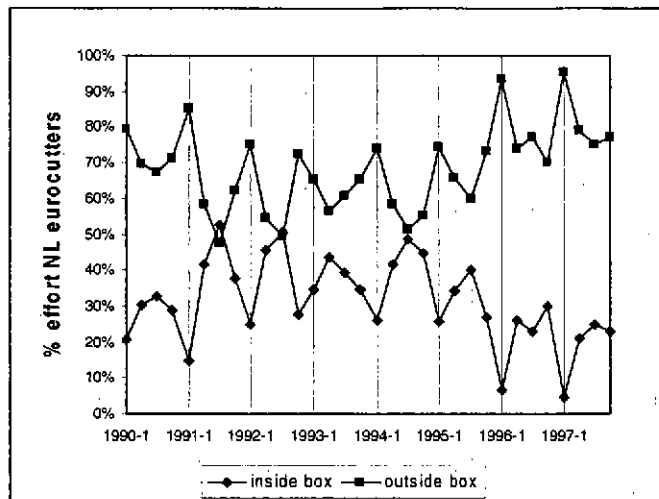
Dutch beam trawl effort before the closure of the plaice box was reconstructed assuming that the overall level of fishing effort for the Dutch fleet was more or less constant in the years prior to the establishment of the plaice box (Pastoors *et al.* 1998a). The distribution of effort was taken from the relative distribution of the mid 1970s which was then combined with the average effort level the late 1980s. Before 1989, the Dutch beam trawl effort in the plaice box was highest in the 2<sup>nd</sup> and 3<sup>rd</sup> quarter. After establishment of the beam trawl effort in the 2<sup>nd</sup> and 3<sup>rd</sup> quarter was greatly reduced, but after reopening the box on October 1<sup>st</sup>, the fleet of vessels larger than 300 hp moved into the box to exploit the rich fishing grounds. The large vessels then re-entered the box in the fourth quarter to exploit the rich fishing grounds. In 1994 the peak in the fourth quarter disappeared due to the 4th quarter closure (Figure 6.5.1 left).

One of the fundamental aims of establishing the plaice box, was to diminish the total level of trawling in the box area and thereby reduce the discard level of undersized plaice. Figure 9.5.1 (right) shows the reconstructed effort level (in horsepower days) for the total Dutch fleet fishing in the plaice box. Fishing effort in the first four years of the box diminished to around 40% of the pre-box level. In recent years, fishing effort in the box is only at around 6% of the pre-box levels.



**Figure 6.5.1 Left:** proportion of fishing effort (days at sea) by quarter of the Dutch beam trawl fleet fishing in the plaice box. The 198X bars were estimated from the mid 1970s effort distribution when around 30% of the fishing effort in the 2<sup>nd</sup> and 3<sup>rd</sup> quarter was exerted in the plaice box. After the establishment of the plaice box in 1989, the larger vessels were excluded from the box in those quarters. **Right:** Fishing effort (HP-days \* 1000) of the Dutch beam trawl fleet fishing in the plaice box. The 198X bar was estimated from the mid 1970s effort distribution (inside box-outside box) at the same overall level of fishing effort. Fishing effort in the first four years of the box diminished to around 40% of the pre-box level. In recent years, fishing effort in the box is only at around 6% of the pre-box levels.

Figure 6.5.2 shows the proportion of effort for the Eurocutter fleet segment which fished either in the plaice box or outside the plaice box. In the period 1990–1993 the proportion of Eurocutters inside the box increased, but since 1995 the proportion has decreased substantially. It can be concluded that although the box was closed for the larger vessels in recent years, the small vessels have not been able to profit from this closure.



**Figure 6.5.2** Proportion effort (days at sea) of the Dutch eurocutter fleet (225–300 hp) inside and outside the plaice-box.

### New technical measures

On 30 October 1998 the Council of Ministers accepted a new regulation of technical measures, in particular with a view to protecting juveniles (850/98), as a replacement for Regulation (EC) No 894/97. New elements in this regulation are that the minimum landings size of North Sea plaice is reduced from 27 cm to 22 cm and that the derogation to fish with mesh sizes of 80 mm is extended from 55° N to 56° N (latitudes east of 5° E). It is expected that these measures will affect the discarding behaviour of the various fleets. It was not known to the WG what the (biological or economic) basis for the new regulations are. An outline of a possible approach to evaluate the effects of reducing minimum landing size is presented in Section 2.4.

#### **6.5.1.3 Structural measures**

According to the Multi Annual Guidance Program (III), covering the period 1–1-'92 to 31–12-'96, benthic fishing fleets, i.e., beam trawling fleets, should reduce their engine power and gross tonnage by 15%. Demersal fleets should be reduced by 20%, for other fisheries no reductions are required. Of the required reductions, 45% may be realised by a (permanent) reduction of effort.

An evaluation of the MAGP III program for the Netherlands and Denmark showed that the Danish fleet was reduced by 29% and the Dutch fleet by 22% relative to GRT/GR (Frost *et al.* 1996). Dutch beam trawlers that were withdrawn under the MAGP program have in general been replaced. For both countries the reduction in number of vessels has been compensated by an increase in the number of fishing days by vessel. Therefore, the effect of the decommissioning program has been an increased economic viability due to reduced fixed costs for the whole fleet and more fishing days for the remaining vessels (Frost *et al.* 1996).

In December 1997 the European Commission agreed a MAGP IV program which was going to last 5 years. In this program the EU fleets are confronted with reductions in fishing effort of up to 30%. Member states are allowed to realise reductions by either capacity reductions and/or limitations on days at sea.

#### **6.5.1.4 Marketing measures**

A number of flatfish species are included in the common withdrawal price system, pursuant to Regulation (EEC) No 3759/92: plaice, megrim, dab, flounder and sole. The withdrawal price for 1995 is set at 83% of the guide price for all of those, except for megrim, where it is 80%.

Not all species have to be included in the withdrawal systems of the individual Producer Organisations. For example, the Dutch P.O.'s do not have a withdrawal price for sole, as it would serve no practical purpose.

The withdrawal prices for plaice are set at two levels: they are lower for the first four months of the year in order to discourage landing lean, low quality plaice during the spawning season. This could have some stock conservation effect as well, although in practice the effect is very limited.

## **6.5.2 National**

In most participating countries the North Sea flatfish fishery, particularly of plaice and sole, is subject to more or less specific national restrictions. These may concern access to the fishery or landing restrictions and are quite complicated in some instances.

### **6.5.2.1 Belgium**

The fishing capacity of the Belgian fleet is controlled by a vessel licensing system, which allows no increase in aggregate engine power and tonnage. A decommissioning scheme serves to stimulate capacity reduction as required by the MAGP. Vessels can be replaced by boats of the same horse power and gross tonnage, or licenses can be joined for new building up to a maximum tonnage of 385 GT, an engine power of 883 kW and a length of 38 m. The latter is not supported by the decommissioning scheme.

The fishery for plaice is not subject to any national restrictions. Individual quota's of North Sea sole are allocated on the basis of engine power; the allocations may be adapted during the year, according to the recorded catches. In 1993 e.g., after successive increases, vessels of 221 kW or less were eventually allowed to land 93 kg/kW and more powerful vessels half of that.

### **6.5.2.2 Denmark**

The fishing capacity in Denmark is in general limited by a licensing system. New capacity can only be commissioned after decommissioning of a similar old capacity. The capacity of individual vessels is expressed in six parameters: gross tonnage, length, width, depth, hold volume and engine power. Increases in any of these parameters require the consent of the Ministry of Fisheries. Conversion of vessels having an engine power over 368 kW to beam trawling is not allowed.

There is no direct regulation on the catch of plaice, as the quota cannot be caught. In fact the plaice fishery is stimulated through a kind of multi-species regulation: fishermen get extra cod quota if their plaice landings exceed 40% of the landing weight of cod.

Catches of sole are regulated by boat quota's per unit of time, varying during the year with the season. In 1994 for instance, sole catches were restricted to 300 kg per week, only as a by-catch of the plaice fishery, during the first quarter. In the second quarter the boat quota's varied between 600 and 1800 kg per two weeks, depending on boat length. The sole fishery was free during the third quarter, but quotas were reintroduced in October to a level of 100 kg per boat per week.

### **6.5.2.3 France**

France has no national regulations on the North Sea flatfish fishery.

### **6.5.2.4 Germany**

The fishery for plaice has no national regulation in Germany. Catches of sole however are restricted by boat quota per calendar month. In 1994 these quota's increased quarterly from 3000 kg/month in the first quarter to 12000 kg/month in the last quarter.

### **6.5.2.5 The Netherlands**

A complex system of national regulations has been introduced in The Netherlands in order to regulate the fisheries, in particular that for flatfish.

Since 1985 licenses on engine power restrict the capacity of the fleet. The licenses are transferable, but are reduced in size by 10% on transfer since 1994. At the start of 1995 a total of 565 licenses were 'active', i.e., had been issued to vessels, having an aggregate engine power of 429 MW.

The catches of plaice and sole are regulated basically with a system of individual transferable quotas. These were introduced in 1976, shortly after the inception of the NEAFC quota management. Originally the ITQ's were allocated on the basis of track records in the same reference years as the national quota allocation had been based upon: 1972-1974. The ITQ's are expressed in kg per year and annually adapted in proportion to the changes in the national quotas, so in effect the ITQ's are shares in the national quotas. Transfer of quotas is restricted in time and in relation to the extent of exhaustion of the relevant quota of both letter/seller and hirer/buyer.

From the start, the ITQ's proved very difficult to control and enforce. On one hand the obligation of first sale of landed fish through auctions had been dropped due to EC market regulations, so fishermen were free to land their fish wherever and whenever they wanted, and sell it to whoever they wanted. On the other hand it appeared to be legally virtually impossible to end the fishing activity of vessels that had exhausted their quotas (as they could still go fishing for other species). Another aspect of the problem is, that the Dutch Government is primarily responsible to enforce the national quotas and the ITQ's, as a tool of equitable allocation, are only of secondary concern.

Very strict rules on and control of landings were introduced during the eighties in order to curb illegal fishing and landing practices. Landing of fish is only allowed in fifteen ports, at designated quays, between certain hours. The inspection service has to be notified of the intention to land 8 hours beforehand (on trips lasting more than one day), and unloading of the catch is not allowed without the consent of the inspection service. All fish on board has to be unloaded in one uninterrupted action.

In order to support the enforcement of national quotas, restrictions on effort were introduced. From 1987 onwards, vessels having ITQ's of plaice and sole were allowed to spend a certain number of days at sea. Vessels having more quotas's than could be caught in the standard number of days were given a derogation. Since 1992 these derogations are based on a combination of ITQ's and engine power.

The basic number of days at sea is now reduced to 100 days but virtually all flatfish cutters have a derogation and are allowed to fish longer. It should be noted that the days at sea allocation system is implicitly based on the assumption that TAC's are set at *status quo* level. As a result allowed sea time does not vary with variations in TAC's, but in fact has gradually increased as a consequence of the reduced catchability of plaice.

In 1993 the management and control of ITQ's and days at sea was to a certain extent transferred to Producers Organisations (PO-groups). In the groups ITQ's and days at sea are pooled and can be transferred more freely than outside the groups, subject to the rules of the group. Group members are obliged to sell all fish landed through an auction. All flatfish fishermen have joined one of the eight groups that have been formed and signed a contract to abide by the rules set by the group.

#### **6.5.2.6 United Kingdom**

The UK operates a fairly complex system of licenses for the regulation of its so called 'pressure stock' fisheries. Both plaice and sole in the North Sea are considered to be 'pressure stocks' and for each a license is required. These licences are not only connected to the stock, but also connected to the capacity of the vessel expressed in a Figure compounded from tonnage and engine power. Replacement of capacity requires decommissioning of 120% of that capacity, but aggregation of licences is allowed.

A decommissioning scheme, involving funds to a total of £ 53 million over a period of five years, has been set up, in order to reach the MAGP targets. For the beam trawler fleet this involves a reduction by 15 percent to 17 600 G(R)T and 81.5 MW.

Management of quotas has been partly devolved to Producer Organisations. Shares of the national quotas are allocated to PO's based on the aggregate track records on the species concerned of their members. In 1994 27% of the North Sea plaice quota was allocated to PO's and 74% of the North Sea sole quota. The PO's can manage their quotas as they wish, e.g., by setting monthly quotas related to boat size, or by allocating annual quotas to individual boats. The fisheries of non-PO-members are managed by the government's 'non sector quotas'.

#### **6.5.3 Control**

Costs of management regulations (both on an EU and national scale) have so far not been included in reviews of this fishery. It should be noted that for a proper evaluation it is necessary that estimates of these costs are available, so that the effectiveness of management can be expressed.

## 6.6 Evaluation of current assessments

### 6.6.1 Biological assessment

Biological assessment reports are available since the early 1970. In these reports changes in assessment methodology are documented. Usually only assessment results of one single assessment method are presented in the reports.

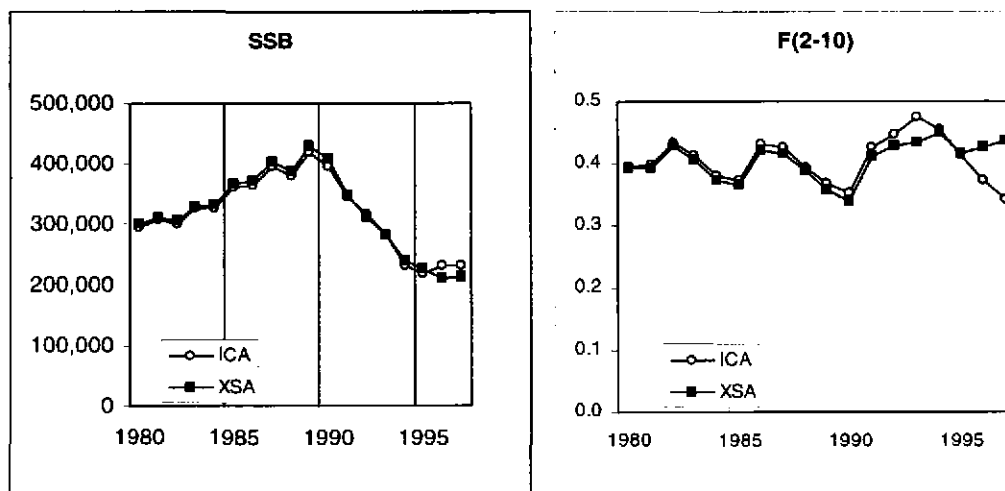
Although only single assessment methods are used in conventional assessment, here a comparison will be made between a number of different methods. A similar exercise was also performed at the ICES WG for Demersal Stocks (WGNSSK) in 1997 when ICA and XSA were compared.

Also, results from survey-only assessments will be presented (Cook 1997) and comments will be made on the estimation of new year classes.

#### ICA vs. XSA

The traditional XSA model specification was tested using an alternative assessment method and implementation (ICA; Patterson & Melvin 1996). A six-year selection pattern was estimated with a reference age 4 and selection at the final age of 0.6. Linear models were fitted for the tuning indices and each survey was given a total weight of 1. The comparison was made with the standard accepted XSA run (ICES 1999).

Results indicate that the SSB estimates in the final years are comparable with the XSA results although ICA seems to give a slightly more optimistic view on the rebuilding of the spawning stock in recent years. Fishing mortality is estimated rather differently, where ICA gives a fishing mortality on age 2 - 10 in 1997 of 0.34 against 0.43 of XSA. The diagnostic plots showed that the behaviour of the fleets is highly comparable between XSA and ICA with the Dutch beam trawl fleet giving a strong negative trend over time and the UK beam trawl fleet a positive trend over time.



**Figure 6.6.1** Comparison of biological assessment results using either XSA or ICA. For details of the procedures see text. ICA shows a marked decrease in fishing mortality over the last five years whereas XSA estimates an almost constant fishing mortality.

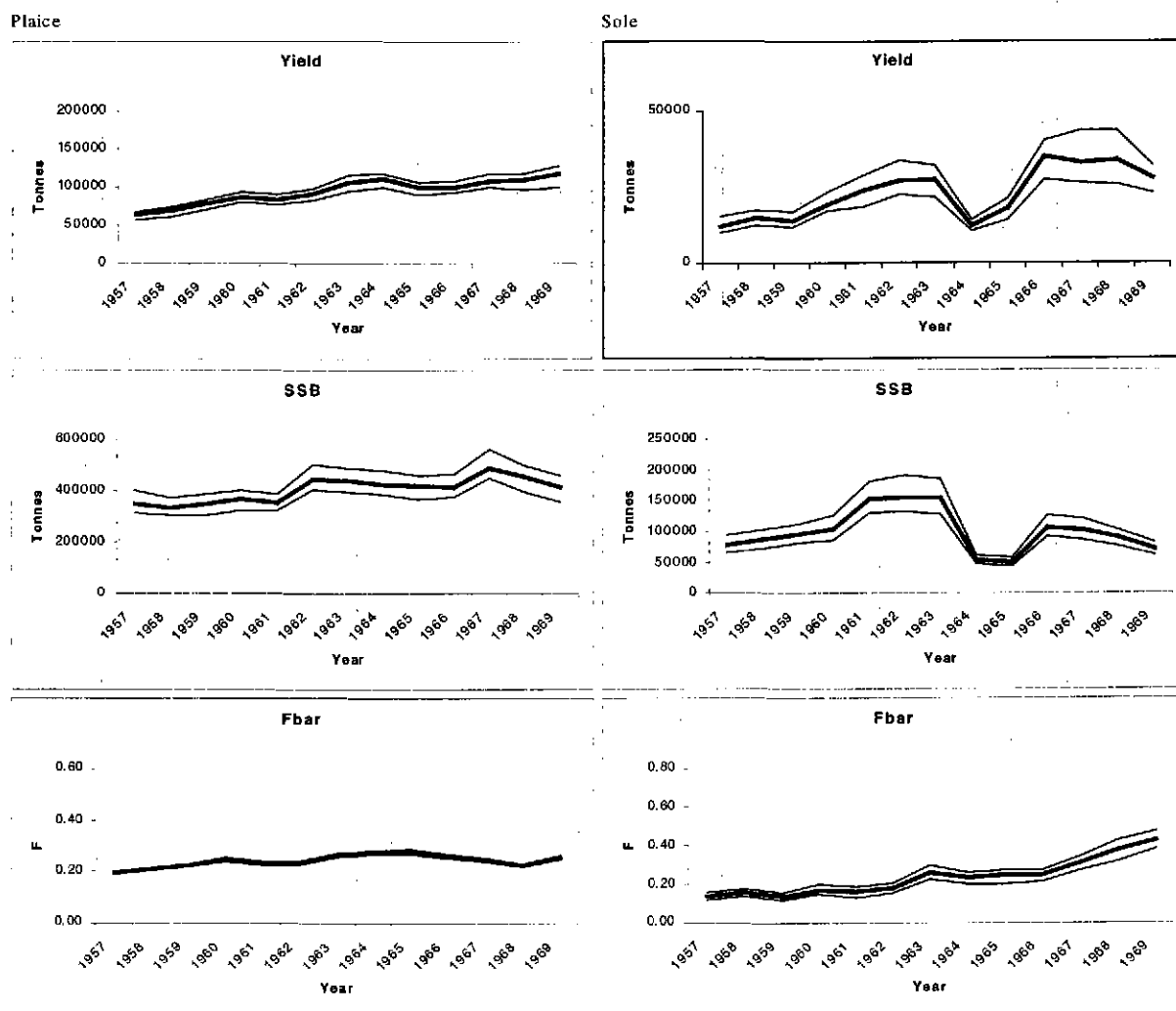
Cook (1997) presented assessments of several North Stocks based on survey data alone. In traditional biological assessments, commercial catch at age data is the basis of the analysis, which is then calibrated using research vessel surveys and commercial CPUE data. Because fishing mortalities are relatively high for the stocks considered, the conventional VPA equations converge rapidly and therefore most of the analysis will be determined by the transformation of the raw catch-at-age data into estimates of population and fishing mortality. Therefore, if there is a bias in the catch data due to mis-reporting this bias is going to translate into bias in estimated stock trends.

Using survey data alone, Cook (1997) concluded that there is poor agreement between the surveys and the ICES assessment. The survey suggests higher SSB values in the late 1980s with a steeper decline in recent years. Also, the surveys suggest much higher fishing mortality rates. The differences may be due to the absence of discard estimates in

the ICES assessment, which would lead to an underestimate of fishing mortality for the younger ages. Comparison of the exploitation patterns between the ICES assessment and the survey estimate, shows that they converge at age 5 with a wide discrepancy for ages 2–4.

For Sole, both survey data series support the trend in SSB as estimated in the ICES assessment of this stock. Also, there is close agreement in the estimated recruitment.

Estimation of uncertainty of the XSA assessment was possible using the Fishlab software. A bootstrapped XSA was run for plaice and sole separately (Figure 6.6.2). The results show that it is possible to evaluate the uncertainty in the assessment using bootstrapped assessment. The method itself should however be thoroughly evaluated before being applied to standard assessments.



**Figure 6.6.2** Historical yield, SSB and Fbar for plaice (left) and sole (right) with 95% CI from the combined Monte Carlo simulation bootstrap of XSA

To do:

- Evaluate appropriateness of tuning indices (e.g., trends in catchability in commercial CPUE series)
- Evaluate effects of misreporting
- Evaluate effects of discarding

### 6.6.1.1 Prognoses (short-term, medium term)

A method developed by Brander (1987) was used to evaluate short-term forecasts by comparing 'current year' forecasts and 'year ahead' forecasts with realised catches after correction for changes in exploitation patterns. Basically, three years are of interest in short term catch forecasts:

- Last data-year (N-1)
- Current year or assessment year (N)
- Forecast year (N+1).

Since the exploitation pattern for year N is unknown, the assumed catch for this year is also a forecast. Thus there are three estimates of the same quantity available.

The method developed by Brander consists of an efficient way to compare these three quantities after corrections for realised exploitation patterns. Exploitation rate is defined as:

$$E = \frac{F(1 - e^{-Z})}{Z}$$

Furthermore, fishing mortality is not calculated in the traditional normal average over a certain age range, but rather as an average weighted by the catch numbers at age ( $F_c$ , Shepherd 1983). F factors are calculated as the ratio between the exploitation rates in two years. Therefore, the F-factor for the year ahead projection is  $E(n)/E(n-2)$  and for the current year projection  $E(n)/E(n-1)$ .

The accuracy of predictions can then be calculated using the average deviation and the root mean square prediction error which is calculated like a coefficient of variation by dividing the root mean squared deviation between forecast and the actual landings by the mean of the actual landings. The average deviation is the mean of the positive and negative individual deviations between forecast and actual landings.

Results for the years 1990 – 1997 indicate that the prediction error for North Sea plaice are around 23% for current year predictions and 29% for year ahead predictions (table 6.6.1). Furthermore, there appears a consistent overestimation of catches with average positive deviations of 17% (current year) and 25% (year ahead).

North Sea sole predictions for the years 1990–1997 show prediction errors of 76% (year ahead) and 43% (current year) which less systematic trends: +7% (year ahead) and –3% for the current year (Table 6.6.2).

PLAICE																	
Year	Landings WG estimate	Cpred		Fc	F(1-e-z)/Z	E(n)/E(n-1)		E(n)/E(n-2)		Cadj		Difference		Squared difference		Percentual difference	
		Current year	Year ahead			current year	year ahead	current year	year ahead	current year	year ahead	current year	year ahead	current year	year ahead		
		n	n+1			n	n+1	n	n+1	n	n+1	n	n+1	n	n+1	n	n+1
1975	109			0.379	0.301												
1976	114			0.296	0.244												
1977	119	94?	86?	0.330	0.268	1.096	0.889	103	76	-16.2	-43.2	262.1	1865.2	-14%	-36%		
1978	114	117?	95?	0.339	0.274	1.025	1.123	120	107	6.0	-7.0	36.2	48.8	5%	-6%		
1979	145	112?	125?	0.450	0.346	1.261	1.292	141	161	-4.3	15.7	18.9	245.0	-3%	11%		
1980	140	139?	114?	0.493	0.372	1.074	1.355	149	154	9.0	14.0	81.9	197.4	6%	10%		
1981	140	148	130?	0.452	0.347	0.934	1.004	138	130	-1.5	-9.7	2.2	95.0	-1%	-7%		
1982	155	153	145	0.540	0.399	1.148	1.073	176	156	21.2	1.0	448.0	1.0	14%	1%		
1983	144	164	165	0.487	0.368	0.923	1.060	151	175	7.3	30.8	53.4	950.7	5%	21%		
1984	156	146	182	0.420	0.327	0.889	0.820	130	149	-26.4	-6.8	695.1	46.9	-17%	-4%		
1985	160	167	147	0.439	0.339	1.036	0.921	173	135	13.3	-24.4	175.6	595.5	8%	-15%		
1986	165		158	0.457	0.350	1.032	1.070		169		3.7		13.8		2%		
1987	154			0.500	0.376	1.074	1.108										
1988	154	201		0.419	0.327	0.870	0.934	175		20.4		415.7		13%			
1989	170	182	178	0.375	0.298	0.913	0.794	166	141	-3.7	-28.5	13.8	811.4	-2%	-17%		
1990	156	189	171	0.404	0.317	1.062	0.970	201	166	44.6	9.6	1985.9	91.7	29%	6%		
1991	148	164	169	0.434	0.336	1.061	1.127	174	191	26.0	42.5	676.6	1807.7	18%	29%		
1992	125	171	160	0.422	0.329	0.978	1.037	167	166	42.0	40.8	1761.1	1661.8	34%	33%		
1993	117	143	170	0.428	0.332	1.010	0.988	144	168	27.4	50.8	750.0	2581.5	23%	43%		
1994	110	114	147	0.456	0.350	1.053	1.064	120	156	9.6	46.0	92.1	2111.6	9%	42%		
1995	98	100	109	0.490	0.370	1.059	1.114	106	121	7.5	23.1	56.7	534.5	8%	24%		
1996	82	90	94	0.455	0.349	0.943	0.999	85	94	3.2	12.2	10.3	148.7	4%	15%		
1997	83	93	96	0.470	0.358	1.026	0.968	95	93	12.2	9.7	149.7	94.3	15%	12%		
1998		115	105														
1999			142														
Summary statistics 1990-1997														avg deviation			
mean	115									mean	685	1129	17%	25%			
stdev	28									RMSD	26	34					
										pred. err	23%	29%					

Table 6.6.1 North Sea plaice. Assessment of prediction error and average deviation in the short term projections.



SOLE

Year	Landings			F <sub>c</sub>	F(1-e-z)/Z	E(n)/E(n-1)		E(n)/E(n-2)		C <sub>adj</sub>		Difference		Squared difference		Percentual difference	
	WG estimate	Current year	Year ahead			current year	year ahead	current year	year ahead	current year	year ahead	current year	year ahead	current year	year ahead	current year	year ahead
	n-1	n	n+1			n	n+1	n	n+1	n	n+1	n	n+1	n	n+1	n	n+1
1975	20.8			0.468	0.357												
1976	17.3			0.458	0.351												
1977	18.0			0.437	0.338	0.962	0.947	103	76	-85.0	-58.0	7224.5	3363.7	-472%	-322%		
1978	20.3			0.460	0.352	1.042	1.003	120	107	-99.7	-86.7	9944.1	7520.4	-492%	-428%		
1979	22.6			0.510	0.382	1.084	1.130	141	161	-118.4	-138.4	14019.0	19155.1	-524%	-612%		
1980	15.8			0.484	0.367	0.961	1.041	149	154	-133.2	-138.2	17740.4	19097.3	-843%	-874%		
1981	15.4	14.1		0.458	0.351	0.957	0.919	13	130	2.0	-114.6	3.8	13132.5	13%	-744%		
1982	21.6	20.4	20.8	0.514	0.384	1.093	1.047	22	22	-0.7	-0.2	0.5	0.0	-3%	-1%		
1983	24.9	24.2	20.3	0.490	0.370	0.964	1.054	23	21	1.6	3.6	2.5	12.7	6%	14%		
1984	26.8	23.0	23.3	0.562	0.411	1.111	1.071	26	25	1.3	1.9	1.7	3.4	5%	7%		
1985	24.2		22.0	0.571	0.416	1.012	1.124		25	24.2	-0.5	588.0	0.2	100%	-2%		
1986	18.2			0.546	0.402	0.967	0.978						0.0				
1987	17.4			0.443	0.342	0.851	0.823										
1988	21.6	17.5	18.0	0.533	0.395	1.154	0.982	20		1.4		2.0		6%			
1989	21.8	24.0	23.3	0.468	0.357	0.904	1.043	22	24	0.1	-2.5	0.0	6.3	0%	-11%		
1990	35.1	38.0	29.0	0.426	0.331	0.928	0.839	35	24	-0.1	10.8	0.0	116.4	0%	31%		
1991	33.5	32.0	31.0	0.476	0.362	1.093	1.014	35	31	-1.5	2.1	2.1	4.4	-4%	6%		
1992	29.3	32.4	25.0	0.418	0.326	0.901	0.985	29	25	0.1	4.7	0.0	22.3	0%	16%		
1993	31.5	27.5	28.9	0.475	0.361	1.109	0.999	30	29	1.0	2.6	1.0	6.8	3%	8%		
1994	33.0	36.0	30.2	0.496	0.374	1.034	1.146	37	35	-4.2	-1.6	17.7	2.7	-13%	-5%		
1995	30.5	32.1	27.0	0.457	0.350	0.937	0.969	30	26	0.4	4.3	0.2	18.6	1%	14%		
1996	22.7	24.0	24.7	0.586	0.424	1.211	1.135	29	28	-6.4	-5.4	41.2	28.9	-28%	-24%		
1997	15.0	20.1	17.4	0.478	0.363	0.856	1.037	17	18	-2.2	-3.1	5.0	9.4	-15%	-20%		
1998		20.4	23.0														
1999			24.4														
Summary statistics 1990-1997																	
mean	29									mean	8	26		avg deviation			
stdev	7									RMSD	3	5		-7%	3%		
										VAR	211	1416					
										pred. err	43%	76%					

Table 6.6.2 North Sea sole. Assessment of prediction error and average deviation in the short term projections.

### **6.6.1.2 Biological reference points**

The occurrence of technical interactions have implication for the estimation and use of reference points. As seen in section 4.2 management may cause unreported catches to be taken in a mixed fishery resulting in biased perception of spawning stock biomass, recruitment and fishing mortality. Since most common reference points are based on these quantities they may also be biased. This is distinct from the more taxing task of choosing appropriate reference points and employing them to manage stocks in a mixed fishery.

### **6.6.1.3 Advice**

A compilation of the ACFM advice and the resulting TAC's and catches is presented in Table 6.6.3 (plaice) and 6.6.4 (sole). The compilation is largely derived from Daan (1997) who noted that it was not straightforward to summarise the information as it is presented in the tables. It seems clear that ACFM has changed motives for choosing particular F's as the basis for the recommended TAC's over time. Two main periods can be distinguished:

- Up to 1987, the stated CFP objective of 'rational' exploitation appears to have been taken literally to mean fishing at  $F_{max}$
- After 1987 the motivation for reducing F is often given in terms of maintaining spawning biomass above a particular minimum level (e.g., MBAL). If stock were above this level, a range of catch options was presented without recommending a specific level.

### **6.6.2 Economic assessment**

No progress was made on incorporating economic assessment methodologies within the comprehensive evaluation of the flatfish fishery.

To be done:

- Evaluation of economic assessment presented to STECF (STECF 1998).
- Assimilation of economic data on costs, investments, prices and revenues from different countries.

### **6.6.3 Management assessment**

To be done:

- Specification of cost of fisheries assessment, management and enforcement
- List number of available management scenario's.

**North Sea plaice**

Year	Recomm. TAC	Agreed TAC	Official catch	Estim. Catch	Estim SSB Basis in 1998 ass.	Motivation	Comments
1977	71		119	119	331		
1978	115		114	114	325	sq F	original advice 95,000 t; revised by ACFM in 1978
1979	120		145	145	313	sq F	
1980	112		140	140	300	sq F	
1981	105	105	140	140	311	20% red. in F	
1982	(70)	140	98	155	304	Red. to Fmax	increase SSB
1983	(181)	164	101	144	329	Within SBL	implicit TAC advice options: 80,000-181,000
1984	(150)	182	116	156	332	Within SBL	options: 80,000-182,000
1985	(130)	200	146	160	366	Within SBL	options: 69,000-147,000
1986	<140	180	128	165	370	Red. below F84	reduction towards Fmax
1987	120	150	131	154	402	Red. below F84	advice: improve expl. pattern
1988	<150	175	138	154	386	Reduce F	high unreported landings! implement plaice box
1989	(175)	185	152	170	430	Reduce F	advice on plaice box reinforced
1990	171	180	155	156	408	sq F	
1991	(169)	175	143	147	348	sq F	F at historic high; SSB well above MBAL
1992	(160)	175	123	125	311	sq F	options: 142,000-169,000. adv. on mixed fishery
1993	(170)	175	115	115	282	sq F	options: 135,000-160,000. adv on mixed fishery
1994	(147)	165	110	111	239	sq F	options: 142,000-170,000. implicit TAC advice
1995	<87	115	97	98	227	Reduce F min. 20%	options: 123,000-147,000. implicit TAC advice
1996	61	81	80	82	210	Reduce F with 40%	assessment changed sign since last year
1997	80	91	82	83	212	Reduce F with 20%	
1998	82	87			274	Fish at F=0.3	Agreement with Norway
1999	106	102				Fish at F=0.3	Agreement with Norway

**Table 6.6.3 North Sea plaice.** ACFM management recommendations, agreed TAC's, official catch, ACFM estimated catch and SSB as calculated in the most recent WGNSSK assessment (ICES 1999). Basis for the advice, motivation and comments. Source: Daan 1997 and ACFM 1999.

**North Sea sole**

Year	Recomm. TAC	Agreed TAC	Official catch	Estim. Catch	Estim SSB Basis in 1998 ass.		Motivation	Comments
1977	7	13	18	18	36			
1978	8	10	20	20	39			Advice by Liaison Committee 1977
1979	13	15	23	23	46	50% reduction in F	To increase SSB	
1980	15	15	16	16	36	Precaut. Approach	Uncertainty in assessment	High natural mort. Due to severe winter
1981	15	15	15	15	25	Precaut. Approach	Uncertainty in assessment	Effect of change legal min. mesh size to 80 mm uncertain
1982	15	21	22	22	35	Precaut. Approach	Uncertainty in assessment	
1983	(15)	20	20	25	42	Reduce to Fmax	To incr. Long term yield and SSB	Options: 9,000-20,000; implicit TAC advice
1984	(14)	20	20	27	45	Reduce to Fmax	F above any BRP	Options: 8,000-22,000; implicit TAC advice
1985	(15)	22	20	24	43	Reduce to Fmax	F above any BRP	Options: 8,000-22,000; implicit TAC advice
1986	(12)	20	13	18	36	Reduce to Fmax	F above any BRP	Options: 6,000-21,000; implicit TAC advice
1987	<11	14	14	17	31	Reduce F by 40%	Rebuild the stock	Advice to close dir. Trawl fishery during spawning time; enforce legal min. mesh size
1988	<11	14	13	22	42	Reduce F	Rebuild the stock above 50,000 t.	Attention to high unreported landings; advice on additional technical measures
1989	14	14	14	22	36	Sign. Reduction F	Allow strong 1987 yc to rebuild SSB	Enforce legal min. mesh size
1990	25	25	26	35	91	Reduce F by 20%	First step in nec. Red. In F by 40%	Enforce legal min. mesh size
1991	27	27	28	34	78	Reduce F by 20%	Ensure SSB will remain above MBAL	TAC regime has not resulted in intended reduction in F. Real effort reduction required
1992	21	25	26	29	78	Reduce F by 20%	First step in nec. Red. In F by 40% to bring SSB above MBAL	Advice on mixed fishery
1993	29	32	30	31	56	F status quo	No long-term gains in increased F	
1994	31	32	31	33	76	F status quo	No long-term gains in increased F	
1995	28	28	29	30	62	F status quo	No long-term gains in increased F. link to plaice	
1996	23	23	21	23	40	F status quo	Mixed fishery, link plaice advice into account	
1997	15	18	14	15	33	80% of F(95)		
1998	18	19			26	75% of F(96)		
1999	20					Fish at F=0.4	Reduce F below Fpa	Introduction of precautionary approach

**Table 6.6.4 North Sea sole.** ACFM management recommendations, agreed TAC's, official catch, ACFM estimated catch and SSB as calculated in the most recent WGNSSK assessment (ICES 1999). Basis for the advice, motivation and comments. Source: Daan (1997) and ACFM (1999).

## 6.7 Scenario modeling

### 6.7.1 Introduction

A simulation model (Figure 6.7.1) of the North Sea plaice and sole fishery has been developed to allow the evaluation of management strategies. This comprises an operating model from which pseudo-data are sampled and then used to assess stock status (using the current single species assessment) and set catches using a simple harvest control rule.

The operating model has both an historical component covering the historical period and a future component. Historical components may be conditioned on the observed time series of fishery data and are conditional upon the adopted assessment method. In this way they are consistent with the latest North Sea plaice stock assessment as undertaken by the ICES Working Group on the Assessment of Demersal Stocks in the North Sea and Skagerrak (ICES 1999). Alternatively a population model with known models and parameters could be used. The future component is essentially a forward VPA where fishing mortality is generated by the catches of fleets with given selection patterns.

The partial fishing mortality of a fleet is a function of its effort and selectivity matrix. Effort is independent of stock and if a fleet is targeting one stock then by-catch will be determined by the selection pattern of the non-target species. If the quota of one species is exhausted but a fleet continues to fish a by-catch will be taken.

Uncertainty in commercial catch, research survey abundance indices, stock weights, natural mortality, maturity and recruitment, and the sampling of these for use within the management procedure are simulated within the operating model.

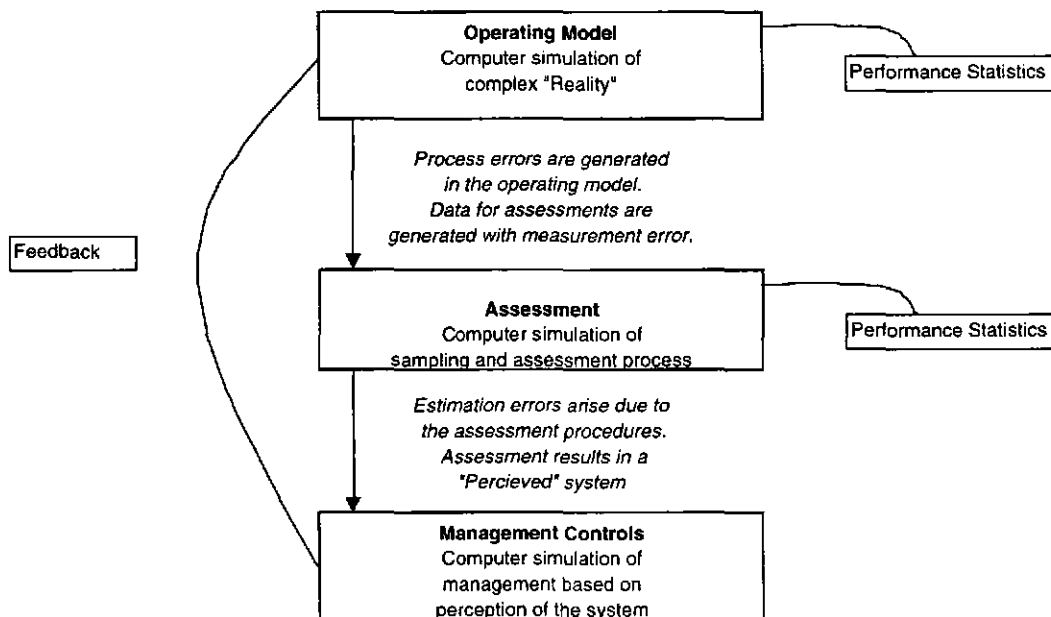


Figure 6.7.1. Simulation model structure (after ICES 1997x,z)

The model used for the simulation was written in Visual Basic for Applications (VBA), calling routines from the Fishlab library. This allows for modular programming and helps to guarantee the integrity of component algorithms.

### 6.7.2 Management objectives

General management for these stocks is outlined in article 2, par.1 of the CFP Council Regulation (No. 3760/92):

...the general objectives shall be to protect and conserve available and accessible living marine aquatic resources, and to provide for rational and responsible exploitation on a sustainable basis, in appropriate economic and social

conditions for the sector, taking account of its implications for the marine ecosystem, and in particular taking account of the needs of both producers and consumers.

Clearly the above objective is not specific enough to derive comprehensive performance criteria for management evaluation as intended in this section.

The EC further agreed to the FAO code of conduct for responsible fishing with regards to the precautionary approach (FAO, 1995).

Specific management objectives for North Sea plaice are agreed in the Fisheries Consultation between the EC and Norway:

*In light of the current stock situation for plaice, the Parties (EU and Norway) agreed to continue to apply a multi-annual management strategy to achieve the objective of reaching a level of spawning stock biomass defined by ICES as the minimum biologically acceptable level (MBAL). For the year 2000, the Parties agreed to adopt a TAC consistent with a fishing mortality rate of 0.3 unless future scientific advice requires modification of this agreement. The Parties agreed that, to provide increased security and greater potential yield, the stock needs to be rebuilt to progressively higher levels (Agreed record of conclusions of Fisheries Consultations between the European Community and Norway for 1999. Brussels, 1 December 1998).*

### 6.7.3 Underlying system model

#### 6.7.3.1 Natural mortality

A natural mortality rate of 0.1 is assumed conventionally for North Sea plaice across all the age-classes in the stock (ICES 1999). There is, however, a great deal of uncertainty about the value of natural mortality and VPA results are sensitive to the natural mortality value.

Multi-species virtual population analysis (MSVPA; Gislason & Helgason, 1985; Sparre, 1991) partitions natural mortality into two components, M1 and M2. M2 is the predation mortality due to all predators included in the MSVPA model; M1 is the residual natural mortality from all other causes such as diseases and predation by predators not included in the model. Predation mortality estimates (M2) are available (ICES 1997a), although these are significantly lower than mortality estimates due to other non-fishing sources of mortality.

Uncertainty in natural mortality was simulated in the operating model by a uniform random variable with a mean of 0.1 and a coefficient of variation (CV) of 20%.

#### 6.7.3.2 Maturity

Maturity at age is taken as a step function representing the difference in maturation of males and females, and is assumed constant over time (ICES 1999); although maturity has been studied in particular case studies and is known to have varied over time.

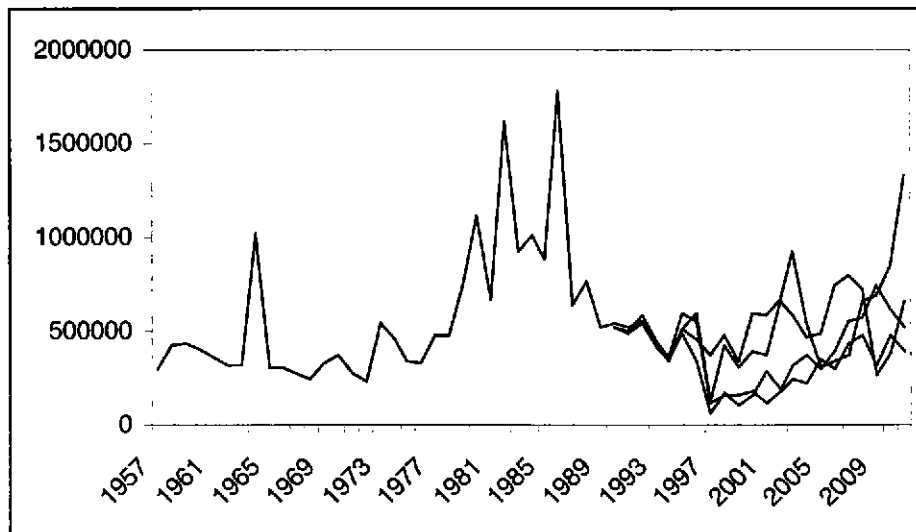
#### 6.7.3.3 Recruitment

**Plaice.** The stock-recruitment (S-R) relationship is generally analysed from the information obtained from VPA calculations. Inter-annual variability in recruitment is rather low, except for the occurrence of three exceptionally strong year classes in 1963, 1981 and 1985. The S-R data do not, however, yield information on the appropriate functional form to adopt for the relationship between stock and recruitment. Non-parametric models fitted to recruitment over both the time and spawning stock biomass (SSB) domains reveal that time is perhaps a better predictor of recruitment than SSB. Recruitment can be modelled as a stationary uni-variate time series apparently independent of the perceived SSB (c.f. Schnute & Richards, 1995; ICES 1998b) with the following model.

Let  $y_t$  denote the natural logarithm of recruitment at time  $t$ ;  $y_{t-1}$  denote the natural logarithm of recruitment at time  $t-1$ ;  $y_{t-2}$  denote the natural logarithm of recruitment at time  $t-2$  and  $\epsilon_t, \epsilon_{t-1}$  denote purely random disturbance terms with a mean of zero and a variance of  $\sigma^2$ . Then recruitment may be modelled by:

$$y_t - \mu = \gamma_1 (y_{t-1} - \mu) + \gamma_2 (y_{t-2} - \mu) + \epsilon_t + \rho \epsilon_{t-1}$$

with the parameters ( $\mu$ ,  $\gamma_1$ ,  $\gamma_2$ ,  $\rho$ ,  $\sigma$ ) estimated by maximum likelihood (Ansley, 1979). Application of the model to the time series of recruitment presented in the latest North Sea plaice stock assessment (ICES 1999) yields parameter estimates:  $\hat{\mu} = 12.94$ ,  $\hat{\gamma}_1 = 0.64$ ,  $\hat{\gamma}_2 = 0.10$ ,  $\hat{\rho} = 0.42$ ,  $\hat{\sigma}^2 = 0.12$ , and the residual in the last year 1997 is -0.024. The fit of the time series model is shown in Figure 4.\*; together with selected series of simulated recruitments generated.



**Figure 6.7.2.** North Sea plaice. Time series plot of recruitment (vertical axes) versus year (horizontal axes); together with 4 realisations of simulated recruitments.

**Sole.** A comparable analysis of sole recruitment was not performed and was therefore modelled as a log-normal random deviate (mean 100.000, cv 80%)

#### 6.7.3.4 Weights-at-age

Catch and stock weights-at-age (kg) are those presented in ICES (1999). The mean weights-at-age in the catch were estimated from market samples taken throughout the year but were adjusted to correct for the sum of products (SOP) by country. The stock weights-at-age were derived from the first quarter catch weights.

In the Monte Carlo simulations, the catch and stock weights-at-age as used by the ICES stock assessment working group were perturbed by the addition of randomly drawn residuals from a local regression smoother (Cleveland, 1979, 1981). The effect of the covariance between the weights-at-age within a year was simulated by randomly selecting a residual vector for a particular year.

#### 6.7.3.5 Catch-at-age and CPUE

The catch-at-age composition of the landings is that presented in ICES (1999). The catch-at-age table has been revised slightly for 1996. These data were used to estimate the selection pattern of the commercial fleets in the simulations. Uncertainty in catch-at-age for the commercial fleets was modelled as a gamma random variable using CVs derived for the English market sampling of age-length-keys and age-length-distributions. The Gamma distribution was used rather than the log-normal since the variance appears to be a linear function of the mean (Figure 6.7.3).

Uncertainty in non-commercial catch-at-age was derived from the bootstrapping of the assessment procedure, XSA (see below). In the Monte Carlo simulations, XSA is refitted each time to estimate expected catchability values and the associated residual. The effect of the covariance between the CPUE indices within a year, and the weights-at-age, was simulated by selecting the residual vector for the same year and the same fleet.

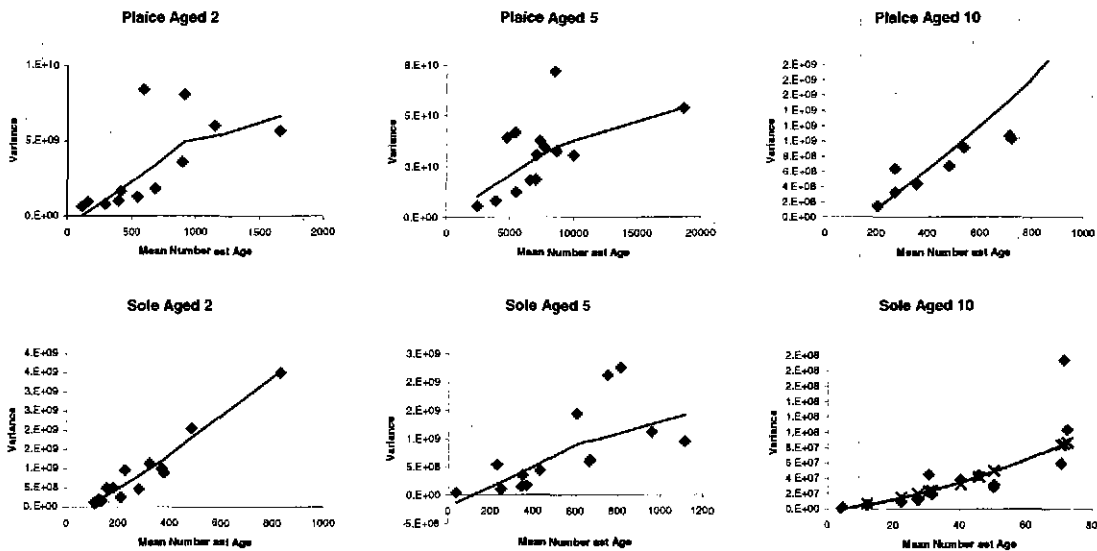


Figure 6.7.3 North Sea plaice. Analysis of English market sampling. X-axis: mean number at age. Y-axis: variance. The fitted line is a LOWESS smoother. Source: CEFAS.

#### 6.7.4 Assessment method

Current assessment practice for flatfish in the ICES Working Group on the Assessment of Demersal Stocks in the North Sea and Skagerrak (ICES 1999) is based on performing a historical assessment using XSA, calculating reference points using the PA Software before performing a medium term projection.

Extended Survivors Analysis, XSA (Darby & Flatman, 1994), is the *de facto* method used for the analysis of commercial catch-at-age data within the ICES area. It is an implementation of sequential population analysis (Doubleday, 1981) that relies upon either cohort analysis or virtual population analysis, VPA, to re-create a stock's historical population structure from the catch-at-age matrix. The VPA estimates of numbers-at-age are conditional upon the time series of values of the natural mortality and the population numbers alive at the oldest age in each cohort - the survivors. XSA is an iterative method of estimating the survivors using the relationship between catch-per-unit-effort, CPUE, and the VPA estimates of population abundance-at-age; the  $N$ 's. The algorithm is initiated with an initial value of the number of survivors and fleet catchabilities-at-age are estimated using the estimates of  $N$  and the indices of CPUE. Predictions of  $N$  by fleet, by age and by year are then made for each observed value of CPUE. The predicted values are projected forward to the final age in each cohort using the estimated fishing mortalities and the assumed natural mortalities. A weighted average of the survivors provides new estimates of the terminal  $N$ 's and the XSA algorithm iterates until successive estimates of the fishing mortality converge.

The XSA algorithm used in these simulations is available in the Fishlab Library.

#### 6.7.5 Harvest control rules

Two simple control rules were defined

Rule 1

$$\begin{aligned} &\text{if } B_t \leq B_{pa} \text{ then} \\ &\quad F_t = F_{target} (B_t - B_{lim}) / (B_{pa} - B_{lim}) \\ &\text{else} \\ &\quad F_t = F_{target} \end{aligned}$$

Rule 2

$$\begin{aligned} &\text{if } B_t \leq B_{pa} \text{ then} \\ &\quad F_t = (F_{status\ quo} + F_{target} (B_t - B_{lim}) / (B_{pa} - B_{lim})) / 2.0 \\ &\text{else} \\ &\quad F_t = (F_{status\ quo} + F_{target}) / 2.0 \end{aligned}$$



where  $B_t$  denotes the spawning stock biomass and  $F_t$  the fishing mortality at the current epoch  $t$ ;  $F_{\text{status quo}}$  denotes the fishing mortality in the previous epoch;  $F_{\text{target}}$  is 0.3 and 0.4 for plaice and sole respectively.

Rule 1 is a simplification of current advice and Rule 2 is an attempted to stabilise effort by not allowing  $F$  to change too rapidly.

The ICES Study Group on the Precautionary Approach to Fisheries Management (ICES 1998a) suggested suitable candidate limit (denoted  $F_{\text{lim}}$  and  $B_{\text{lim}}$ ) and precautionary (denoted  $F_{\text{pa}}$  and  $B_{\text{pa}}$ ) reference points for fishing mortality and spawning stock biomass, respectively. However, the values have recently been revised by the ICES Advisory Committee on Fishery Management that met in October 1998.  $B_{\text{lim}}$  is set equal to 210,000 and 25,000 tonnes,  $B_{\text{pa}}$  is proposed as 300,000 and 35,000 tonnes and  $F_{\text{pa}}$  as 0.3 and 0.4 for plaice and sole respectively.

No harvest control rule was implemented that closely linked fishing effort in the mixed fishery, mainly because of time constraints, but it is recognised that this would be a useful and necessary extension of the scenario model.

### 6.7.6 Results

Results of the simulations concerning the technical interactions are presented in Section 4.2 They can be briefly summarised as follows:

The single species experiments with perfect assessment are essentially an improvement on the current historical assessment and projection used by the working group. As the combined Monte Carlo simulation and bootstrap of XSA provides better estimates of uncertainty in stock status and carries these through into the calculation of reference points and the projection.

When the current population status is assessed using XSA the mean fishing mortality in the true population is reduced for longer resulting in lower catches in the short term and higher SSB in the long term compared to when the working group has perfect knowledge of stock status. This is because the working group underestimates  $N$  and over estimates  $F$ , probably due to the assessments inability to track fast changes in the stock.

If the historical stock status in a mixed fishery situation is assessed using VPA the occurrence of mis-reporting in total catch causes the assessment to quickly go wrong. This is due to the feedback between the bias in the assessment and misreporting. This is exacerbated by the form of harvest control rule 1 which enables a fishery to continue despite the fact that the TAC for one of the target species is set to zero. Harvest control rule 2 implemented a simple form of effort control, which was considered more realistic as changes in fishing mortality are likely to be more gradual than implemented harvest control rule 1. The result is that in changes in  $F$  and SSB were more gradual and the ability of the working group to assess the stock improved.

Although the scenarios examined were relatively simple they illustrate the importance of technical interactions and the link between management and our ability to assess the stock. More work needs to be done on modelling the response of fleet dynamics and hence the response of fishing mortality to management in mixed fisheries.

### 6.7.7 Further work

Although the work presented here is a step forwards towards a comprehensive evaluation of the North Sea flatfish fishery, there is still a substantial amount of work left to be done. This includes:

- Evaluation of by-catch in the flatfish fisheries; overview of turbot, brill, dab, cod, haddock and whiting catches in the beam trawl fishery.
- Historical overview of stocks
- Evaluate recent economic report (STECF 1998).
- Evaluate costs.
- Present overview of import / export by country.
- Evaluate revenues by fleet segments.
- Evaluate processing sector
- Evaluate markets for flatfish
- Proceed on scenario modelling including different Harvest Control Strategies among which a full effort control based harvesting strategy.

The WG recommends to continue the comprehensive evaluation of this fishery intersessionally and to report final results to the next COMFIE meeting. It is suggested to have a final report available before the start of the meeting so that the report can be reviewed beforehand.

## **6.8 Summary and conclusions**

This comprehensive fishery evaluation started with an outline of the general question to be addressed in a comprehensive assessment:

What do detailed biological, environmental, economical, managerial and behavioral data and analysis contribute to our understanding of the status and prospects of the North Sea flatfish fishery

Further, it was put forward that an evaluation would be undertaken of the effects of including technical interactions in the operation of general management rules according to the precautionary approach.

It is shown that the comprehensive assessment of the North Sea flatfish fishery – although far from finished – provides important additional insights into the status and management of the stocks considered here:

- Long term management intention to reduce fishing capacity effort in the MAGP programs has not resulted in an observable reduction in effort although the capacity was reduced. One of the reasons was that the remaining vessels were allowed to fish more which compensated for the reduction in capacity. It is recognised that reductions in single species TAC's in a mixed fishery need to be coordinated with corresponding effort measures, if discards are to be avoided and single species TAC's are not taken as limits.
- The beam trawl fishery generates high discard rates of juvenile fish, mainly plaice and dab which are currently not accounted for in the assessment procedures. Taking into account the mixed nature of this fishery shows that the perception of the system dynamics is greatly influenced by the balance between the two species and their catches.
- Technical interactions between fleets, gears used and target species play a predominant role causing among other things discarding of fish if quota are overshot
- Analysis of international vessel trip data is essential in understanding the dynamics of the various fishing fleets and their response to management and for the construction of reliable CPUE series.
- Need to assimilate sensitive national data into an international database for research purposes
- Price elasticity for the dominant prey species enables to compensate to a large extent possible reductions in quota
- With the complex management system with both EU regulations and national regulations it is difficult to predict likely consequences of global TAC advice, since the catching opportunities are modified in different ways for different fleets.

## **6.9 Recommendations**

The COMFIE WG recommends that the ICES demersal WG (WGNSSK) takes up the following issues in the assessment of North Sea plaice and sole:

- Further evaluate the implication of multi-species assessments of North Sea flatfish fisheries
- Evaluate the implications of including 0-group estimates in the VPA assessments rather than using RCT3 for recruitment estimation
- Further evaluate the implications of not including discards in the traditional VPA assessments
- Further explore model-misspecification using different assessment models e.g., selection pattern
- Review the work of the COMFIE report as concerning the comprehensive fishery evaluation.
- Explore the possibilities of incorporating economic analysis in the total assessment procedures.

## 7 COMPREHENSIVE ASSESSMENT OF NORWEGIAN SPRING SPAWNING HERRING

### 7.1 Introduction

The Norwegian spring spawning herring was a case study also at the previous COMFIE meeting (ICES 1997x) and the biology and stock history was then dealt with in some detail.

The subsequent assessment work with this stock (ICES 1998ö) has revealed two problems of special importance:

- The tuning procedure is inherently unstable. Different plausible assumptions may yield very different perceived stock sizes in the terminal year.
- The stock is now declining because of poor year classes being recruited to the fishery. There is need for pre-agreed management measures. Management rules proposed for this stock lack a basis in simulation experiments.

This meeting cannot give a final solution to these problems. However, the following describes an attempt to elucidate the problems by tuning and simulation experiments as a basis for further work.

### 7.2 Tuning

#### **The effect of different distributional assumptions for the acoustic survey series used in the tuning**

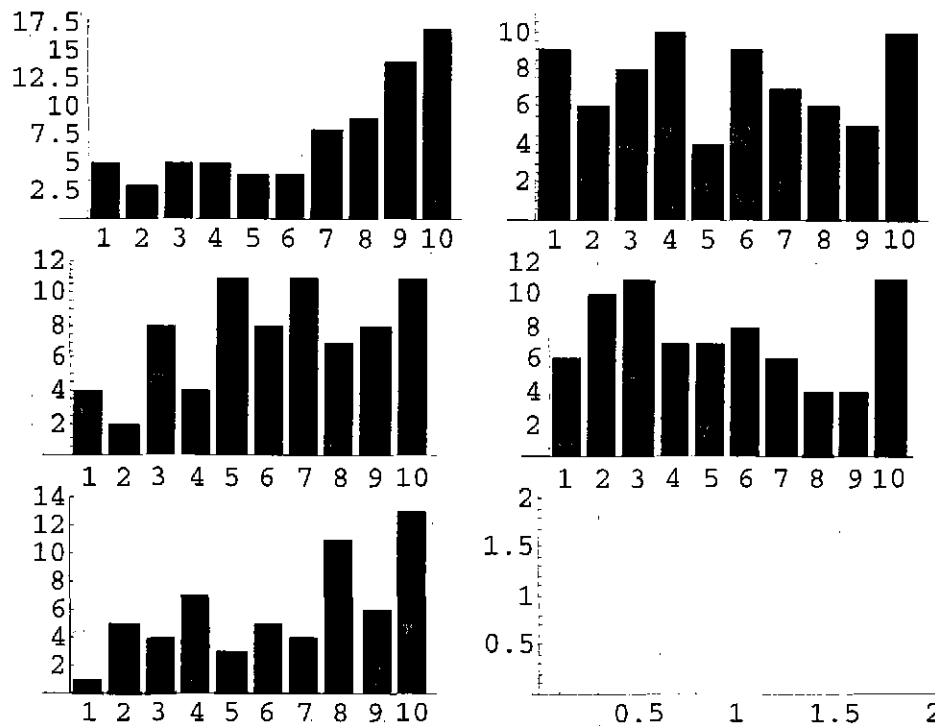
The VPA for the Norwegian spring spawning herring is tuned on four trawl-acoustic series: December in the overwintering area, January in the overwintering area, February-March on the spawning grounds and in the feeding areas in the Norwegian Sea in May. In addition, data from a tagging experiment that started in 1987 are used, where the probability function for tag returns is assumed Poisson.

The tuning is done by maximising the total likelihood with 8 parameters: the F-value in the final year of the 1983 year class, the F-value in the final year of the 1992 year class, 4 survey calibration parameters, survival during the first year of the tagged fish and one variance/CV parameter for the survey distributions. The other F-values in the final year were assumed constant for age 8 in 1998 and older and linearly interpolated down to the F-value of the 1992 year class.

For the surveys different distributional assumptions have been tried by the Northern Pelagic and Blue Whiting Working Group (ICES 1998:ö): normal, log-normal and gamma, where in the latter the variance was an extra parameter to be estimated. Here, the properties of the different distributional assumptions will be investigated somewhat further. When using the gamma distribution the PDF will be different for each of the year-age points in each survey because the expectation value – which is taken from the backcalculated stock – is different in each case. Therefore there will be no pool of residuals from which one could calculate some characteristic (mean, variance, kurtosis) and compare with the assumed distribution in a test.

Numbers following a PDF can be generated by drawing one number  $y$  at random from a uniform distribution on the interval  $[0,1]$  and calculating the  $x$ -value that corresponds to this  $y$ -value in the corresponding CDF. Conversely, for a given PDF the calculated  $y$ -values in the associated CDF should be uniformly distributed. If one has many different PDFs, draw one number from each and calculate the corresponding CDF values, then these numbers should be uniformly distributed. This also gives a method for comparing across families of distributions.

The expectation values for all age-year points from all surveys were divided in two halves. Figure 7.2.1 shows the distribution of CDF values for the gamma distribution, the normal distribution and the log normal distribution (from top to bottom) using constant variance (left) and constant CV (right). In each panel the values corresponding to high expectation values are shown in red (the lower part of each vertical bar) and values corresponding to low expectation values are shown in blue (the upper part of each vertical bar). The height of the bars is the number of points in the decile given on the  $x$ -axis, 74 points in total.



**Figure 7.2.1.** Distribution of cumulative density frequency values, explanations in the text (lower right panel is a dummy plot).

Using constant variance (left panels) the CDF values for expectation values smaller than the average tend to aggregate towards high levels for all three models, while the CDF values corresponding to high expectation values look reasonably uniform, with the exception of the lognormal distribution.

Using constant variance several of the gamma shape parameters in cases of low expectation values were below 1, giving probability density functions that were decreasing with expected value, which is not very appealing. The probability density functions in cases of high expectation values were very narrow, being not very tolerant to outliers. When constant CV was used the probability density functions had the same shape parameter, in no cases below zero.

Using constant CV gives a reasonably uniform distribution in the cases of small expectation values but the CDFs in cases of high expectation values tend to fall off at the ends.

Therefore, neither assumption seems wholly satisfactory. It should be investigated how the distribution of CDFs relate to different model specifications (i.e., taking into account increased natural mortality of fish infected by *ichthyophonus hoferi*, changing overall value of the natural mortality).

### Retrospective properties

Using the gamma distribution the assessment was done in both 1998 and 1997 and with the different treatments of the data that also were tried by the WG:

- Default values
  - Data before 1991 are not used because the WG felt that technological changes made them incompatible to data from 1991 and later.
  - Using all four surveys
  - Excluding the youngest fish which were felt not were fully recruited to the surveys
  - Excluding the smallest cohorts that were felt only would contribute noise to the tuning
  - Using data before 1991
  - Deleting the international survey in the Norwegian Sea (to compare with the assessment done in 1997 where this series was not used)

The text table below shows the results for the number (billion) of the 1983, 1991 and 1992 year classes in 1997, as determined with 1998 or 1997 as the final year.

### Constant variance

Estimated Survey Log 1983 in 1997 1991 in 1997 1992 in 1997

variance variance Likelihood

Default,	1998	0.78	0.83-212.10	1.73	15.86	23.14
Default,	1997	0.79	0.89-201.02	2.41	15.49	24.16
Earlier data,	1998	0.62	0.59-333.80	2.10	19.16	27.47
Earlier data,	1997	0.63	0.66-430.06	2.55	16.53	26.9
Deleting Norw.	1998	0.76	0.80-188.48	2.03	17.49	23.11
Deleting Norw.	1997	0.73	0.72-175.77	2.11	12.59	15.49

### Constant CV

Default,	1998		1.85-186.85	1.19	9.86	12.81
Default,	1997		4.76-183.18	1.13	8.09	13.79
Earlier data,	1998		3.27-192.94	0.74	6.71	11.19
Earlier data,	1997		31.88-177.86	0.75	5.45	7.92
Deleting Norw,	1998		1.68-168.45	1.17	9.80	12.88
Deleting Norw,	1997		1.78-159.57	1.20	6.94	7.14

Using constant CV instead of constant variance increases the survey variance because the assessment is more tolerant to outliers. The likelihood, though, is higher indicating an improved fit.

For the 1992 and 1991 year classes the differences between doing the assessments in 1997 and 1998 are not very large if the survey in the Norwegian Sea is used. However, using constant CV instead of constant variance nearly halves the stock abundance. This survey was not available at the assessment in 1997 and without using it the assessments done in 1998 and 1997 give different results for these year classes.

Using constant CV instead of constant variance gives much lower stock abundance in all cases. It is clear that the assessment is overly sensitive to the distribution used for the surveys. Since neither of the tested distributions seem fully satisfactory, one should look for a distribution that might fit the data better. Also, the systematic deviances from the assumed distribution may point to a tuning model mis-specification of some kind.

### 7.3 Problems connected to the surveys in the overwintering area

When overwintering in the Lofoten area the herring stock occupies a very restricted area and the density becomes high. The trawling time is counted in seconds. A multi-sampling trawl was introduced in 1996 giving the possibility to take separate catches at different depths and, perhaps more important, to open and close the trawl at the trawl depth. Without this trawl it was difficult to get age representativity in cases of stratification of age with depth, which might occur, since the trawl fishes not only at the trawl depth but also on its way up and down.

Also, because of the dense concentration of fish, the shadowing problem may become important. In addition, the target strength differ with depth (Huse and Ona, 1996) and yearly fluctuation oceanographic conditions that may lead the herring to different mean depth may cause problems when using the series for tuning.

### The natural mortality

The M-value of the stock is traditionally inferred from tagging experiments and has been estimated by the Atlantoscandian herring and capelin WG (later Northern Pelagic and Blue Whiting Fisheries WG) to 0.16 in 1969, to 0.13 in 1985, to 0.23 in 1993 when *ichthyophonus hoferi* prevailed in the adult stock and to 0.16 in 1996 (0.08 if M was allowed to vary during the tuning) (WD by Sigurd Tjelmeland). The tagging mortality (inverse of tagging survival that is used here) was changed in 1993 from 30% to 40%, the latter value being based on tank experiments.

At the assessment meetings of the ICES Northern Pelagic and Blue Whiting Fisheries WG the natural mortality was allowed to vary along with the other parameters in a Bayesian estimation, and the marginal probability distribution of M peaked towards the lowest allowed value (0.1). This is also seen for Western herring (WD by Kenneth Patterson). A value of M close to zero seems improbable, and this effect may point to problems with the survey as such.

When M was allowed to vary in the tuning exercises made at this meeting, a value of 0.051 was obtained. However, this value is suspiciously low and it should be further investigated whether this low value is an indication of problems elsewhere in the tuning procedure.

Low M-values correspond to low terminal stocks and are therefore necessarily accompanied by high values of the survey calibration factors. More importantly, a low M-value is connected to a low value of the tagging survival. The text table below shows the value of these entities when the tuning is made for M-values of 0.15, 0.10 and 0.05. The runs were made using a gamma distribution with constant variance for the surveys.

M	CalSpawn	CalDec	CalJan	CalNorw	F1983	F1992	SpSurv	Gamma Variance	Log Likelihood
0.15	0.56	0.36	0.59	0.85	0.26	0.09	0.50	0.78	-212
0.10	0.76	0.49	0.81	1.19	0.36	0.12	0.33	0.76	-208
0.05	0.96	0.63	1.03	1.53	0.47	0.15	0.23	0.76	-207

Tagging survivals much below 0.5, which has been estimated to be about 0.6 in tank experiments, seem highly improbable. It might be possible, though, that the tagged herring would have a higher mortality during its whole life span. If this is the case, such a mortality should be built into the tuning procedure. However, it would be very difficult to obtain an independent estimate.

For an M-value of 0.05 the international survey in the Norwegian Sea is a 50% overestimate, which also seems hardly probable.

The same tuning experiments were done with a constant CV, where M was estimated at 0.10:

M CalSpawn CalDec CalJan CalNorw F1983 F1992 SpSurv GammaCV LogLikelihood

0.15 0.80 0.77 1.00 1.59 0.41 0.16 0.43 0.38 -186.9  
 0.10 0.93 0.90 1.18 1.83 0.47 0.19 0.31 0.38 -186.1  
 0.05 1.06 1.03 1.36 2.10 0.54 0.22 0.23 0.37 -186.7

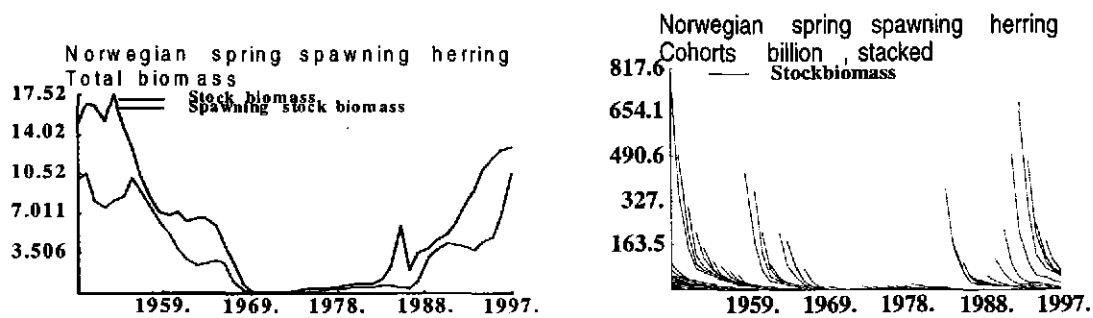
This case displays the same trend, only the change in likelihood is very small.

#### 7.4 Conclusion

There is a serious problem of unknown origin in the tuning of Norwegian spring spawning herring. It has been shown that the assumptions made about the probability distribution of the surveys is crucial. Further work should be done on this stock to find a suitable distribution function and/or rectify unknown errors in model specifications and treatment of survey data.

#### Harvesting control rules for Norwegian spring spawning herring

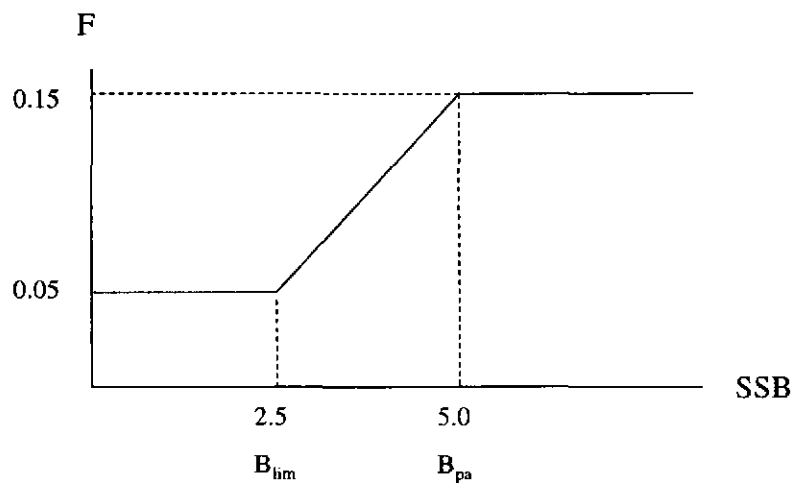
The Norwegian spring spawning herring is characterised by strong recruitment dynamics, where the stock as a rule is dominated by a single or a few very strong year classes. Figure 7.3.1 shows the recent stock development in biomass terms (left panel) and the cohorts in number (right panel) where each cohort has been stacked on top of each other. The cohort plot shows the important role of the 1950, 1959, 1983, 1991 and 1992 year classes. By the late 1960s the stock collapsed, probably to a large extent driven by excessive fishery where a large part of the catch was taken on the juvenile (cannable) part of the population. Based on a stock-recruitment plot the strategy for rebuilding the stock was to reach a spawning stock of 2.5 million tonnes, below which it was thought that the spawning stock would be too small to yield good recruitment, even if the conditions for recruitment were good (Dragesund *et al* 1980).



**Figure 7.3.1.** Recent stock history of Norwegian spring spawning herring, stock biomass (million tonnes, left) and cohorts in numbers (youngest age is zero, billion, right).

The 1983 year class brought the spawning stock above 2.5 million tonnes and the recruitment improved, the 1991 and 1992 year classes were especially strong and as a consequence the stock rose to its former heights of about 10 million tonnes by the mid 1990s.

Clearly, there is need for a harvesting control rule for this stock that aims at maximising long-term yield (with due account taken of avoiding new stock collapses) rather than keeping the spawning stock above a minimum threshold. ACFM adopted a rule of a constant  $F$ -value at 0.15 combined with a catch ceiling of 1.5 million tonnes. The Northern Pelagic and Blue Whiting WG suggested a precautionary rule showed in Figure 7.3.2 based on lowering the  $F$ -value when the spawning stock fell below a threshold defined as the  $B_{pa}$  for this stock (ICES 1998b).



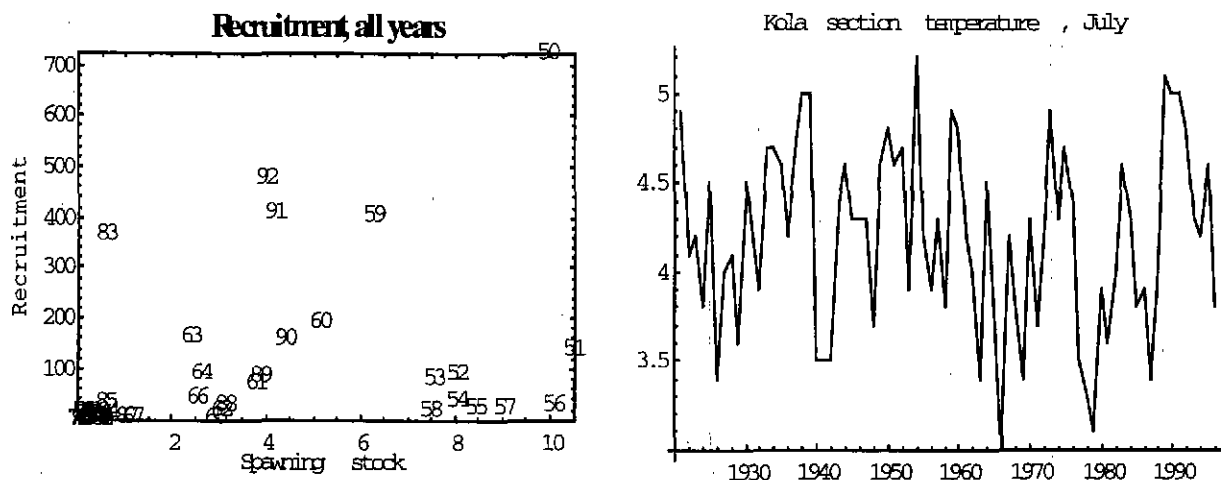
**Figure 7.3.2.** Precautionary rule to be used during stock decline for Norwegian spring spawning herring as proposed by the Northern Pelagic and Blue Whiting Fisheries WG.

The  $B_{lim}$  is taken as 2.5 million tonnes. It is at present debated whether the  $F$ -value should be reduced to zero at  $B_{lim}$ , reduced proportionally to zero as the spawning stock reduces to zero or kept at the lowest  $F$ -value.

The numerical values put forward are to some extent provisional and dictated by the necessity of arriving at a pre-agreed harvesting control rule as early as possible now the stock is declining due to poor year classes recruiting to the fishery.

## 7.4.1 Recruitment functions

Figure 7.3.3 shows the spawning stock recruitment pairs (left panel) and the mean 0–200 m temperature along the Kola meridian (Tereschenko, 1996) (right panel). The recruitment numbers are based on a VPA run to age 0 (ICES 1998æ).



**Figure 7.3.3.** Spawning stock-recruitment plot for Norwegian spring spawning herring (left) and July temperature along the Kola meridian (right).

The recruitment pattern has these important features:

- A few year classes (1950, 1959, 1983, 1991 and 1992) are especially outstanding.
- For spawning stocks below 2 million tonnes the recruitment is poor, with exception of the 1983 year class.
- In the period 1951 to 1958 there was no good year classes despite very high spawning stocks.
- In the period 1960 to 1966 there was good recruitment from spawning stocks in the range 2–5 million tonnes.

It is difficult from the temperature time series to see any clue for environmental effects that might cause relatively long periods of similar recruitment pattern. Nevertheless, relatively long periods of recruitment consistently high for mid-range levels of spawning stock and consistently low for high spawning stocks makes it plausible that population variables important for recruitment are autocorrelated and with abrupt shifts in mean level.

The weight at age for the 1950s are not known and constant values are used in the ICES data base, making it for the moment impossible to investigate whether the poor recruitment in this period was due to poor growth. However, data might exist that could be used for this purpose. Especially, the Russian drift net data from the Norwegian Sea could be useful for this purpose.

Of importance could also be cannibalism. Larvae drifting northwards after spawning may overlap with juvenile herring migrating from the Barents Sea (Holst and Røttingen, 1994) or be eaten by the post-spawning herring. It would be possible that this effect also is environmental-dependent since migration - and thereby overlap - could be environmentally-driven.

Two different recruitment functions will be used during the testing of harvesting control rules:

- 1) Beverton-Holt with a normal distribution of errors on a logarithmic scale.
- 2) Beverton-Holt with a normal distribution of errors on a logarithmic scale, but using two different functions, one for "good" recruitment conditions and one for "poor" recruitment conditions. The probability of changing from a state of "good" or "poor" recruitment conditions to the opposite state is determined from the observed historical length of such conditions. "Good" and "poor" are defined according to whether the value of the spawning stock that yields half the maximum recruitment being below or above the average value, where the maximum recruitment in the Beverton-Holt function has been fixed at 1.5 the maximum observed recruitment.



In the regime shift models the probability of changing from a poor-recruitment regime to a good-recruitment regime is 0.38 and the probability of changing from a good-recruitment regime to a poor-recruitment regime is 0.32.

Simulation model

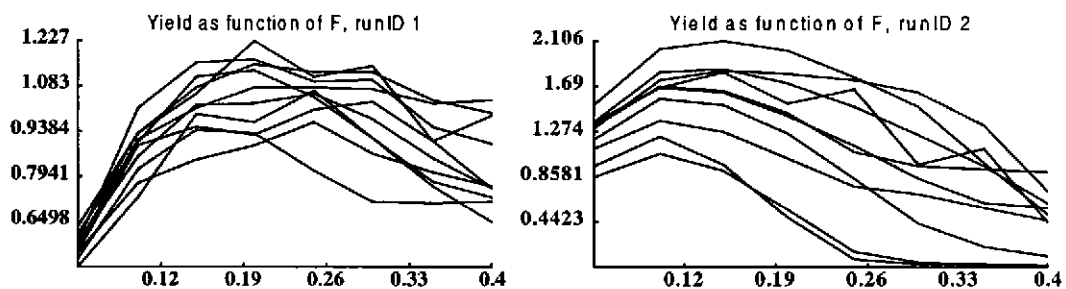
The simulations were in each case run 300 years, and no observation error was assumed, i.e., the harvesting control rule were followed exactly. The simulations were run for different values of F, each time resetting the random number generator to the same value. Each series of runs using different F was repeated 10 times with different seeds for the random number generator.

Two different harvesting control rules were tried: Either a fixed F or a fixed F combined with a catch ceiling of 1.0 million tonnes and a lowering of F to 0.0 at 2.5 million tonnes when the spawning stock decreases from 4.0 million tonnes (precautionary rule).

The weight at age in the catch, the weight at age in the stock and the proportion mature by age were taken from the ICES data files and were in each year taken from a randomly selected historic year. That is, abundance dependence effects were not accounted for. The exploitation pattern was kept fixed at the exploitation pattern in the final year of the VPA. The number of recruits were kept below 2 times the maximum observed number of recruits.

Yield as function of F – constant F rule

Figure 7.3.5 shows the mean long-term yield as function of F for the 10 replicates for the two different recruitment models.



**Figure 7.3.5.** Average yield curves for F-values in the range 0.05–0.40 for long-term simulations using logarithmic Beverton-Holt (upper left) and logarithmic Beverton-Holt with regime shift (upper right).

Each case gives a different management message. The simple logarithmic Beverton-Holt yields an optimal F-value of about 0.2, somewhat higher than the value proposed by ACFM. Using the regime-shift model gives an optimal F-value that is substantially lower and with a higher mean catch. That is, if regime shifts are operating as modelled, fishing at the proposed F of 0.15 is to clearly exceed the F-value corresponding to maximum long-term expected sustainable yield. Conversely, it seems from the curves that if one used the regime-shift model and the apparent regime shifts were accidental, some catch would be lost.

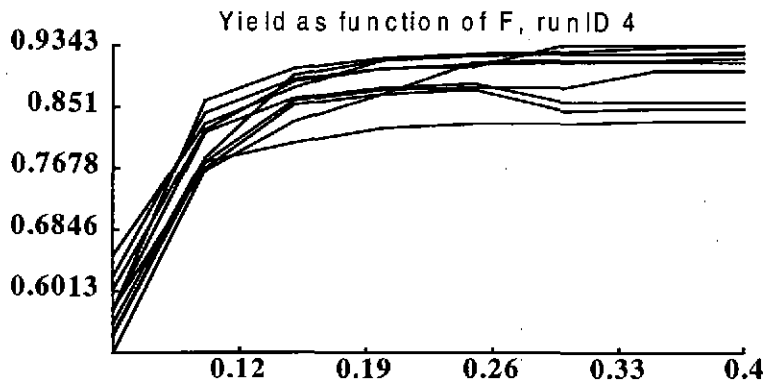
The numeric values calculated from these few runs are given in the text table below:

Model	OptimalF	Stdv.	Mean Catch Stdv.	
Log BH	0.184	0.030	1.067	0.108
Log BH regime shift	0.121	0.018	1.627	0.319

The mean long-term catches corresponding to the regime shift model is of comparable magnitude to the yearly catches taken in the period prior to the collapse of the stock and may therefore seem unrealistic. However, during that period a large part of the catch was taken on juvenile fish, giving a very irrational exploitation pattern, so whether the catches given above are unrealistic or not seems unclear.

Effect of the precautionary rule

Figure 7.3.6 shows a simulation run using fixed F-values but combined with a catch ceiling of 1.0 million tonnes and a lowering of the F below 4.0 million tonnes to 0.0 at a spawning stock of 2.5 million tonnes. The recruitment model used is the default Beverton-Holt model with logarithmic errors.

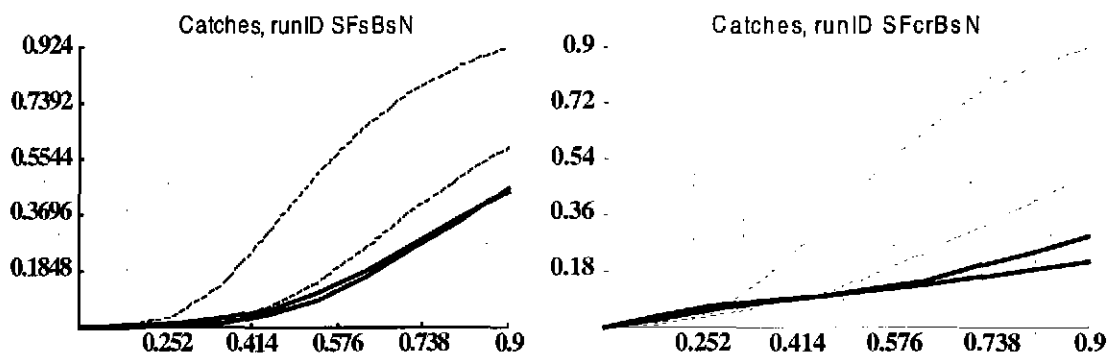


**Figure 7.3.6.** Average yield curves for F-values in the range 0.05–0.40 for logarithmic Beverton-Holt and using a catch ceiling of 4.0 million tonnes combined with lowering the F-value to 0.0 at a spawning stock of 2.5 million tonnes.

There is no downward bending slope because the precautionary rule takes over when the stock falls below 4 million tonnes at high F-values.

In lowering the F-value when the stock declines the decline time is prolonged and there is a higher probability that there will be good recruitment once the conditions for recruitment is good. The result will be that catches, even if small, will be kept over a longer time. The ceiling part of the precautionary rule has the effect of transferring catch from "good" times to "bad" times, but this is paid for by a natural mortality cost. But also this should enhance stability of catches.

Figure 7.3.7 shows the probability of the catch being below the value on the x-axis for F-values 0.05 and 0.10 (dashed lines) and 0.15 and 0.20 (solid lines).



**Figure 7.3.7.** Cumulative distribution of catches for F-values 0.05 and 0.10 (dashed) and 0.15 and 0.20 (solid). Default model in left panel, model applying precautionary approach rule in right panel.

It is seen that the rule really effected its goal. With the default model there is about 45% probability of the catch to be below 0.9 million tonnes, with the precautionary rule the probability of the catch being below 0.9 million tonnes is reduced to about 25%.

Economic considerations

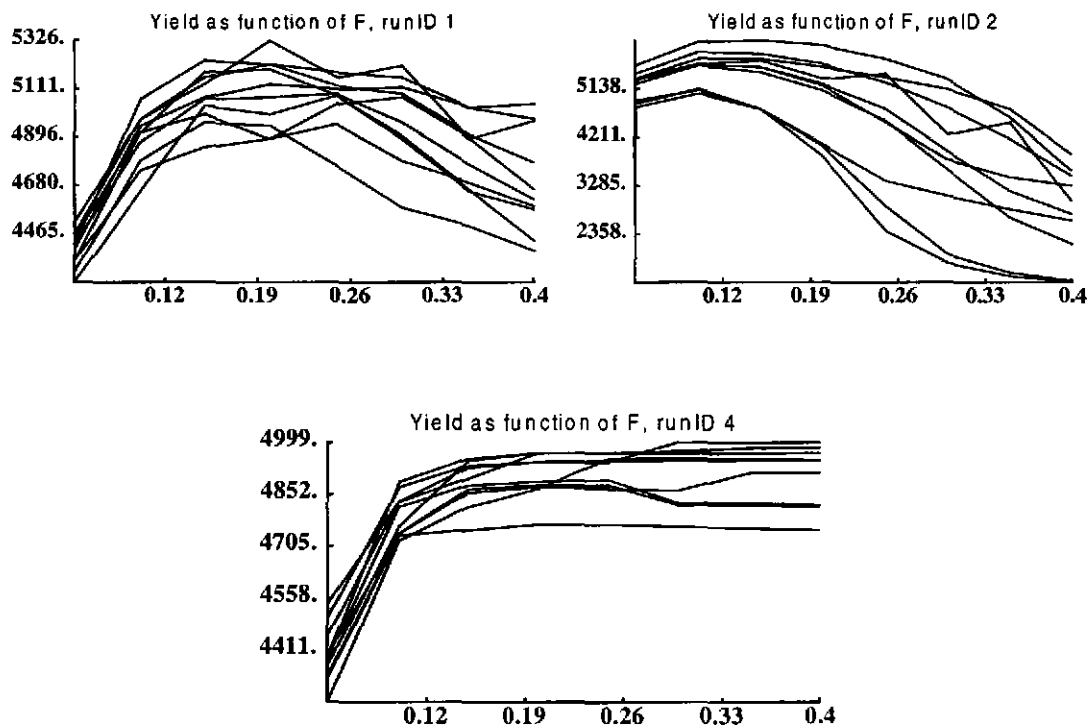
If the TAC in a particular year is low the price per kg as a rule will be higher, so what is lost in catch quantity to some extent can be compensated for by higher prices. Based on Norwegian data from the period 1980–1993 the price per kg can be expressed as:

$$P = 1.0 + 29.84 \text{ TAC}^{-0.745}$$

where P is the price in NOK/kg and TAC the total catch in 1000 tonnes (Sandberg *et al* 1998). Variable costs are in (Sandberg *et al* 1998) assumed proportional to unit catch, but these data were not available. It could be argued that the costs per catch unit would be lower when the stock is high than when the stock is low. However, for a schooling fish like the Norwegian spring spawning herring this argument would probably not be valid. Including variable costs proportional to unit catch would enhance the economic effect.

Using Norwegian prices implies an assumption that the various countries' relative share of the stock remains unchanged and that the same price elasticity function is valid in all countries, which is not very likely. If the stock declines to below, say, one million tonnes it may well be confined to Norwegian coastal waters again. If the stock changes its overwintering area to the pre-collapse area east of Iceland the relative shares would also be affected. Including these effects calls for an elaborate analysis where also the migration is addresses. This is not feasible at the moment.

Figure 7.3.8 shows the yield curves in economic terms using the above function.



**Figure 7.3.8.** Average yield (billion NOK) curves when price elasticity is included. Default model in upper left panel, regime shift model in upper right panel and using the precautionary rule on the default model in lower panel.

For all models there is a large increase in the yield corresponding to the lowest F-value tested (0.05) to the runs made for the next lowest F-value (0.10), the message being that bringing in the economic dimension favours smaller F-values when one keeps the harvest on the cautious side of the stochastic MSY-point.

The magnitude of this effect, which merely has been illustrated in the present calculations, should be carefully evaluated using as realistic models for economy and biology as possible.

The fishing fleet that operates on the Norwegian spring spawning herring is in Norway diverse, ranging from small coastal vessels that also are used in fisheries like the coastal saithe fishery and large purse seiners that also operate on the mackerel, blue whiting and capelin fisheries. It would in the future be interesting to look at harvesting control

strategies that viewed a set of fisheries together (P. Sandberg, pers comm), which might be termed economic interactions as opposed to biological and technical interactions. For instance, if a large fishery on Barents Sea capelin could be allowed – which may in the future happen in some years – the quota on Norwegian spring spawning herring might be lowered, saving some catch for the future.

#### **7.5 Further advance of comprehensive assessment for Norwegian spring spawning herring**

The two most immediate needs are sorting out the tuning problems for the stock and to implement a precautionary harvesting control rule that the countries involved can agree upon in an early state of the present decline of the stock and some directions for this work have been suggested.

The Norwegian spring spawning herring once present in the Barents Sea may influence that ecosystem radically by strongly hampering capelin recruitment. Also, there is the possibility of interaction through competition for food with blue whiting in the Norwegian Sea. Large parts of the same fleet operate on all these species and the total bio-economical system should be studied, where the role of the herring in Norwegian Sea – Barents Sea complex should be better understood.

Also, investigations to reveal mechanisms affecting the large-scale migration of herring should be better understood and included in a comprehensive analysis.

### **8 NORTH SEA HERRING COMPREHENSIVE ASSESSMENT**

#### **8.1 Introduction**

No resources have been allocated to COMFIE to complete this task. However, some of the work done in other ICES Working Groups and under EU auspices is briefly reviewed and commented on here. The following review is based on comparing work in progress against the targets and objectives outlined by COMFIE in its previous report (ICES 1997x)

#### **8.2 Stock Structure**

No comprehensive review of the stock structure of herring in the North Sea and adjacent areas has been completed. However, some recent studies have improved knowledge about the discrimination of herring between ICES Divisions IIIa and IV (using otolith increment analysis) and between VIa and IV (based on multivariate analyses of herring size- and age-structure).

##### **8.2.1 Discrimination between Herring of the IIIa + Western Baltic Stock and North Sea Herring**

Present belief about migration of North Sea Herring and IIIa+Western Baltic Spring-spawning herring is that a mixture of herring from the two stocks occurs in the Kattegat, Skagerrak and the Eastern part of the North Sea. Some of the adult IIIa+WB spring-spawning herring make a partial migration into the North Sea area. Until recently, it was believed that only juveniles of the North Sea autumn-spawning stock migrated into the Skagerrak and Kattegat, and that they returned to the North Sea at ages 2–3 years. However, recent otolith-based determinations of the stock identity of herring in Division IIIa suggest significant abundances of adult North Sea Herring occur in Division IIIa. The validity and precision of this method is not yet well established.

##### **8.2.2 Discrimination between herring of the VIa(N) [West Scotland] area and North Sea herring**

Herring to the West of 4 degrees W have been treated as a separate stock from the North Sea herring. The validity of the separation has been questioned in some fora, and attempts were made by the Herring Assessment Working Group to validate this stock discrimination based on information from IBTS and acoustic survey information. The following population variables were available from the survey time-series:

Cumulative proportion of fish at each age from 1 to 8, by year  
Length at each age from 2 to 9+, and three growth parameter estimates, by year.

This information was calculated from 77 trawl hauls in Division VIa(N), 97 hauls in the North Sea, and 18 hauls in the intermediate area between 3 degrees East and 5 degrees West.

Two classification methods were used: classical discriminant analysis, and an artificial neural network system. Results from both methods supported the hypothesis that populations with different growth and age-structures exist in the two areas. The methods appeared to provide discrimination in the samples from the two areas with about 95% accuracy, as tested by applying the discriminant function (or neural network) to test data of known origin. This analysis indicated some improved separation between the two stocks might be achieved by changing the administrative division from a North-South line following the fourth meridian to a NW/SE line from 59 degrees N 3 degrees 30 W. However, the major populations are largely separated by the existing area division.

Analyses of this type may prove misleading if fish populations undertake migrations which are highly size-specific or age specific, as in such cases sampling at different locations will obviously find fish with different size or age-structures, without the populations necessarily being at all discrete. Support for this type of analysis using methods that allow migrations to be traced directly (e.g., tagging or shoal-following) should if possible be used to support inferences from distributional studies.

### **8.3 Management Measures**

An arrangement for the management of the North Sea herring was agreed upon by the European Community (EC) and Norway in December 1997. The essence of this is to derive yearly quotas according to agreed levels of fishing mortalities, which shall be 0.12 for  $F_{0-1}$  and 0.25 for  $F_{2-6}$ . If the SSB estimate falls below 1.3 million tonnes, special measures shall be negotiated to bring the SSB above this level. The Norwegian share of the quota for the human consumption fishery in the North Sea is 29% and the EU has a 100% share of the by-catch fishery.

At present, the SSB is below the 1.3 million t limit. The quotas for 1999 correspond to a fishing mortality of 0.2 for adults with a juvenile by-catch quatum of 30,000 t (which corresponds to a low  $F$  0.047).

The main difference between this arrangement and previous ones is that it restricts the fishing mortality at all ages, including juveniles. Previously, there was also an SSB-based rule for the share of the quota between EC and Norway, which is now substituted by a fixed percentage.

It should be noted that although the fleets in broad terms exploit either juvenile or adult herring, there is some overlap, in particular for the human consumption fishery in Division IIIa. Accordingly, if the only restriction is through overall  $F$ 's for juveniles and adults, the problem of allocation to the various fleets has multiple solutions, even if each fleet is assumed to maintain its fishing pattern.

There exist a number of technical measures such as closed areas and maximum bycatch percentages which are not reviewed in detail here.

A certain amount of the EC quota in the North Sea is set aside for fishery in Division IVc and Division VIId. In recent years the Downs quatum was set at 25,000 tonnes.

### **8.4 Review of Surveys**

COMFIE does not presently have sufficient resources to evaluate work in progress on evaluating North Sea herring surveying and data collection. At least one EU-funded project has been initiated on this topic. However, in recent assessments a marked divergence in the trends in stock sizes that could be inferred from the various possible indices of stock size has been a persistent yet unresolved problem. This is illustrated in Figure 8.5.2.

### **8.5 Assessment Models**

#### **8.5.1 Historic and Present Assessment Methodology**

North Sea herring assessments before 1994 (ad hoc tuning methods) had a tendency to underestimate fishing mortality and to overestimate stock size by around 50% compared to current stock size estimates (Figure 8.5.1). Since the assessment was changed to a statistical catch at age model (ICA; Patterson & Melvin 1996), it has been closely consistent. However, problems remain in that survey divergence is a persistent yet unexplained feature of the data set (Figure 8.5.2).

In the current assessment procedure all surveys are given an equal weight. Because a separable model is used, the length of the separable period times the number of ages determines the total weight of the catch at age matrix as compared to the survey estimates. In the most recent assessment the separable period was 6 years and the number of ages in the

separable constraint was 7 (1-9+; excluding the reference age 4 and the final age 9) giving a total weight of 42 which should be compared to a total survey weighting of 5 (i.e., 5 survey indices).

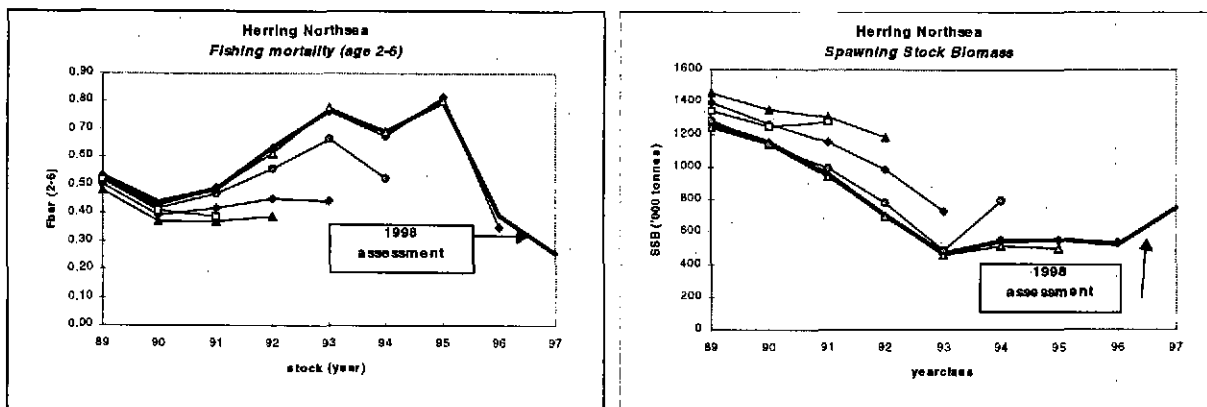


Figure 8.5.1 North Sea herring assessments as they were done by the respective working groups. Left: fishing mortality (2-8). Right: spawning stock biomass. Source: ICES (1990, 1998) and intermediate assessments.

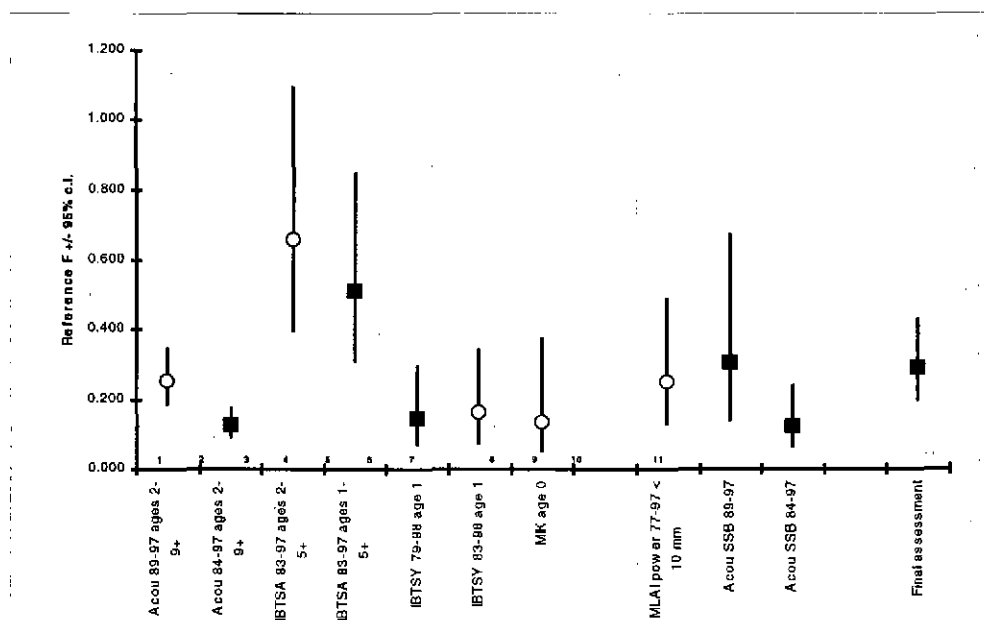
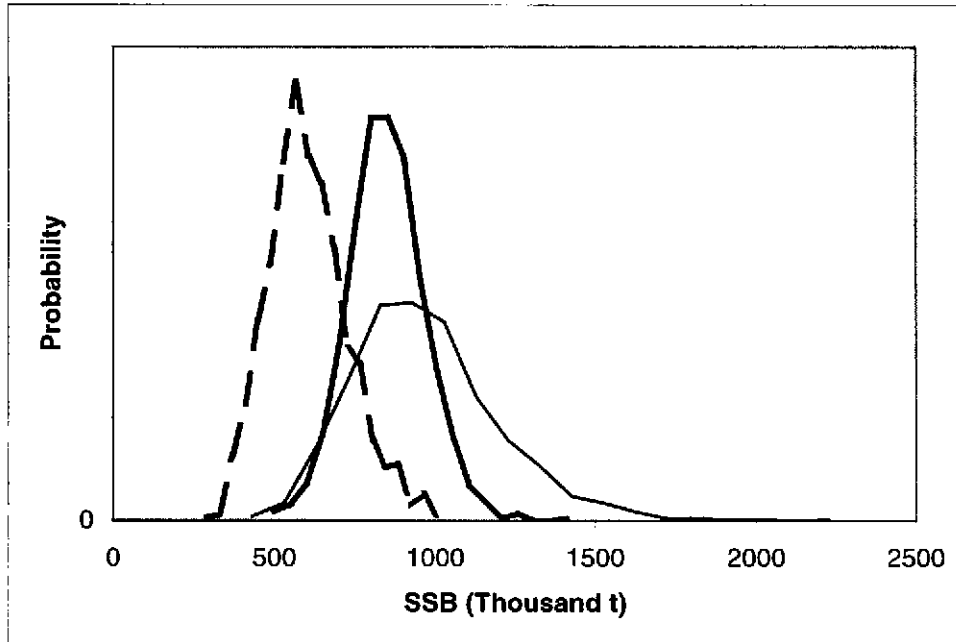


Figure 8.5.2 Herring in Sub-area IV, Divisions VIIId and IIIa. Estimates of fishing mortality (+/- 95 c.i.) in population models fitted to the separate indices and the catch at age matrix. Each index is given an equal weight. The open circles indicate the indices that were used in the final assessment (ICES 1998v).

### 8.5.2 Estimation of Survey Variances

The assessment model for North Sea herring in current use is based on a weighted least-squares minimisation including terms for catch at age data, acoustic survey data, larvae surveys and trawl (IBTS) surveys. The 'weighting factors', (corresponding to the reciprocal of the variance of each data series) are unknown, yet because of a marked divergence in stock size trends for each series, these weighting factors have a large influence on the final stock size estimate used for management purposes. In recent years, these weighting factors have all been assigned arbitrary values = 1, and the assessment model has performed with a consistency considered acceptable for assessment purposes. However, the herring assessment Working Group reported that uncertainty due to model specification, including the choice of the weighting factors, accounted for about half of the uncertainty in terminal-year fishing mortality estimates, compared with the uncertainty due to divergence of the abundance index series. Despite the good performance of the assessment model in recent years, the topic remains of some concern.

An alternative approach to the treatment of weighting factors was suggested in Patterson (1998). Instead of making an arbitrary assumption for these values, an alternative could be explicitly to define the factors as uncertain over a range defined as *a priori* acceptable, as formal priors in a Bayesian calculation. Other parameters were assigned uniform priors. A Markov Chain Monte Carlo integration could then be used to calculate marginal posterior distributions for parameters of interest. In an example calculation, it was shown that markedly different perceptions of stock size (and the concomitant uncertainty estimates) are derived by this approach compared with those obtained by making an assumption of equal weighting factors (Figure 8.6.1).



**Figure 8.6.1.** Alternative perceptions of North Sea herring stock sizes derived from three approaches to specifying 'weighting factors' (= reciprocals of variances) for survey and catch-at-age data. Bold line, Estimated marginal posterior distribution from Bayes MCMC calculation admitting variances to be uncertain. Dashed line, estimated distribution from frequentist calculation assuming variances to be equal. Fine line, estimated distribution from frequentist calculation conditioned on iteratively-estimated variances.

At present, this alternative approach is considered promising but is not recommended for adoption until the merits of this type of calculation have been evaluated further. The very large computational burden (e.g., 10 days) of this method tends to restrict the extent to which simulation testing can be used for evaluation. The theoretical basis underlying this sort of calculation does however seem *a priori* appropriate to the structure of the perceived problem.

### 8.5.3 Stock-Recruit Models

An ICES study group on stock-recruitment relationships for North Sea herring was held in May 1998 (ICES 1998b). The group revised data back to 1947 to provide an amended series of stock-recruit pairs, and considered several ways of modelling the relation between stock and recruitment.

A preliminary revision of the stock weights for the period 1947–1960 was made, based on the assumption of density dependent growth. Data on length at age in the Belgian fishery for the central and southern North Sea, and Scottish data from the Buchan area were applied to a simple model relating the growth rate ( $K$  in the Ford growth equation) to the biomass. Fitting a Beverton - Holt function to these data gave parameters fairly similar to those used previously with a shorter time series. The estimates of the SSB in the earliest years was about 5 million tonnes. The group could not address the question of changes in maturity ogive, but noticed, both that this should be done, given the evidence for density dependent growth, and that alternative measures of effective fecundity should be considered. A simple experiment with a proxy for the fecundity obtained by substituting stock weights by weights raised to the power 1.3 was presented.

Several areas where the data still could be improved were identified. These include:

- More extensive revision of the catch numbers and stock weight, covering the whole period 1947–83, where standard weights are used in the assessment.
- Revision of maturity at age.
- Use of fecundity - weight relationships to enable calculation of effective fecundity.
- Attempt to include catches from Division IIIa for the years (1947 - 1979) for which this has not been done.

The group suggested that work on these items should be followed up by the Comprehensive Fisheries Evaluation Working Group.

The group also considered several possible formulations of the relationship between stock and recruitment. These include parametric functions, non-parametric regression functions, time series models and 'assumption-free' methods. The use of these methods for different purposes, in particular for establishing limit reference points and for estimating equilibrium points, were discussed. The study group did not have time to make systematic comparisons between these models. However, the outcome by the various approaches did not vary very much, which may be due to the wide range of SSB's and the modest variance of the residuals in this stock. A common finding was an anomaly where the recruitment was far more successful in the recovery from the collapse in the early 1980s than in the years prior to the collapse. Another general impression was that the slope at the origin of the stock-recruitment relation, which is an important basis for defining limit reference points, would be poorly defined for most stocks.

#### 8.5.4 Prediction Methods

The fishery for North Sea herring can broadly be partitioned into two categories, fishing for adult herring for human consumption, using trawl or purse seine, and fishing juvenile herring with small meshed trawl for reduction purposes. The juvenile herring is to a large extent bycatch in industrial fisheries. Furthermore, the fishery in the North Sea (ICES Sub-area IV) and in the Skagerrak-Kattegat (ICES Division IIIa) have separate quotas and is to a large extent performed by separate fleets, and the herring that is exploited in Division IIIa is a mixture of juvenile herring of the North Sea stock, and of Baltic herring.

In the ICES Herring Assessment Working Group for the area south of 62°N, five fleets have been defined on these grounds:

##### North Sea

- Fleet A: Directed herring fisheries with purse seiners and trawlers
- Fleet B: All other vessels where herring is taken as by-catch

##### Division IIIa

- Fleet C: Directed herring fisheries with purse seiners and trawlers
- Fleet D: Vessels fishing under the mixed clupeoid (sprat) quota
- Fleet E: All other vessels participating in fisheries where herring is taken as by-catch

*Short term predictions* for management advice (quota recommendations) are done with reference to these five fleets. The Working Group has in recent years used a spreadsheet programme for this purpose, which performs a deterministic stock projection and partitions the predicted catch into fleetwise catches. The essence of this is to use the last year's catches to define partial fishing mortalities with respect to the North Sea herring for each fleet separately, and to assume a yearly split factor how the North Sea juvenile herring are distributed between Sub-area IV and Division IIIa. The spreadsheet program has been revised several times, and work is in progress to substitute it with a program which also may take uncertainties into account.

Medium term predictions have developed along two lines, one is using the projection part of the ICA assessment programs (Patterson & Melvin 1996), the other is a two-fleet stochastic projection with some options for simulating harvest control rules (HCR; Skagen, 1997). In both programs, the distributions of future outcome are obtained by Monte Carlo simulation of stock projections. The stochastic variables can be initial stock numbers, recruitment and weight and maturity at age. The HCR simulation program also allows for testing the robustness to bias in the assessments and to overfishing of the calculated quotas. None of the methods recalculate assessments as part of the projection process.



### 8.5.5 Harvest Control Laws

Harvest control rules for North Sea herring were evaluated by a subgroup of the Herring Assessment Working group for the area south of 62°N in 1997 (Patterson et al, 1997), at the request of the EU commission and the government of Norway. The ACFM advice following this study was an important background for the management regime described in Section 8.3.

The mandate given was to *'advise on appropriate management régimes (i.e., 'harvest control laws') including reference points at which immediate remedial action should be taken and appropriate time scales for actions which might be used in future management of the stock and which takes into account sustainable exploitation rates and appropriate biomass thresholds'*.

Experience, not the least with the North Sea herring, has shown that a seemingly well designed management regime can go wrong if the assumptions underlying the decision rules do not hold. (Torsion et al, 1997). Among the causes identified for the North Sea herring is that the stock size was overestimated for several years. management regime should therefore be shown to be robust to such model failure. This led to a design where different rules would apply at different levels of SSB, with a harvesting aiming at a high yield in the long term associated with a low risk of deteriorating the stock when the stock is in a good shape, and a more cautious regime when this is not so. Moreover, the stock biomass at that time was below the MBAL, which called for a separate recovery plan. Since juvenile and adult herring are exploited by different fleets, there was a specific need to evaluate the trade-off between juvenile and adult fishery, and to specify the exploitation specifically for both juveniles and adults.

The tools for evaluating the candidate harvest control rules were the medium term projection programmes used by the Herring Working Group modified for this purpose (see Section 8.5.4).

The harvest control rules explored defined a limit SSB, a precautionary SSB and different F-constraints at the corresponding three levels of SSB. As the limit SSB ( $B_{lim}$ ) the previously well established MBAL of 800 000 tonnes was adopted. For the precautionary SSB ( $B_{pa}$ ) several levels ranging from 1.5 to 2.5 million tonnes were explored. The finally recommended value was 2.2 million tonnes, corresponding roughly to BMSY. This estimate is in agreement with recent estimates derived by the ICES Study Group on Stock-recruitment Relationships for North Sea Herring.

At  $SSB < B_{lim}$ , the fishery ideally should be closed. At  $SSB > B_{pa}$ , the ambition was to recommend a fishing mortality near FMSY. Since FMSY is uncertain for many reasons, not the least because the data on potential stock size and productivity at moderate exploitation are scarce,  $F_{0.1}$  (= 0.21) was recommended as a provisional  $F_{pa}$ . With SSB between these values, a linear reduction of F with reduced SSB and a fixed F regime were explored.

The fishing mortality of the juveniles has only a modest influence on the reference points for the adult stock, although the yield will depend on how much of a year class is left to become adult. Thus, the main trade-off would be between the fleets exploiting juvenile and adult herring - gains for those fishing for juveniles would have to be paid by those fishing for adults.

The robustness of this regime to assessment bias was tested using the medium term projection method mentioned in Section 8.5.4, by assuming that next years quota is set after applying a normally distributed multiplier to the simulated SSB. It was found that an average bias factor of up to 1.3 only rarely led to true  $SSB < B_{lim}$ .

A recovery plan for the immediate future was also suggested, where quotas would be as recommended for 1996, and further reduced if there were indications that this would be insufficient to increase the SSB.

## 8.6 Prices and Costs in the herring fishery

Since the previous COMFIE meeting a substantial contribution to price and cost estimation for North Sea herring has been completed under an EU-shared cost project (Bio-economic evaluation of management options for North Sea herring). A draft of the final report of that project was available to COMFIE and is briefly reviewed here.

### 8.6.1 Introduction

This research project was designed with the goal of completing sufficient econometric estimation of price, cost, and demand functions to enable forecasts to be made of revenue and profit by fleet sector. Applying these functions to the ACFM catch forecast scenarios, corresponding forecasts of revenue, profit and employment could be provided in order

to assist management decisions. This analysis was only applied to short-term forecasts and so was not used to calculate the relative benefits of lower catches in the short term and higher catches from a stable stock in the longer term.

### 8.6.2 Market Study and Price Formation

An extensive study of markets and of price formation in North Sea herring was completed in Anon. (1998b) and also reported on by Brown (1998). Parts of that study most relevant to the North Sea herring comprehensive fishery evaluation have been extracted and summarised below.

Herring is mostly traded as a high-volume, low-value product, suitable for markets in countries where a high proportion of the population have low incomes (e.g., Africa, Russia and Eastern European countries). New markets are also developing in Asia and China, although economic difficulties arising in Asia and Russia in 1998 have created significant risks in the development of these markets. Despite this overall perception, some niche products remain which offer high values for low volumes (e.g., 'maatjes haring' in the Netherlands; smoked herring sold to Caribbean islands).

Most herring traded internationally is in the form of frozen herring. This allows fish traders to maintain significant stored supplies which can be used to ensure regular sales despite irregular supplies, and is a relatively recent development in herring trading. Small volumes of speciality products such as salted, smoked, cured, dried or pickled herring, sold at relatively high prices into niche markets, can account for significant percentages of export values. One such niche product is herring roes sold into the Japanese market, but due to recent economic declines in that country this market has contracted considerably. Most Pacific herring catch is used for roe production.

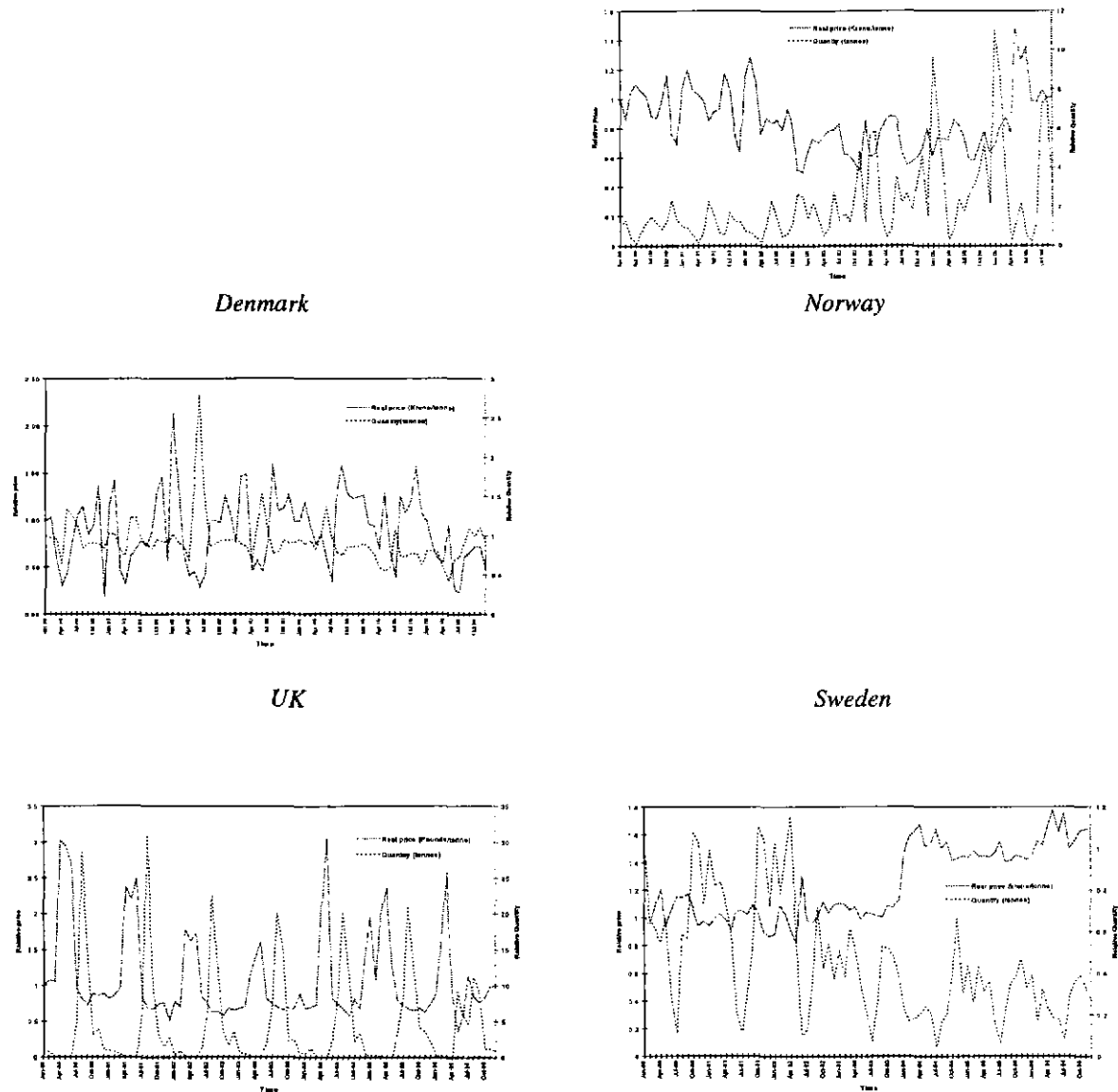
A number of elements have been highlighted as significant in affecting the price of fish. These include the quantity landed, the volume of imports, the size of the fish, advertising and various quality factors (Gates, 1974; Ioannidis and Whitmarsh, 1989; Jorgensen, Rodgers and Smit, 1989; Rodgers, 1990; Jorgensen, Rodgers, Smit and Valatin, 1991; Ty and Gates, 1992; Ioannidis and Matthew, 1995; and Ioannidis and Matthews, 1995.)

In each of these studies a different demand function was constructed. However, they all came to similar conclusions; changes in the above influences are all associated with changes in the price and, as a result, each factor will have implications for fisheries management. The studies could not be used to conclude that any one model is categorically superior to any other since they all come to their own conclusions based on the conditions of the study. The choice of variables in the demand function will simply depend on the individual market and the purpose of the model and there is no 'best' model.

The model of price formation was based on the following data, provided by month by institutes participating in the study, and adjusted for inflation and exchange rates (the prices in the model being expressed in ECU):

- a) the real landings price of herring landed in Denmark by Danish vessels
- b) the real landings price of herring landed in UK by UK vessels
- c) the real landings price of herring landed in Sweden by Swedish vessels
- d) the real landings price of herring landed in Norway by Norwegian vessels
- e) the real landings price of herring landed in Netherlands by Dutch vessels
- f) the average price of herring landed
- g) the quantity of herring landed into UK
- h) the quantity of herring landed into Denmark
- i) the quantity of herring landed into Norway
- j) the quantity of herring landed into Sweden
- k) the quantity of herring landed into the Netherlands
- l) the quantity of imports of herring in UK
- m) the quantity of imports of herring in Denmark
- n) the quantity of exports of herring in UK
- o) the quantity of exports of herring in Denmark
- p) the total quantity of herring landed

The best available estimates of prices and quantities were used, these not necessarily being official statistics. The time series of available information extended from January 1990 until December 1996 (Figure 8.6.1.)



**Figure 8.6.1.** Prices and quantities of herring estimated for Norway, Sweden, Denmark and UK, by month from January 1990 - December 1996. Includes herring from all stocks. Source: Brown, 1998. Full lines indicate prices; dashed lines indicate quantities. All values rescaled.

The modelling approach used was based on an assumption of constant price elasticity of demand, having the form (Varian 1993):

$$\ln(Q) = \ln(A) + E \cdot \ln(P)$$

where Q represents the quantity supplied, A is a constant (conceptually the price at which the first unit of production can be sold), E is a parameter corresponding to the price elasticity of demand and P represents the price at which quantity Q can be sold.

Significant seasonality was detected in the UK fishery, but there was no very consistent seasonality in the landings of herring in other countries. This is thought to be due to the fact that some of the landings figures include all stocks of herring landed and not just North Sea herring. Due to the apparent likely importance of seasonal effects, attempts were made to model such seasonality in herring prices.

It has been assumed throughout that the supply of herring is perfectly inelastic, due to quota regulations. *ie*, whatever the percentage change in price there will be no change in the quantity supplied. So supply is unresponsive to changes in demand and the demand side of the landings market can be modelled in isolation. The parameter estimation procedure assumes that when estimating the dependent variable via a number of explanatory variables, only one explanatory

variable will change at a time, otherwise one cannot be sure which variable has caused the resultant change in the dependent variable.

The price formation model was based on the form:

$$\ln(\text{Price}) = \ln(X_1) + \ln(X_2) \dots \ln(X_i) + \ln(M_1) \dots \ln(M_{12}) + \epsilon$$

*i.e.* a simple log-transformed multiple regression with various explanatory variables (specific to each country) and 12 dummy variables to represent monthly effects. Individual national models were constructed, and it was also required that an international model be built to take account of those countries thought to be price takers, a price taker being a supplier (*i.e.*, country) who has no significant influence on the price reigning in their market. The individual models were based on the foundation of simply regressing prices against quantity and were done in the currency of that country and in ECU's to aid comparison. The international model was constructed by regressing the average price of all participant countries, given in ECU's, against the quantities in each individual country. The average price being calculated by summing the real prices in each country and dividing by the number of countries for each observation. Standard statistical procedures were used in the estimating process.

**Table 8.6.1.** Summary results from the UK price function estimation procedure

'Best' Model	Explanatory Variables	Estimated Coefficient from Monthly Model	Estimated Constant
AR1 (£)	Quantity	-0.27	7.93
AR1 (Ecu)	Quantity	-0.27	8.17

Model Results: UK ( $r^2 = 81\%$ )

*i.e.*,

$$\begin{aligned} \ln \text{Price(GBP)} &= 7.93 + (-0.27 * \ln \text{Quantity}) \quad (R^2 = 81\%) \\ \ln \text{Price(ECU)} &= 8.17 + (-0.27 * \ln \text{Quantity}) \end{aligned}$$

'Best' Model	Explanatory Variables	Estimated Coefficient from Monthly Model	Estimated Constant
AR1 (Krone)	Quantity	-0.09	8.53
AR1 (Ecu)	Quantity	-0.09	6.54

Model Results: Denmark ( $r^2 = 81\%$ )

Model Results: Norway ( $r^2 = 84\%$  (NoKr); 71% (ECU))

'Best' Model	Explanatory Variables	Estimated Coefficient from Monthly Model	Estimated Constant
AR1 (Krone)	Quantity	-0.15	9.30
AR1 (Ecu)	Quantity	-0.15	7.21

Model Results: Sweden ( $r^2 = 90\%$ )

<b>'Best' Model</b>	<b>Explanatory Variables</b>	<b>Estimated Coefficient from Monthly Model</b>	<b>Estimated Constant</b>
AR1 (krona)	Quantity	-0.17	2.44
AR1 (Ecu)	Quantity	-0.17	7.23

Model Results: International' Model (Price dependent on catches by Norway, Sweden, Denmark and UK). Other countries assumed to be price-takers.  $r^2 = 88\%$ .

<b>'Best' Model</b>	<b>Explanatory Variables</b>	<b>Estimated Coefficient from the Monthly Model</b>	<b>Estimated Constant</b>
AR1 (Ecu's)	Sum of Catches by Sweden, Norway, Denmark and UK	-0.11	6.62

Although herring price can apparently be modelled adequately and simply as a function of supply, it is difficult to disentangle the pricing of herring caught from the various stocks. Herring is an internationally-traded commodity with a world production of around 3 Million tonnes and of value approximately ECU 1 000 Million in 1996. Flow matrices of herring catches and exports are given in Tables 8.6.1 and 8.6.2. From available statistics Anon. 1998 estimated that only approximately 23% of world herring exports originated in the North Sea in 1994 (214 000t compared with 933 000t for all herring exports worldwide). At this time catches of Norwegian Spring-Spawning herring were still quite low, and the influence of North Sea herring production on world prices is likely to be presently very small. For this reason, it appears infeasible to calculate a price model for North Sea herring alone. For bio-economic forecasting purposes use of a recent average price appears preferable unless a multiple-stock price-formation model can be calculated.

**Table 8.6.1.** Atlantic herring flow matrix by weight, 1996 (Quantities in tonnes)

<b>Weight (t)</b>	<b>Landings</b>	<b>Exports</b>	<b>Imports</b>	<b>Consumption</b>	<b>Total Output</b>
<b>EU</b>					
Belgium		162	3983	3821	3983
Denmark	153009	114566	103909	142352	256918
Finland	94623	8176	1808	88255	96431
France	17364	20537	14234	11061	31598
Germany	42153	38866	125396	128683	167549
Ireland	71953	19930	281	52304	72234
Netherlands	77605	103331	45998	20272	123603
Sweden	132153	92172	16699	56680	148852
UK	120935	34887	10534	96582	131469
<b>Non EU</b>					
Faroes	57693	25296	358	32755	58051
Iceland	265413	267206		-1793	265413
Norway	761211	694394	24612	91429	785823
Estonia	45296			45296	45296
Latvia	27523	7552	38	20009	27561
Lithuania	4257	3052	48596	49801	52853
Poland	31246	33172	47012	45086	78258
Romania	1794			1794	1794
Russia	134380		161779	296159	296159
Canada	187022	60347	10764	137439	197786
USA	104002	36227	16594	84369	120596
<b>Total</b>	<b>2329632</b>	<b>1559873</b>	<b>632595</b>	<b>1402354</b>	<b>2962227</b>

Consumption = landings + imports- exports

Output = consumption + exports

Modified from Anon. 1998. Data from FAO, Eurostat, Statistical Bureau of Iceland, Statistics Faroe Islands, National Marine Fisheries Service, Statistics Canada, Central Bureau of Statistics of Norway.

**Table 8.6.2. Atlantic herring flow matrix by value, 1996 (Values in Euros)**

<b>Value (thous. Euro)</b>	<b>Landings</b>	<b>Exports</b>	<b>Imports</b>	<b>Consumption</b>	<b>Total Output</b>
<b>EU</b>					
Belgium		85	2699	2614	2699
Denmark	19458	53939	37594	3113	57051
Finland	12033	2184	286	10135	12319
France	2208	5657	7489	4039	9696
Germany	5361	6661	40251	38951	45611
Ireland	9150	9845	113	-582	9262
Netherlands	9869	52837	22711	-20258	32579
Sweden	16806	17983	10968	9790	27773
UK	15379	13945	5432	6866	20811
<b>Non EU</b>					
Faroes	7337	3461	83	3959	7419
Iceland	29826	15553	526	14799	30352
Norway	96801	153598	4740	-52057	101541
Estonia	5760			5760	5760
Latvia	3500	4614	38	-1077	3538
Lithuania	541	1369	14416	13588	14957
Poland	3973	2171	1120	2923	5093
Romania	228			228	228
Russia	17088		40572	57660	57660
Canada	23783	19185	7118	11717	30902
USA	13226	23842	13446	2830	26672
<b>Total</b>	<b>292326</b>	<b>386928</b>	<b>209599</b>	<b>114997</b>	<b>501925</b>

Modified from Anon. 1998. Data from FAO, Eurostat, Statistical Bureau of Iceland, Statistics Faroe Islands, National Marine Fisheries Service, Statistics Canada, Central Bureau of Statistics of Norway.

### 8.6.3 Cost Function Estimation for the Norwegian Fleet

This analysis was based on the following data, which were available on an individual vessel basis by year in three vessel categories: purse seiners, industrial trawlers, and seiners:

#### Inputs

- Costs of fuel, product fees, social costs, insurance, vessel maintenance, gear maintenance, labour (= crew's share of the catch), and depreciation.
- Replacement value of vessel.
- Length, tonnage units, gross registered tonnes and engine horsepower.

#### Outputs

- Gross harvested quantity (t) and revenue (NOK)
- Quantity (t) and revenue(NOK) from
  - North Sea herring
  - Norwegian Spring-Spawning Herring
  - Mackerel
  - Other Species

These data were only available on an annual basis for 1994–1996. Variable costs could not be allocated to the various seasonal fisheries.

#### Modelling Approach

Modelling and parameter estimation is based on an assumption that harvesting herring can be modelled by a Cobb-Douglas functional form and that vessels are operated so as to minimise their costs.

From this assumption, an estimation equation for costs  $C$  is derived in terms of a number of price indices. These indices represent variations in the costs of each input to the individual vessel. For example, fuel cost will depend mostly on engine size (but also on other factors). A fuel cost index therefore represents the shadow cost of the input to a vessel

- The fuel cost index,  $P_f$ , is defined as the cost of purchasing fuel divided by the horse power of the engine.
- The maintenance index,  $P_v$ , is defined as the expenditure on insurance and maintenance of the vessel and gear divided by a quantity index of the vessel size.
- The price index of other inputs ( $P_o$ ) is defined as expenditure on product fees and other expenses, divided by the quantity index of vessel size.
- The quantity index of size ( $K$ ) is a proxy for the capital value of the vessel. This was calculated by assuming Cobb-Douglas form in which the replacement value of the vessel is regressed on the length of the vessel and the horsepower of the engine (with Cobb-Douglas parameters restricted to sum to one).

The cost function is defined over two outputs: North Sea herring catch ( $Q_h$ ) and the catch of other fish ( $Q_{of}$ )

$$\ln(C) = \beta_c + \beta_f \ln(P_f) + \beta_v \ln(P_v) + \beta_o \ln(P_o) + \beta_k \ln(K) + \beta_h \ln(Q_h) + \beta_{of} \ln(Q_{of}) + \varepsilon$$

The  $\beta$  represent the estimated parameters. After estimation, input demand elasticities are defined as:

$$\eta_i = \beta_i - 1, \text{ where } i = f, v, o$$

and output cost elasticities are defined as

$$\beta_j; j = h, of$$



Input demand elasticities so estimated are shown in Table 8.6.3.

Year		Input Demand Elasticities		
		Fuel	Vessel	Other
1994	Purse Seine	-0.795(0.043)	-0.419(0.033)	-0.774(0.018)
	Trawler	-0.893(0.026)	-0.434(0.021)	-0.683(0.032)
1995	Purse Seine	-0.865(0.026)	-0.443(0.029)	-0.727(0.016)
	Trawler	-0.861(0.011)	-0.425(0.012)	-0.765(0.011)
1996	Purse Seine	-0.852(0.018)	-0.451(0.020)	-0.730(0.020)
	Trawler	-0.788(0.050)	-0.522(0.043)	-0.779(0.077)
	Seiner	-0.851(0.016)	-0.459(0.015)	-0.723(0.017)

Year		Cost Elasticities		
		Herring	Other Fish	Capital
1994	Purse Seine	0.19(0.11)	0.609(0.065)	2.44(0.21)
	Trawler	0.021(0.011)	0.721(0.166)	1.24(0.420.032)
1995	Purse Seine	0.13(0.106)	0.515(0.104)	1.91(0.16)
	Trawler	0.024(0.011)	0.721(0.166)	2.36(0.55)
1996	Purse Seine	0.011(0.010)	0.585(0.065)	2.18(0.16)
	Trawler	0.021(0.110)	0.621(0.098)	1.92(1.33)
	Seiner	0.126(0.017)	0.969(0.116)	1.84(0.39)

**Table 8.6.3.** Demand and Cost elasticities for the Norwegian Purse Seine fleet (From Bjørndal and Conrad, 1998). Mean values with standard errors in parentheses.

Based on the above it was possible to calculate summary cost functions of form:

$$\ln(\text{Cost}) = \alpha + \beta \ln(Q_h)$$

for North Sea herring, based on 'average' levels of the variables other than North Sea herring harvest. Values of  $\alpha$  and  $\beta$  by fleet are given in the table below:

	Purse Seine		Trawler		Seiner	
	$\alpha$	$\beta$	$\alpha$	$\beta$	$\alpha$	$\beta$
1994	15.33	0.19	14.68	0.021		
1995	15.43	0.13	15.15	0.024		
1996	15.91	0.011	14.89	0.021	12.17	0.12

In conclusion, where data are available this approach appears successful in modelling harvesting costs for North Sea herring where sufficient data are available. However, few statistical diagnostics (e.g., no residual plots) are presented by Bjørndal and Gordon, making it difficult to evaluate the appropriateness of the use of this type of model.

#### 8.6.4 Denmark

The Danish fishery for herring is composed of a directed fishery for human consumption and a small meshed fishery for reduction purposes with herring as by-catch ("industrial fishery"). The target species of the small meshed fishery are sandeel, Norway pout and sprat. The small-meshed fisheries is constrained by special industrial quotas.

To model the Danish fishery the following components were accounted for:

Two fisheries:

- 1) Directed herring fishery for human consumption
- 2) Small meshed industrial fishery taking herring as by-catch,

Two fishing areas:

- 1) North Sea
- 2) Kattegat/Skagerrak

Four fleets within each fishery:

- 1) Small trawlers,
- 2) Medium trawlers,
- 3) Large trawlers and
- 4) Purse seines.

Five species: Herring, sprat, sandeel, Norway pout and "other industrial species"

The quarters of the year.

The model is formulated as a linear programming short-term problem, which allocates the effort to the fleets constrained by the capacity of the fleets and the quota given by ACFM.

The model maximises the revenue generated collectively by the fleets.

The catch quota was distributed on the fleets according to the Danish regulations, where individual quotas are given to the larger vessels, and smaller vessels get a common pool restricted by maximal catches per month. 85% is allocated to larger vessels and 15% to smaller vessels. Of the quota for larger vessels, the trawlers get 60% and the purse seines 40% (in 1997).

The total net revenue per year is derived as follows:

Net annual revenue = Total annual revenue - Total annual costs

Total annual revenue =  $\sum \sum \text{Price/kg} (\sum (\sum \text{Effort} * \text{CPUE}) * f(\text{stock size}))$   
quarter species fleet area

where "f" is a function which accounts for the relationship between CPUE and stock size.

Effort is measured in fishing days.

Total annual costs =  $\sum \sum \text{Variable costs} + \sum \text{fixed costs}$   
quarter fleet fleet

Variable costs = Operating costs (oil, ice, etc.) + Harvest costs (salary, crew share, cost of sale etc.)

The operation cost is a linear function of the gross tonnage and the harvesting costs is proportional to the total revenue.

The fixed cost is a linear function of the gross tonnage.

Catch, effort and price data were extracted from the logbook database and the sales slip database supplemented by the database for the species composition of industrial landings. Data from 1997 were used for reference.

Denmark was assumed to be a "price taker", with respect of the international herring market, that is, price per kg was assumed to be independent of the size of Danish herring landings.

Costs data were available only on an annual basis and data from year 1996 were used.

ACFM Options for 1999 (Total yield 000't)	Gross revenue (Million DKK)		Costs of fishing for human consumption fishery		Net revenue for human consumption herring fishery (Million DKK)
	Human Consumption fishery	Industrial fishery	Variable costs (Million DKK)	Fixed costs (Million DKK)	
I (320)	189	23	74	61	53
II (322)	187	27	74	61	52
III (386)	225	27	89	63	74
IV (395)	225	29	88	63	74
V (403)	214	34	84	61	69
VI (410)	204	41	80	61	63
VII (392)	231	27	91	63	77

**Table 8.6.4.** Costs and earnings for the Danish herring fisheries in 1999, relative to the catch-options of ACFM Unit: Million DKK

No cost data specifically related to the costs of herring by-catch in the small meshed fishery were available, and consequently, net revenues were estimated only for the directed human consumption fisheries.

#### 8.6.5 Sweden

A similar methodology was used for the Swedish trawler and purse-seine fleets as for the Norwegian fleets.

#### 8.6.6 Netherlands-based freezer trawlers

No data on Netherlands-based freezer trawlers were included in the bio-economic evaluation of the North Sea herring fishery. The Netherlands based freezer trawler can be divided into two categories:

- Dutch freezer trawlers
- Foreign freezer trawlers (French, German and English) which are owned by combined Dutch/foreign interests and which have their main harbours of landing in the Netherlands.

This freezer trawler fleet is an important catching sector in the EU pelagic fleet, and the lack of this information was seriously deleterious to the validity of the project conclusions.

Beside the freezer trawler fleet there is a relatively small part of the beamtrawl fleet that fishes for herring using pair-trawling.

The following data could be readily made available for research purposes:

- Logbook data containing days at sea, ICES rectangle and catches by trip of Dutch freezer trawlers and pair-trawlers (1990-mid 1998)
- Logbook data containing days at sea, ICES rectangle and catches by trip of Foreign freezer trawlers that land their catch in the Netherlands (1995-mid 1998)
- Composition and specification of the Dutch fleet (1990-1998)

It is currently uncertain whether the following data could also be made available:

- Prices
- Variable and fixed costs
- Revenues

### 8.6.7 Other EU Countries

As noted above, no econometric analyses of costs could be calculated for the Netherlands-based freezer trawler fleets (England, France, Germany and Netherlands). An attempt was made to model costs of the Scottish purse seine fishery based on costs of demersal vessels, but this approximation was considered poorly justifiable.

On account of these lacunae in the analysis, the Norwegian analysis of costs was used to base cost assumptions for the fisheries of Scotland, Belgium, Netherlands, France, and Germany.

### 8.6.8 Summary Results of Project

The econometric analyses described above were used to estimate revenue and costs for the EU and Norwegian fisheries for 1999, under each of seven alternative fleet-specific catch scenarios proposed by ACFM in May 1998. Catch forecasts were subdivided into national quotas on the assumption that national shares agreed for 1998 would apply for 1999 also. The economic forecasts were calculated separately for the following sectors:

- Danish directed herring fisheries
- Swedish trawlers
- Swedish purse-seiners
- Norwegian Trawlers
- Norwegian purse-seiners
- Norwegian seiners

In all cases net revenue was calculated as 'revenue - variable costs', i.e., capital costs were excluded.

Results provided to STECF and the Norwegian Ministry of Fisheries in November 1998 are summarised below. Details of the catches corresponding to ACFM options I to VII can be found in the May 1998 report of ACFM.

EUROPEAN UNION						
ACFM OPTION					Revenue (Million ECU)	
					Gross	Net
I					73.8	42.9
II					73.0	42.4
III					87.1	50.5
IV					87.0	50.4
V					83.2	48.4
VI					79.5	46.1
VII					89.2	51.6

NORWAY						
ACFM OPTION	Crew Shares		Revenue (Million NOK)		Revenue (Million ECU)	
	NOK	Million	Gross	Net	Gross	Net
I	67.2	7.7	202	121.5	23.2	13.9
II	62.1	7.1	186	112.0	21.4	12.9
III	76.9	8.8	231	141.6	26.5	16.3
IV	76.7	8.8	230	141.2	26.4	16.2
V	73.0	8.4	219	133.8	25.1	15.4
VI	68.8	7.9	207	125.5	23.8	14.4
VII	78.7	9.0	236	145.1	27.1	16.7

### **8.6.9 Conclusions about Methodology**

The Working Group does not dispose of sufficient expertise in economic analysis to make a critical evaluation of this study. However, it notes that:

- Market studies appear successful in describing changes in herring prices, but were not able to provide stock-specific price models due to the ease with which herring from different stocks may be substituted in the various markets. If market considerations are to be taken into account in managing herring stocks, this points to a need to formulate a multiple-stock marketing model for advisory purposes.
- Cost function analysis appears to be feasible in cases where an extensive database on costs was available. Presently Norway, Sweden and Denmark had adequate data for estimating cost functions. Attempts to derive shadow costs were not considered convincing. But development of cost-databases is strongly encouraged.
- The failure to include the freezer-trawler fleet sector was a serious shortcoming of this analysis.
- Given adequate data on the various fisheries, providing advice on economic consequences of alternative catch options appears to be a tractable problem.
- Provision of advice only in the short-term obscures the longer-term benefits to be achieved from fishing at lower fishing mortalities, but calculation of medium-term forecasts becomes highly sensitive to assumptions about market developments, potentially driven by changes in other herring fisheries.
- There has been no progress so far in modelling the reinvestment of profits in this project.

### **8.6.10 Recommendations**

If provision of economic advice on herring fishing is required, consideration should be given to extending the analysis to address the issues mentioned in the previous paragraph.

### **8.7 Evaluation of Data Collection**

An evaluation of commercial market sampling information is being implemented in 1999 and 2000 (EU project CFP 98/075). It aims to evaluate the adequacy of international market sampling programs for North Sea herring, plaice and cod and to enhance the optimal allocation of resources for these sampling programs. Measures of the uncertainty in the estimated catch in numbers and mean weights at age will be obtained. Moreover, it is important to have historical accessibility to dis-aggregated catch data so that the consistency of data series is warranted. Procedures for combining national catch statistics will be developed and applied. The following countries will participate in the project: The Netherlands, England, Scotland, Denmark and Belgium.

To increase participation within the ICES community, an ICES Study Group (by correspondence) has been initiated at the ASC 1998 with the same goals as defined in the EU proposal. ICES countries that are not participating in the project will be invited to participate in the project workshops.

### **8.8 Conclusions**

The North Sea herring assessment model used recently appears to have performed adequately and although methodological improvements could be proposed, such methodological developments are not seen as an urgent issue.

Significant progress has been achieved in the estimation of costs and prices for fishing North Sea herring, but the scope of existing parameter estimation is presently insufficient to allow economic parameter estimation in medium and long-term forecasts to be made.

If the issues perceived as problematic in the economic analysis could be addressed as described here, it would be possible to identify economic performance criteria and monitor their response in medium term simulation along with biological performance criteria. This is considered to be an essential component in the formulation of a comprehensive fishery evaluation for this stock.

9.1 Introduction

Some aspects of the population, growth and maturation dynamics of the stock of haddock in Icelandic waters are reviewed as part of the continuing work on the comprehensive assessment of haddock in Icelandic waters.

As indicated in the last report on the comprehensive assessment there are local variations in growth. Data from the groundfish survey indicate highly significant variations in length at age at different stations (ICES 1998x). The data used in the annual assessments have indicated a change in age of maturity and to a lesser degree a change in length at age. The importance of this to the stock and possible causes need to be investigated further. Part of the problem may be that males mature younger than females and their age of maturity is also more variable. Year class strength of haddock around Iceland exhibits marked fluctuations. Sources of variability in year class abundance along with the effect on the population are of interest, in particular potential density dependent effects on growth and age of maturity need to be considered.

As growth has the potential to affect maturation and maturation may also affect growth these factors need to be considered together. An initial study to investigate the relationship of growth and maturation has involved the use of otoliths to look at the growth history of the fish (Taylor and Stefánsson 1999). Initial results in the grouping of the growth curves of mature and immature fish are promising. The use of scales to investigate the growth-maturation process is also possible and is currently being considered. There are indications from the comparison of scale and otolith radii with length at catch that scale radii are a better predictor of length but there have been problems reading the rings. This may be due the the way the scales have been treated and their use needs to be considered further.

9.2 Icelandic Groundfish Survey and Areas

The areas within the 500m depth contour used in the Bormicon Multispecies project (Stefánsson and Pálsson 1997) are indicated in Figure 9.2.1. These areas are used in some of the following analysis.

The analyses presented here have used data from the Icelandic groundfish survey which is conducted annually in March. The stations are mostly fixed in time and the 432 stations indicated in Figure 9.2.2 have been used in the analyses presented here. The stations selected were those taken in all survey years from 1985–1998. As can be seen in Figure 9.2.2 no stations in area 7 are used in the analysis.

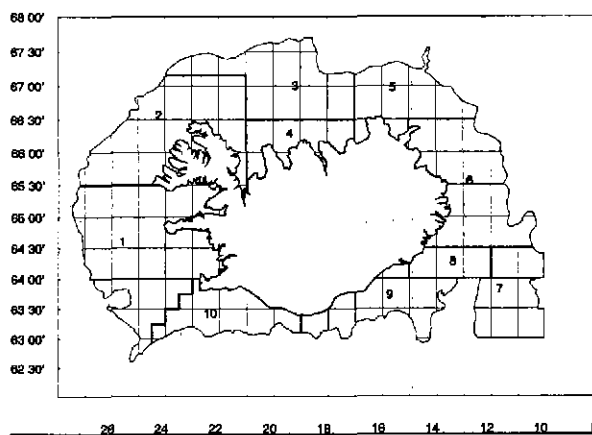
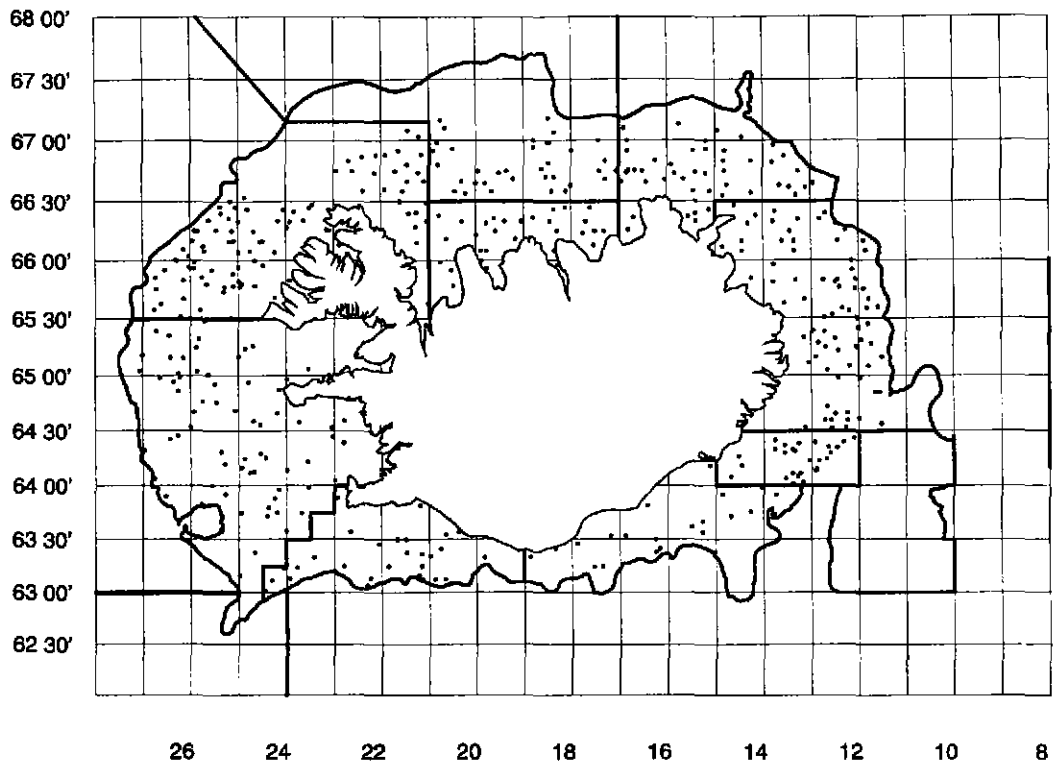


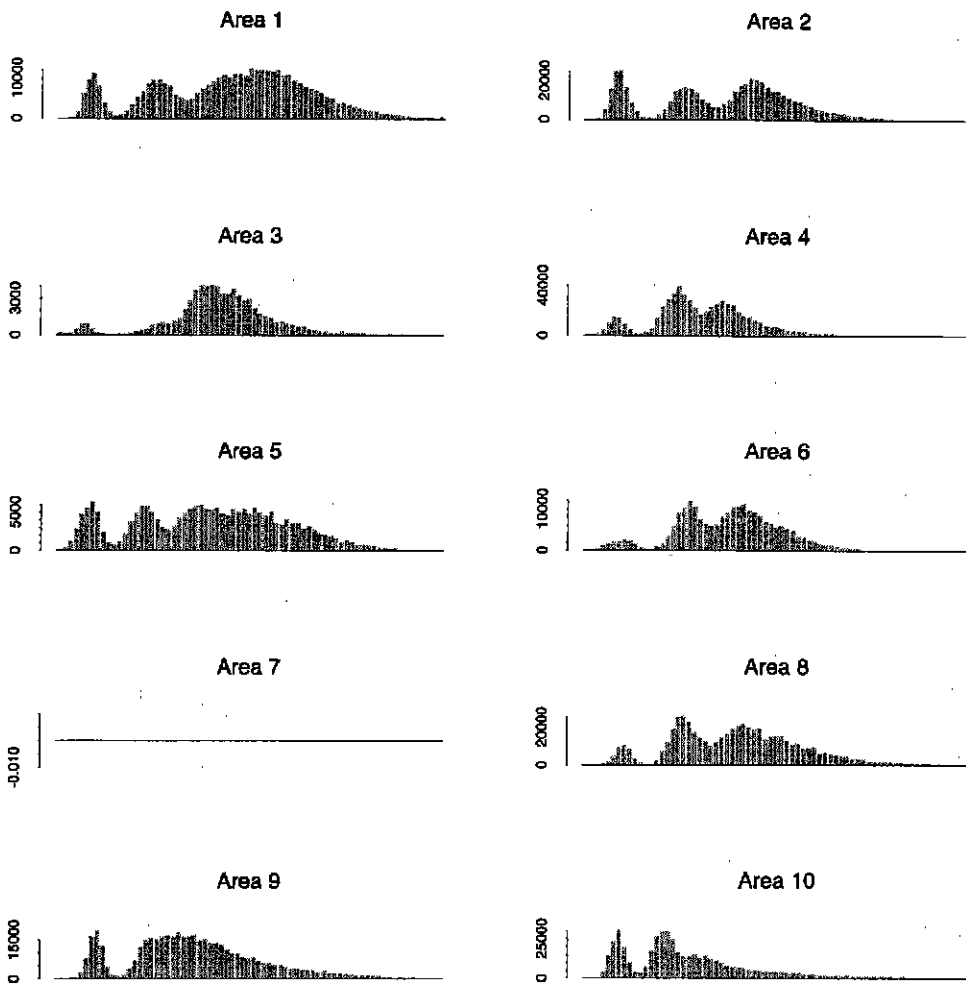
Figure 9.2.1 Bormicon Areas within the 500m depth contour around Iceland.



**Figure 9.2.2** Stations (H32) used in the analysis

### 9.3 Length distributions

Length distributions within each region are scaled by the number of fish caught per towing mile at the station. Figure 9.3.1 of annual length distributions shows how year class strength varies between years and Figure 9.3.2 illustrates the distribution by area. Clear length groups are visible for smaller fish and it can be seen that large year classes may dominate the population. The modes of the length groups have different modes in different years. The distribution by area also varies considerably e.g., there are few small fish in area 3 compared to area 5.



**Figure 9.3.2 Scaled length distribution by area**



## 9.4 Length at age

The original data contained 9 age groups, 9 regions (there being no annually sampled stations in area 7) and 14 years giving a total of 1134 observations. Region-year groups with fewer than 100 otoliths sampled were discounted, reducing the number of observations to 936. This left only 3 years of data for area 3 so it was omitted leaving a total of 909.

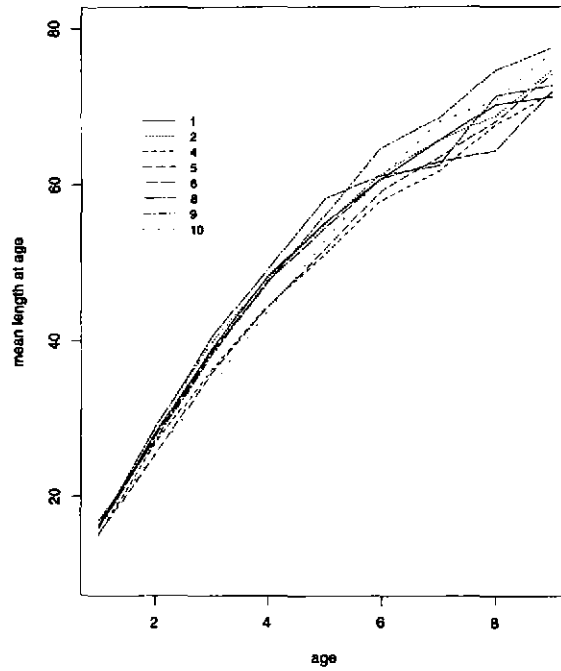


Figure 9.4.1 Mean length at age by area over all years

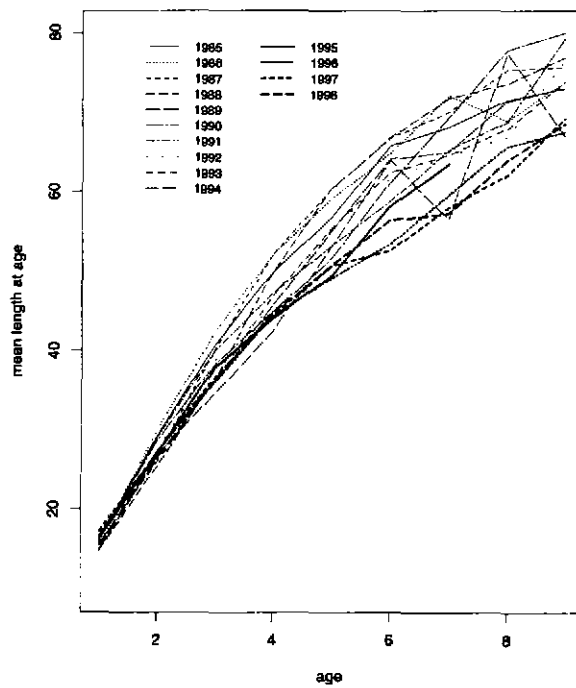
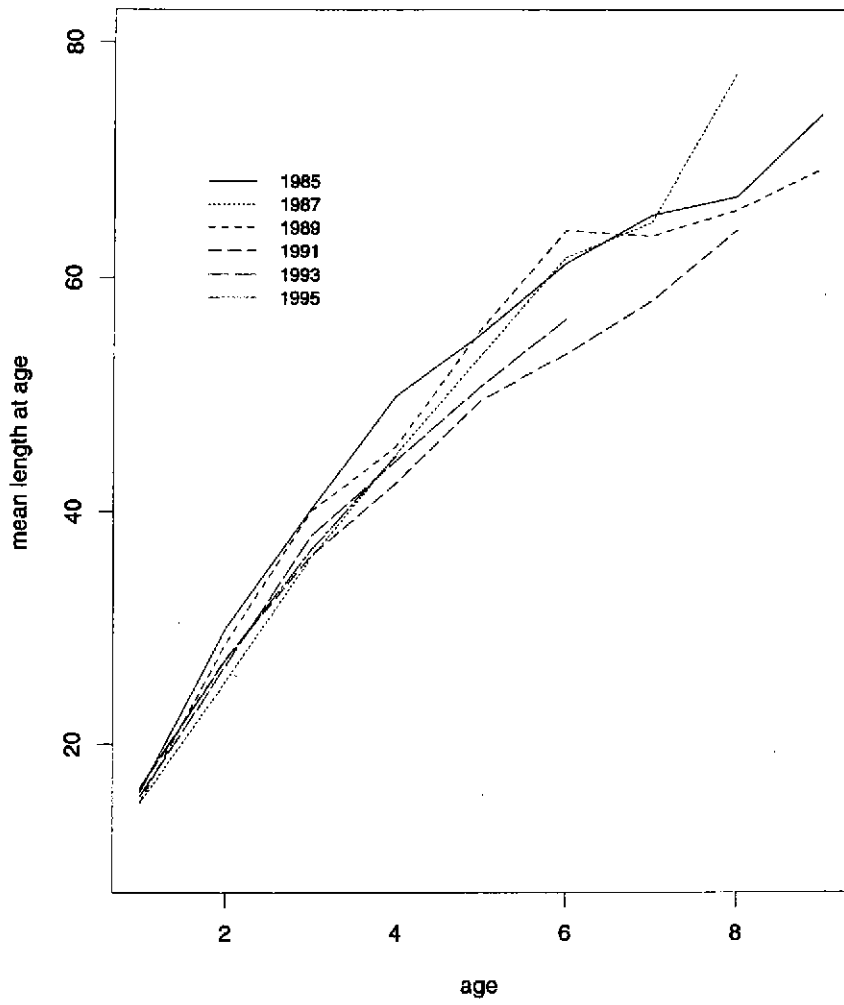


Figure 9.4.2 Mean length at age by year for all areas

Figure 9.4.3 Mean length at age by cohort



#### 9.4.1 Factors describing length at age

Length at age can be modelled using only factors to investigate the relative importance of density dependent and environmental effects. The initial ANOVA used is of ln(mean length at age) varying as a function of age, year, cohort, region, sex and maturity with the results summarised in Table 9.4.1.1.

**Table 9.4.1.1** Analysis of variance where a – age, ycl – cohort, r – region, y – year, mat – maturity.

Terms added sequentially (first to last)

	Df	Sum of Sq	Mean Sq	F Value	Pr(f)
factor(a)	8	306.0217	38.25272	4875.569	0.00000000
factor(ycl)	21	6.3199	0.30095	38.358	0.00000000
factor(r)	7	2.0259	0.28941	36.887	0.00000000
factor(sex)	1	0.4503	0.45032	57.397	0.00000000
factor(y)	12	0.6487	0.05406	6.890	0.00000000
factor(mat)	1	0.0744	0.07441	9.483	0.00210117
Residuals	2022	15.8642	0.00785		

Most of the variability is explained by the age effect but all the factors in the model are significant. Maturity is only significant with the prior inclusion of sex.

##### 9.4.1.1 Spatial variation

From the model it can be seen that mean length at age is different in different areas, as is also apparent from Figure 9.4.1. Spatial variation in growth may be due to migration (partly influenced by sexual maturity) or large year classes covering a greater area and growing slower due to density dependence. Large year classes may also appear smaller if they are large due to the survival of weaker slower growing fish which may not have survived in other years. Quality of food supply and environmental conditions in different areas may also effect growth.

##### 9.4.1.2 Sex differences

Sexual differences in length may be linked to the age of maturity for males being younger than for females. If maturation slows growth then the growth of males would be expected to slow younger. As the groundfish survey is conducted shortly before the spawning season (mid April to mid June), mature fish will tend to be in the spawning ground which may generate differences in the distribution of males and females in year classes with significantly more mature males than females.

**Table 9.4.1.2**

Terms added sequentially (first to last)

	Df	Sum of Sq	Mean Sq	F Value	Pr(F)
factor(a)	8	305.1185	38.13981	4692.135	0.00000000
factor(r)	7	1.8094	0.25848	31.800	0.00000000
factor(sex)	1	0.4710	0.47101	57.946	0.00000000
Abundance	1	0.7737	0.77371	95.185	0.00000000
factor(mat)	1	0.0655	0.06549	8.057	0.004577398
factor(ycl)	21	5.8002	0.27620	33.979	0.00000000
Residuals	2023	16.4439	0.00813		

#### 9.4.1.3 Environmental effects

The importance of the year effect along with the cohort effect indicates the annual differences in length are not solely due to the size of the cohort. Environmental factors such as hydrographical conditions may also be important and vary by region. Environmental conditions in different years may also influence the timing of migration to the spawning area – influencing the extent to which migration has taken place by the time of the groundfish survey.

**Table 9.4.1.3**

Terms added sequentially (first to last)

	Df	Sum of Sq	Mean Sq	F Value	Pr(F)
factor(a)	8	305.1185	38.13981	5766.550	0.00000000
factor(r)	7	1.8094	0.25848	39.081	0.00000000
factor(mat)	1	0.0340	0.03399	5.139	0.02350830
factor(ycl)	21	6.5473	0.31178	47.139	0.00000000
factor(y)	12	0.6395	0.05329	8.057	0.00000000
a:r	56	1.7208	0.03073	4.646	0.00000000
a:mat	8	1.0616	0.13270	20.064	0.00000000
r:mat	7	0.3682	0.05260	7.953	0.00000000
ycl:mat	20	0.3338	0.01669	2.523	0.00021823
mat:y	12	0.2164	0.01803	2.727	0.00114602
Residuals	1910	12.6327	0.00661		

#### 9.4.1.4 Density dependence

From the significance of the cohort effect, different year classes cannot be considered to have had the same pattern of growth throughout their age range (as may also be seen in Figure 9.4.3). This may indicate density dependent growth which may be tested further by the inclusion of abundance estimates.

With an estimate of abundance included for each region/year observation, abundance is significant without the prior inclusion of cohort and cohort is a significant addition further to abundance.

#### 9.4.1.5 Interactions

Interactions between various factors were tested. Age:region, age:maturity, cohort:maturity and maturity:year were significant interactions.

### 9.5 Mean condition index by age

Weight data, including liver weight, gutted weight and ungutted weight have been collected as part of the groundfish survey from 1993 and the data was organised by age, region and year as for the length data. Condition index is defined to be  $\text{weight}/\text{length}^3$

#### 9.5.1 Factors describing condition index at age

Condition index at age can be modelled as for length at age using a similar ANOVA, where mean condition index at age varies as a function of factors of age, year, cohort, region, sex and maturity.

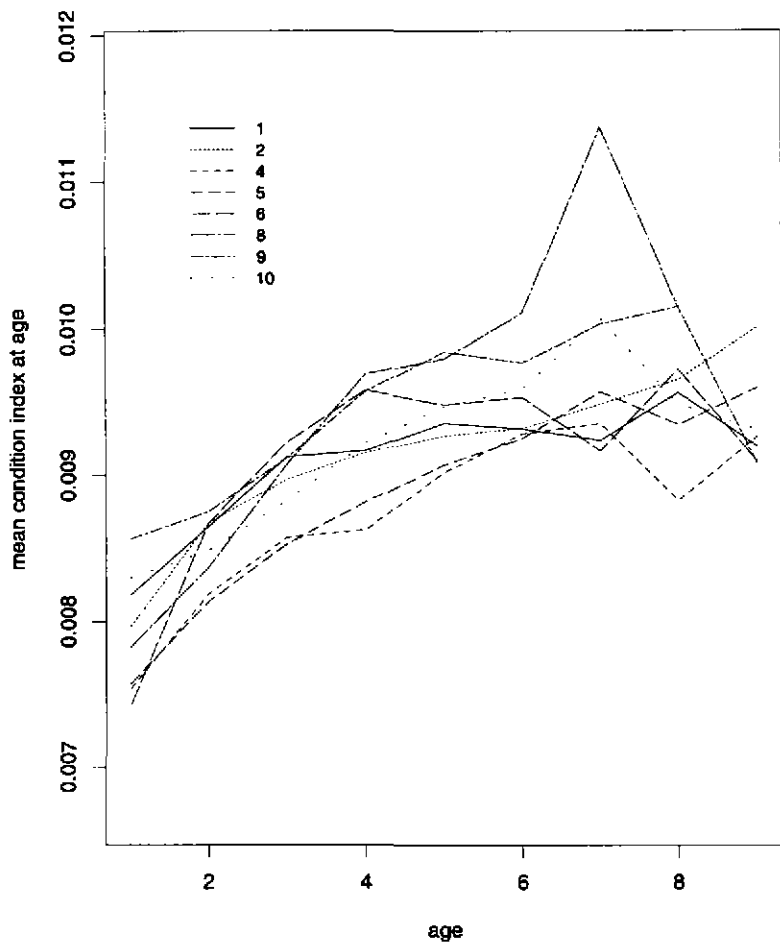


Figure 9.5.1 Mean condition index at age by area over all years

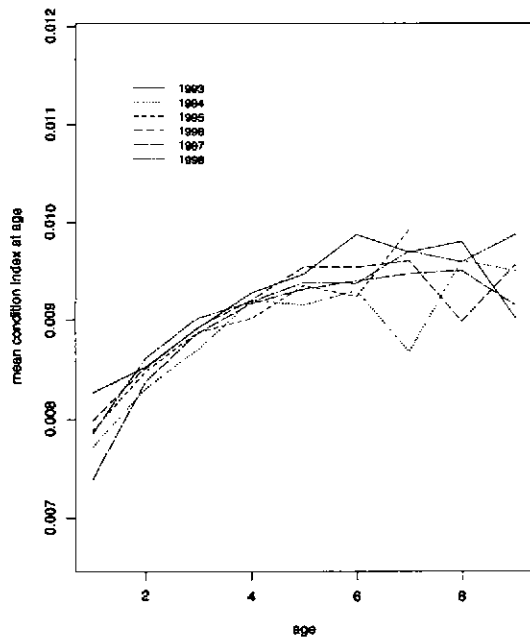


Figure 9.5.2 Mean condition index at age by year over all areas

**Table 9.5.1.1** Analysis of variance

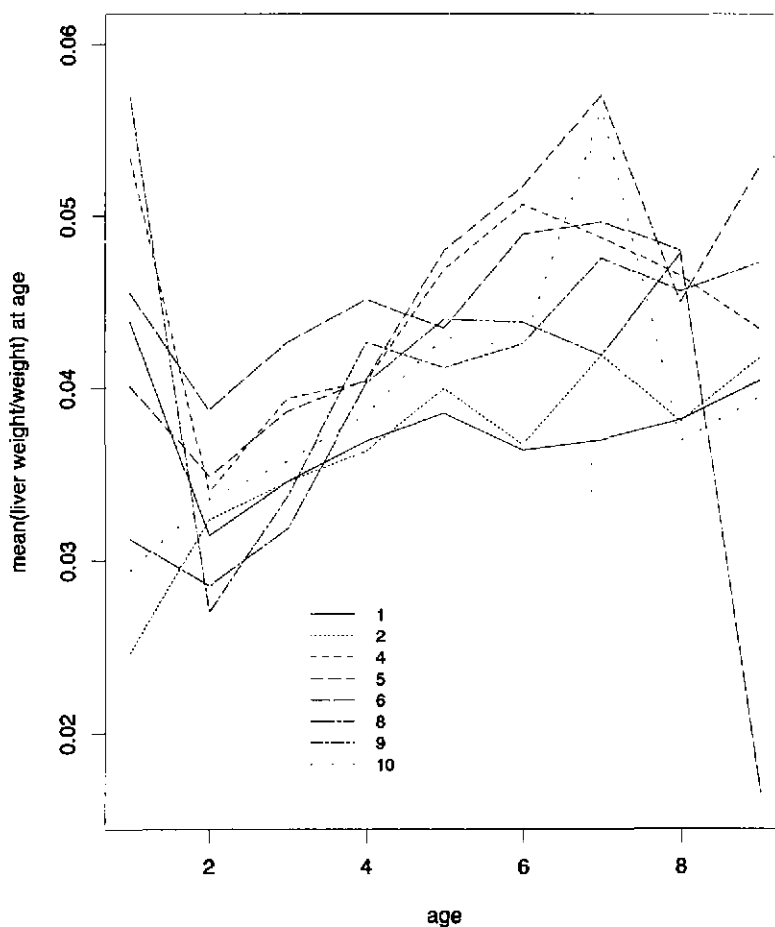
Terms added sequentially (first to last)

	Df	Sum of Sq	Mean Sq	F Value	Pr(f)
factor(mat)	1	0.000053961	0.000053961	202.9280	0.000000000
factor(r)	7	0.000041450	0.00000592	22.2685	0.000000000
factor(a)	8	0.000092503	0.000011563	43.4844	0.000000000
factor(sex)	1	0.000009376	0.000009376	35.2619	0.000000004
factor(y)	5	0.000005156	0.000001031	3.8777	0.001775744
Residuals	766	0.000203687	0.000000266		

As may be seen in Figure 9.5.1.1, Figure 9.1.5.2 and Table 9.5.1.1 the variation in condition index at age is considerably more variable by region than year. Unlike length at age, maturity is the most important effect - condition of mature fish is greater. Region is more important than year and cohort insignificant. The spatial distribution is linked to the distribution of mature fish and as mentioned in 9.4.1.2 the timing of migration to the spawning ground may vary between years. Regional influences may also be due to food availability and environmental conditions.

### 9.6 Liver weight at age

Mean (liver weight/ungutted weight) at age plotted by year and region.



**Figure 9.6.1** Mean(liver weight/weight) at age by arca over all years

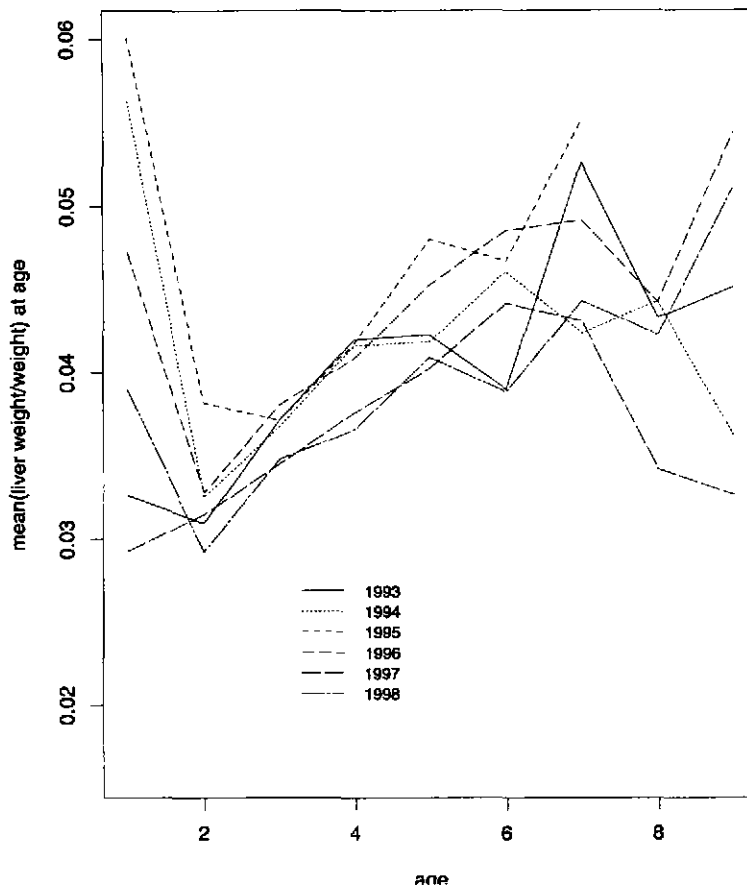


Figure 9.6.2 Mean(liver weight/weight) at age by year over all areas

9.6.1 Factors describing (liver weight/weight) at age

Liver weight/weight at age can be modelled as for condition at age. The following ANOVA is used, where mean (liver weight/weight) at age varies as a function of factors of age, year, cohort, region, sex and maturity.

Table 9.7.1.1

Terms added sequentially (first to last)

	Df	Sum of Sq	Mean Sq	F Value	Pr(F)
factor(mat)	1	0.01678583	0.01678583	239.3309	0.00000000
factor(r)	7	0.01415777	0.00202254	28.8372	0.00000000
factor(y)	5	0.00568064	0.00113613	16.1988	0.00000000
factor(a)	8	0.00332691	0.00041586	5.9293	0.00000022
factor(sex)	1	0.00121637	0.00121637	17.3429	0.00121637
factor(ycl)	12	0.00150771	0.00012564	1.7914	0.04566980
Residuals	754	0.05288291	0.00007014		

As for condition index at age, maturity and region are important effects. Year and cohort are more important for liver weight than condition.

## 9.7 Maturity at age

The proportion mature at age was analysed by count data with counts of the number of mature and immature fish for each age, length (in 5cm groups), sex, region and year combination. The count data is be modelled using a generalised linear model with a binomial distribution and a logit link function. The proportion mature may be written as

$$[ P_{\text{larys}} / 1 - P_{\text{larys}} ]$$

with factors of age, year, cohort, region, sex and length on the RHS.

**Table 9.8.1**

Terms added sequentially (first to last)

	Df	Deviance	Resid. Df	Resid. Dev.	Pr(Chi)
NULL			5156	18154.53	
factor(a)	8	10188.96	5148	7965.57	0
factor(r)	7	2303.40	5141	5662.17	0
factor(s)	1	795.92	5140	4866.25	0
factor(l)	14	427.52	5126	4438.72	0

The initial analysis indicates that there are no significant changes in maturity by year and cohort not explained by the age, region, sex and length composition of the population.

## 9.8 Future Issues

The following issues need to be considered in more detail in order to elucidate some of the issues concerning growth in Icelandic waters.

- the growth-maturation process
- density dependent growth
- effect of year class abundance on growth
- male-female differences in growth
- local patterns of growth despite migration
- stomach content data - regional differences in growth, condition and liver weight
- implementation of haddock in Bormicon

## 10 COMPREHENSIVE ASSESSMENT OF SOUTHERN GULF OF ST. LAWRENCE COD

### 10.1 Introduction

Work on the comprehensive assessment of Southern Gulf of St. Lawrence cod is ongoing in the Canadian Regional Assessment Process (RAP), the Canadian High Priority Project on the Precautionary Approach (HPPPA) and COMFIE. This section presents a summary of previous work in these other for a and new analyses presented to this year's meeting of COMFIE.

### 10.2 Natural Mortality (M)

Closure of the cod fishery and the existence of a relatively precise groundfish survey presents a unique opportunity to estimate natural mortality of this stock. Previous analyses, described in the last COMFIE report (ICES 1997x) indicated that M may have increased on this stock sometime over the past 20 years. The possible timing of such a changes was investigated in the last formal assessment of the stock (Sinclair *et al.* 1998) by examining changes in the residual mean square error (rmse) of the SPA calibration and qualitative examination of retrospective patterns of SPA estimates in response to a change in M. Normally, M is assumed to be fixed at age and year. However, if M in fact changed at some point during the time period for which SPA data are available, but it was assumed to have been constant, then the fit of the calibration model to the data would deteriorate resulting in a higher rmse (all other things being equal). It follows



that by varying M systematically, either by year or year class, the rmse should reach a minimum where the assumed M equals the true M. Temporal trends in SPA residuals will lead to retrospectivity in SPA estimates.

M was varied by year and year class between values of 0.2 or 0.4, according to

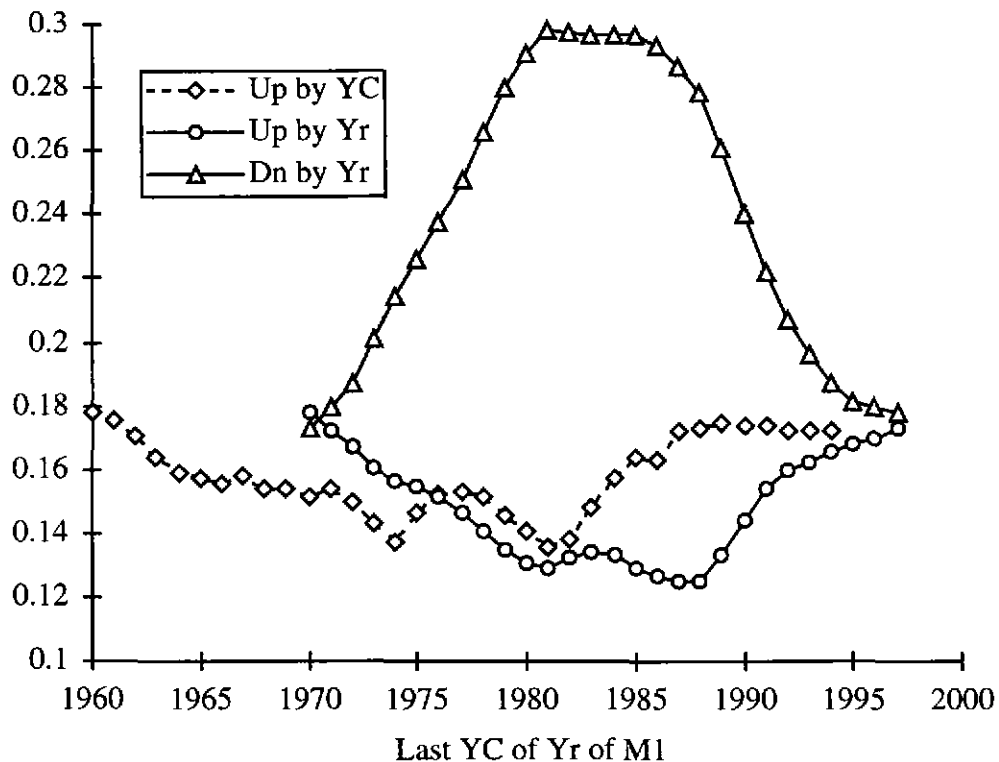
$$M_y = \begin{cases} M_1 & \text{for } y \leq i \\ M_2 & \text{for } y > i \end{cases}$$

where  $y$  = year or year class

$M_1$  is one level of M,  $M_2$  is the other  
evaluated for  $I = 1971 - 1997$

Two M scenarios were contrasted:  $M_1 = 0.2, M_2 = 0.4$ ; and  $M_1 = 0.4, M_2 = 0.2$ . The results supported the hypothesis that M increased (i.e., the former scenario) during the assessment period and that the change was year-dependent as opposed to year class dependent. The lowest rmse was found for a year-dependent increase in M, with the increase occurring between 1988–1989 (Figure 10.2.1). However, the rmse values varied little for a change in M between 1981–1988. For the increasing M scenario, the rmse increased and was much higher than for a decrease in M. It was interesting to note that the SPA calibration made virtually no distinction between fixed Ms of 0.2 and 0.4 as the rmse was similar for cases where M was constant at either 0.2 or 0.4 for the entire period.

A series of retrospective analyses were performed to determine if the year of change in M could be identified with more precision. If the SPA was calculated with a fixed M, there was a relatively large retrospective pattern. A qualitative examination of the plots suggested that increasing M from 0.2 to 0.4 in 1985 produced the most favourable retrospective patterns.



**Figure 10.2.1:** Residual mean square error (rmse) from ADAPT calibrations of southern Gulf cod using different assumptions of a change in M during the assessment period. The x-axis is the last year or year class for the first M value. Up by YC presents results when M increases from 0.2 to 0.4 by year class, Up by Yr is when M increases by year, Dn by Yr is when M decreases by year.

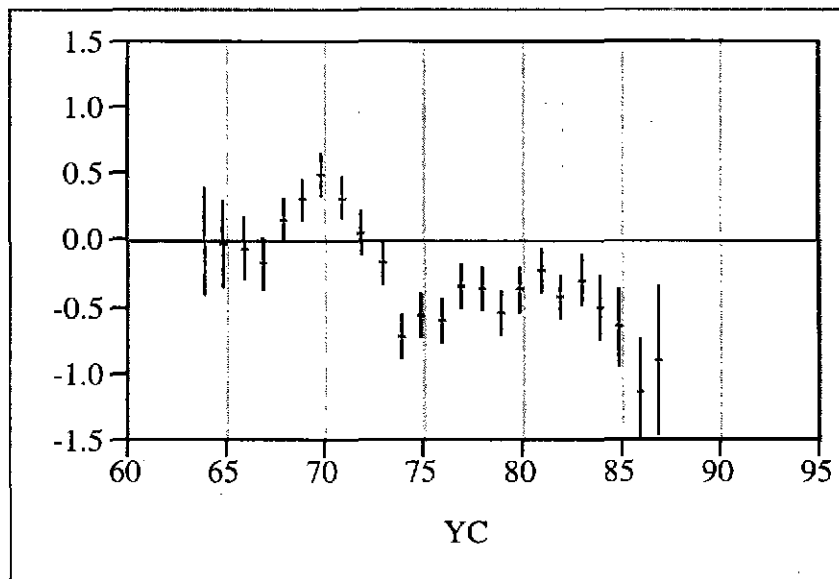
Other factors may produce a similar effect on the calibration, including an increase in the rate of catch underreporting in the late 1980s, or an increase in survey catchability in recent years. In terms of changes in survey catchability,

comparative fishing experiments were performed with each change in survey vessel, and conversion factors were applied where indicated. Cod mean size at age also changed during the assessment period, being above average in the late 1970s, then declining to below average values by the mid-1980s. It is possible that a reduction in size at age could lead to a decline in survey catchability at age due to selection by the gear. However, examination of the calibration residuals from an SPA with  $M = 0.2$  and the mean length at age measured in the annual surveys revealed the opposite trend, i.e., residuals (catchability) were highest when length at age was lowest. Thus, if there was a change in catchability as a result of the change in size at age, it was overwhelmed by other processes occurring at the same time. Catch misreporting was known to have occurred, however, the main problem was in discarding of undersized fish. While discarding will affect SPA estimates of young fish (ages 3–4 in SPA) abundance, and their associated residuals, it will not affect estimates of older fish.

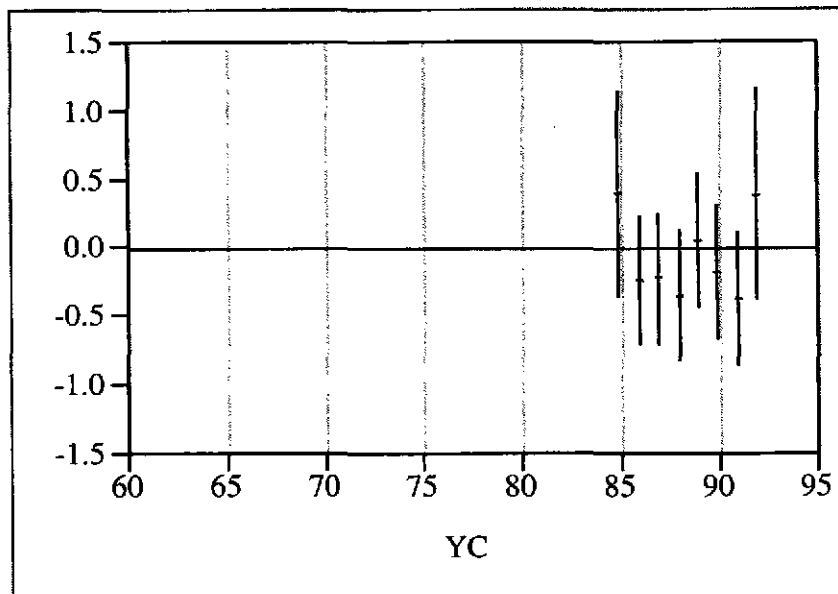
### 10.3 Changes in Size at Age

Length at age of southern Gulf cod have declined in the last 20 years from historical (since 1950) high values in the late 1970s to the lowest values observed in the mid-1980s. These changes have had a considerable effect on stock production but the causes have not been well described. Results of an analysis of size selective mortality, and its possible contribution to these changes in length at age were presented to this meeting.

Year class specific regressions of backcalculated length at age 3 vs. age of capture were used to test for size selective mortality. In the absence of size selective mortality one would expect no change in the backcalculated length at age 3 as the year class gets older. Otoliths collected from annual research vessel surveys (1971–97) were used. The first series of regressions were restricted to otoliths collected before the fishery was closed in 1993. Two periods of size selective mortality were identified (Figure 10.3.1). Selection was for larger fish in the 1968–71 year classes, as indicated by positive slopes. The situation reversed for the 1974–87 year classes where these slopes were negative. In an analysis restricted to data collected after the fishery closed, there was no evidence for size selective mortality (Figure 10.3.2) This suggests that fishing may have been the main contributor.



**Figure 10.3.1:** Slopes of regressions of backcalculated length at age 3 vs age of capture for the 1964–1987 year classes of southern Gulf of St. Lawrence cod. The data were taken for the period 1971–93, when the commercial cod fishery was active.



**Figure 10.3.2:** Slopes of regressions of backcalculated length at age 3 vs age of capture for the 1985–1992 year classes of southern Gulf of St. Lawrence cod. The data were taken for the period 1994 - 97, after the commercial cod fishery was closed.

Two indices of size selective mortality were calculated for inclusion in subsequent analyses of factors influencing size at age. An annual size selection index was calculated as the difference in mean size at age estimated in the year of capture and the backcalculated size at the same age and year class from fish caught in the following year. For a fish of year class  $j$  caught at age  $i+1$  in year  $i+j+1$ , we can backcalculate the length ( $L$ ) at its last annulus (age  $i+1$ ) and its second last annulus (age  $i$ ). The annual size selection index is thus

$$S_{i,j+i,j} = L_{i,j+i+1,j} - L_{i,j+i,j}$$

Positive values indicate that the mean length of fish surviving to year  $i+j+1$  was larger than the mean length of the same year class in year  $i+j$ . A cumulative size selection index ( $S_3$ ) was defined as the year class ( $j$ ) specific difference in mean backcalculated length at age 3 ( $L_3$ ) between the year of capture and when the fish were age 3. For a fish of year class  $j$  caught at age  $i+1$  in year  $i+j+1$ , we can backcalculate the length at its third annulus ( $L_{3,j+i+1,j}$ ), and compare this length to the backcalculated length at age 3 when the year class was 3 years old.

$$S_{3,j+i+1,j} = L_{3,j+i+1,j} - L_{3,j+3,j}$$

A negative value indicates that smaller, and thus slower growing, age 3 fish had a higher survival rate than larger fish and one would expect lower growth rates than if the index were positive.

Two analyses of the influence of size selective mortality, population density ( $N$ ) and temperature ( $T$ ) on growth of southern Gulf cod were conducted. A linear model of growth increments ( $D$ ) as a function of initial length and environmental factors ( $E$ ) was used.

$$D_{i,j} = \beta_4 + \beta_5 E_{i,j} - \beta_3 L_{i,j}$$

The standard von Bertalanffy parameters are

$$L_{\infty} = \frac{\beta_4}{\beta_3}$$

$$k = -\ln(1 - \beta_3)$$

One used growth increment data collected from individual fish and for the last complete year's growth. Only the cumulative size selective index was used in this analysis. The results indicated that size selective mortality was the most important factor, followed by population density. The temperature effect was of marginal significance. The second analysis used growth increments calculated as the difference of annual mean length at age. The annual size selective mortality was the most important factor, followed by the cumulative size selective index, density and temperature.

The working group noted that one difficulty with analyses of growth increments is the potential confounding of effects by the inclusion of initial size at age on both sides of the model equations. It was suggested that non-linear forms of the analysis, as described by Millar and Myers 1990, should be investigated.

#### **10.4 Biological Reference Points**

The impact of changes in biological characteristics of the stock, including weight at age, F at Age (selectivity), and natural mortality on per recruit and stock production biological reference points is discussed in section 3 of this report. It appears that these changes have greatly reduced the surplus production of the stock to the point that it would be prudent to monitor future changes within a stock production framework.

#### **10.5 10.5 Future Work**

Considerable progress has been made by the WG in investigating elements of a comprehensive evaluation of this stock. The WG recommends that this information be combined into a complete review of current knowledge for the next meeting of COMFIE.

### **11 COMPREHENSIVE ASSESSMENT OF NE ARCTIC COD**

#### **11.1 Introduction**

In the last COMFIE report, the focus was on description and analysis of survey data, together with a comparison of the trends in abundance and mortality in the XSA-based assessment with trends from the surveys. In this year's report, the work followed three main lines:

- (i) description of stock biology
- (ii) reference points
- (iii) assessment methodology

The developments in assessment methodology for this stock are discussed in Section 12.3.

The survey and catch data were extensively described in the last COMFIE report. The introduction of a length-based assessment model (Flexibest) required that all survey and catch data had to be recalculated to obtain the number of fish by age and length. When the recalculated data and the data presently used in the assessment were compared on an age basis, large discrepancies were found in several cases (see Ulltang, 1999). The reasons for most of these discrepancies have been identified. Before the next meeting in the Arctic Fisheries Working Group, the goal is to have completed the revision of all catch and survey data back to 1983. It is also an aim to agree on the methods used for calculation of commercial catch-at-age data.

#### **11.2 Stock biology**

In the last COMFIE report, the biology of the Northeast Arctic cod stock was only discussed briefly. Ulltang (1996) provided an overview of the biology and indicated how knowledge about the biology could be used to improve the assessment of this stock. Below, a brief description of the causes of variability of the main biological processes is given. The status of work on developing mathematical models for these processes for use in assessment of the stock is also described.

##### **11.2.1 Growth**

The individual growth of Northeast Arctic cod is highly variable. An example illustrating this is that the average weight of age 4 cod in the Norwegian winter survey increased from 0.41 kg in 1988 to 1.37 kg in 1991. Several studies of the causes of variation in growth have been made (e.g., Mehl and Sunnanå 1991; Jørgensen 1992; Nilssen *et al.* 1994; Ozhigin *et al.* 1996; Bogstad and Mehl 1997; Michalsen *et al.* 1998). These studies indicate that availability of suitable prey, especially capelin, affects growth strongly, as do temperature. There is need for a comprehensive statistical

analysis of the factors influencing cod growth to obtain a model for use in stock predictions. This could be done e.g., along the lines of Steinarsson and Stefánsson (1991). The time series used by the ICES Arctic Fisheries Working Group contains constant weights at age for the period 1946–1982, and it is important that real weight at age data for this period are provided. Bioenergetic models (Ajiad *et al.* 1994) could also help explaining the variability in cod growth.

### 11.2.2 Maturity

The maturity ogives used in the ICES assessment are based on Norwegian and Russian surveys for the period from 1982 to the present, showing average age at maturity to be approximately 7 years. For earlier years knife-edge maturity at age 8 is used. There are estimates of maturity also from this period (e.g., Ponomarenko *et al.* 1980), but only one series which covers nearly all the years. This series is published by Jørgensen (1990) and is based on analysis of the spawning rings in the otoliths. The series, starting with the 1938 year class, shows a declining trend in the age at maturity up to the early 1980's (the last complete ogive is for 1982), whereas the more recent ogives based on surveys show no such trend (Figure 11.2.1). The overlap with the survey series is poor, but indicates a difference in age at 50% maturity of approximately half a year (Jakobsen 1992). Furthermore, the ogive is generally steeper in Jørgensen's series and the SSB estimates are generally well below those based on knife-edge maturity at age 8.

All available evidence (e.g., age composition on the spawning grounds) supports the declining trend in age at first maturity seen in Jørgensen's series. This coincides with a general decline in the stock and could indicate density dependence (Figure 11.2.2). In the more recent years, however, the survey data show no relationship between stock biomass and age at maturity. The historical maturity ogives are important in the calculation of the stock and recruitment relationship and for exploring possible density dependence and will thus have a large effect on the estimation of biological reference points. Therefore, it is important that all available information is used to obtain reliable estimates of this series. This will include Norwegian and Russian data prior to 1982 from commercial sampling and surveys, as well as an evaluation of the spawning ring method used by Jørgensen (1990) which also should be extended to give more overlap with the survey series.

Maturation is closely related to growth. A model relating maturation to growth and possibly other variables is needed for stock prediction purposes. The portion of mature fish by age and the spawning biomass are not the only quantities of interest, however. Marshall *et al.* (1998) showed that the variation in survey-based estimates of total egg production is greater than that observed in VPA-based SSB and much closer to the level of variation in recruitment. Work is in progress to hindcast the reproductive potential of the Northeast Arctic cod stock based on a relationship between total liver energy and total egg production for the period 1985 to 1996 (C.T. Marshall IMR, Bergen, pers.comm). This work utilises a long-term Russian data base describing temporal and size-dependent variation in the liver condition of Northeast Arctic cod (Yaragine and Marshall submitted ms.) The possibility that total liver energy is an improved correlate of recruitment will be examined for the historical time period.

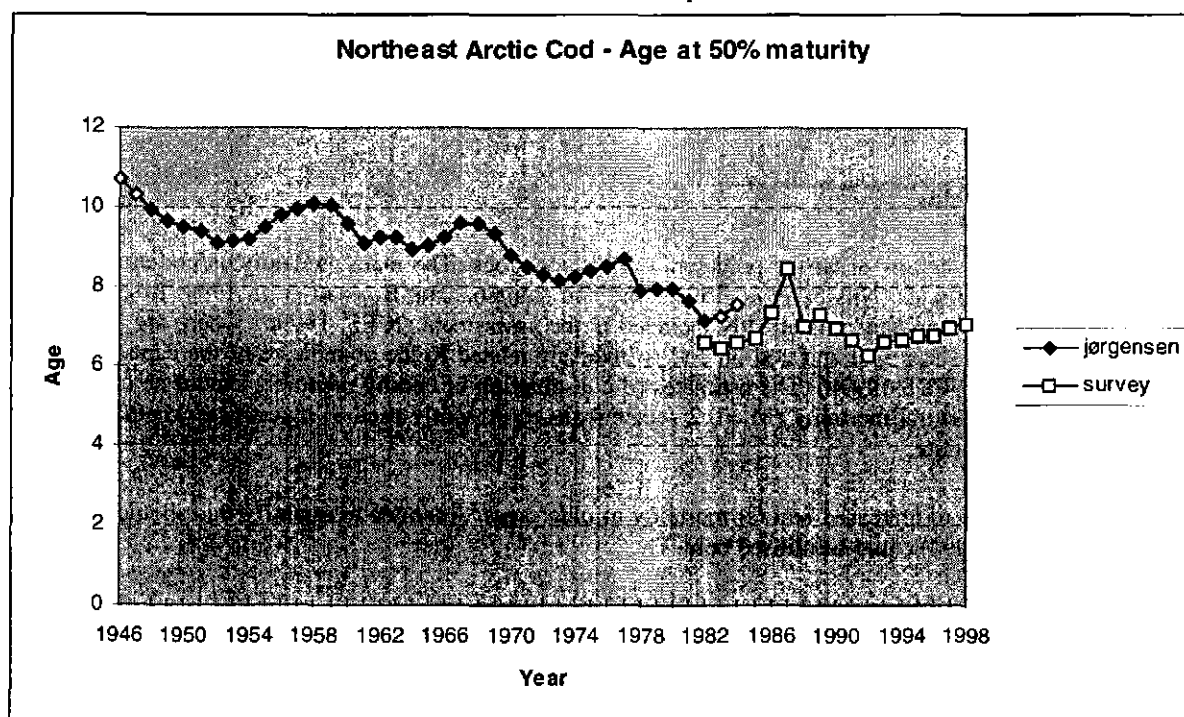


Figure 11.2.1 Age at 50% maturity as calculated from Jørgensen (1990) and from research surveys.

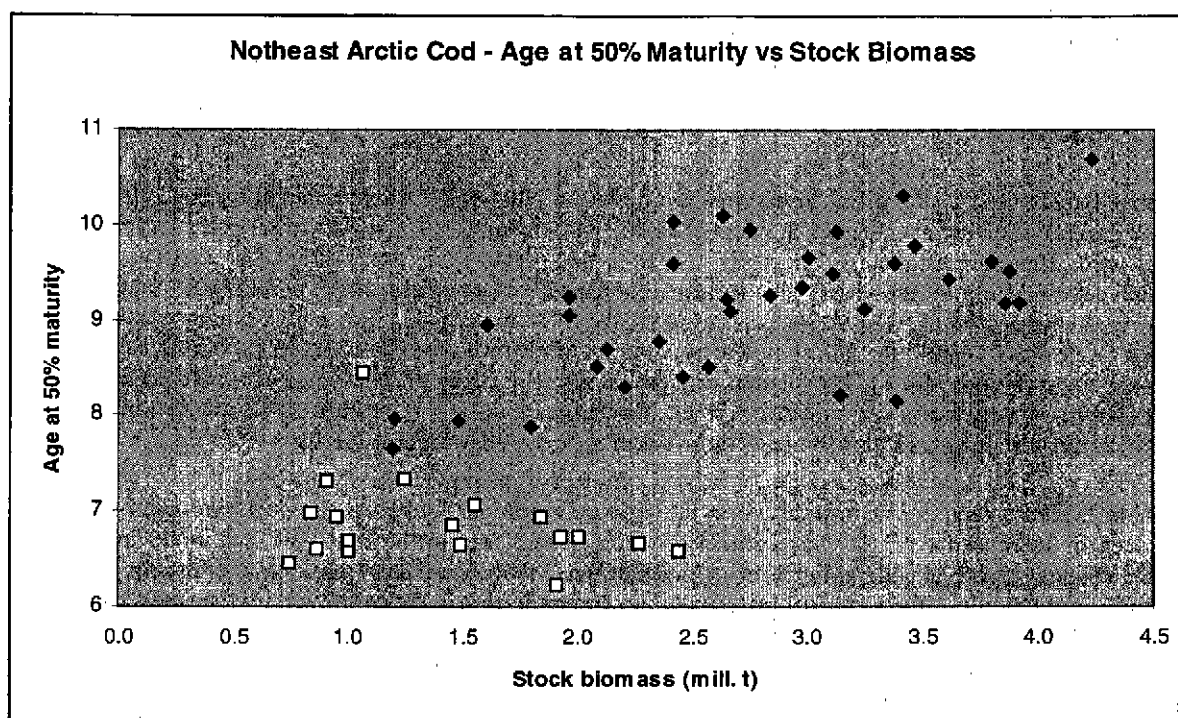


Figure 11.2.2 Northeast Arctic cod. Age at 50% maturity vs. total stock biomass.

### 11.2.3 Recruitment

Recruitment is highly variable in this stock (at age 3, the strongest year class in the time series is 13 times stronger than the weakest). Good recruitment occurs only in years with relatively high temperatures. Several studies have addressed the variability in cod recruitment (e.g., Sætersdal and Loeng 1987; Ellertsen *et al.* 1989; Sundby *et al.* 1989; Nilssen *et al.* 1994). After these analyses were completed, new data sources have become available. Thus, a new study is currently in progress to incorporate all available data to construct a mathematical model to predict recruitment. The time series of survey indices and environmental variables available for such a study are listed in ICES (1998b).

It should also be noted that the recruitment numbers are calculated by a VPA where cannibalism is included only for the period 1984–1997, and thus there is an inconsistency in the time series. Russian qualitative stomach content data are available back to the 1940s (Ozhigin *et al.* 1996), and those could be utilised to model cannibalism in the period prior to 1984.

### 11.2.4 Natural mortality

Pálsson (1994) gives an overview of all the predators on this cod stock. The main predators on cod are minke whales (Folkow *et al.* 1997), harp seals (Nilssen *et al.* 1999), and cod (cannibalism, Bogstad *et al.* 1994; Bogstad and Mehl, 1997). The effects of cod-cannibalism is already included in the assessment (ICES, 1999b). There are indications that predation by minke whale, harp seal and cod on cod is inversely related to the abundance of the capelin (Haug *et al.* 1997; Nilssen *et al.* 1999; ICES 1999b). The estimates of consumption of cod by predators should be compared to the biomass removed by the natural mortality ( $M = 0.2 + \text{cannibalism}$ ) presently used to investigate whether this value of natural mortality is appropriate.

Also, there are indications of increased natural mortality at older ages. Attempts to quantify this should be made. The work of Beverton *et al.* (1994) could be utilised here.

## 11.3 Reference points

### 11.3.1 Present management objectives

This stock is shared between Norway and Russia and the TAC is set by the Mixed Russian-Norwegian Fishery Commission. The ACFM advice has been the basis for the TAC and in 1990–1996 the Commission was careful not to go outside the options ACFM provides. However, in 1997 and 1998 the quota was set at a considerably higher level than the maximum catch advised by ACFM (654 000 vs. 514 000 tonnes for the 1998 quota and 480 000 vs. 360 000 tonnes for the 1999 quota). The Commission states that  $F_{med} = 0.46$  (arithmetic mean, age groups 5–10) should not be exceeded and that the spawning stock should be kept above 0.5 million t (MBAL). However, because recent  $F$  levels have been well above  $F_{med}$ , the current strategy is a gradual reduction towards  $F_{med}$ . The Arctic Fisheries Working Group (ICES, 1999) suggests the following values for the reference points in order to comply with the precautionary approach:  $B_{pa} = 0.5$  million t,  $B_{lim} = 0.112$  million tonnes (lowest observed),  $F_{pa} = 0.42$  (5<sup>th</sup> percentile of  $F_{loss} = 0.70$ ). It is interesting to note that  $F_{max}$  is lower than  $F_{pa}$ , implying that reducing  $F$  below  $F_{pa}$  will increase the yield.

### 11.3.2 Previous studies

Both Jakobsen (1993) and Nakken *et al.* (1996) argue that a spawning stock of 0.5 million tonnes should be viewed as a limit reference point, i.e., the spawning stock should not be allowed to decrease below this level. Nakken *et al.* (1996) argue for a target fishing mortality in the range from 0.20–0.40. In these studies, stochastic recruitment was not taken into account. In runs with the multispecies model AGGMULT, Tjelmeland (1995) obtained an optimal fishing mortality for cod of about 0.4, taking recruitment variability into account.

### 11.3.3 Simulations carried out here

We investigated the performance of various harvest control rules, assuming constant biological parameters and variable recruitment.

The initial values (and uncertainty of these) for stock number at age as well as the values for weight at age in the stock and in the catch, maturity at age, fishing pattern and natural mortality were the same as used for 1998 in the last assessment (ICES 1999). Simulations were carried out using the same spreadsheet as used by the AFWG for medium-

term risk analysis. A Beverton-Holt spawning stock-recruitment relationship  $R = \frac{\alpha SSB}{1 + \frac{SSB}{K}}$  was fit to the spawning

stock and recruitment (age 3) data from the latest assessment (ICES 1999) (Fig 11.3.1). Values of  $\alpha = 4.42$  and  $K = 212$  were found. A CV of 0.7 on log scale was found to give a similar variability in the recruitment as in the historical time series (sample plot given in Figure 11.3.2). To avoid recruitment outside the range given by the time series, the maximum recruitment was set to 2 billion individuals at age 3, which is slightly higher than the highest value in the time series (1818 million, the 1970 year class). The effect of using a Ricker spawning stock-recruitment relationship as well as a relationship including periodicity should also be investigated. Also, simulations should be made using the stock-recruitment plot given by Ulltang (1996), which is based on the maturity ogive calculated by Jørgensen (1990). An updated version of the plot is given in Figure 11.3.3. It should be noted that the spawning stock biomasses in Figure 11.3.3 are considerably lower than Figure 11.3.1.

As the uncertainty in the assessment of this stock has been large in recent years (Nakken 1998), it was decided to investigate the effect of assuming an assessment error or not.

Assessment uncertainty was represented by a standard error in reference  $F(F_{5-10})$  of 0.3 on a log scale, the same value as the Arctic Fisheries Working Group used in the medium-term projections carried out at its last meeting. This can also be considered as representing uncertainty in growth and natural mortality.

The harvest control rule used was of the form

$F = F_{pa}$  for  $SSB > B_{pa}$ ,  $F = (SSB - B_{lim}) * F_{pa} / (B_{pa} - B_{lim})$  for  $B_{lim} < SSB < B_{pa}$ ,  $F = 0$  for  $SSB < B_{lim}$ . This is the same kind of rule as illustrated in Fig 7.3.2, but with  $F = 0$  at  $B_{lim}$ . Simulations were made for  $F_{pa}$  values of 0.24, 0.46 and 0.78, corresponding to  $F_{max}$ ,  $F_{med}$  and  $F_{high}$ , and for  $B_{pa}$  values of 0.5 and 0.112 (=  $B_{lim}$ ) million tonnes respectively.

The effect of the maximum annual change in TAC to 25% was also tested.

In all simulations, the maximum catch was set to 1.5 million tonnes and the maximum value of the realised  $F$  was set to 1.5. A 100 year simulation period was used, and for each combination of harvest control rule and recruitment function, 1000 simulations were performed.

The results of the simulations are summarised in Table 11.3.1.

F	BPA	Assessment error (SE on log scale)	Limit on relative change in catch from one year to next	Average Yield	Average value of maximum catch in period	Average value of minimum catch in period
0.24	0.500	0.3	No limit	0.682	1.466	0.180
0.46	0.500	0.3	No limit	0.601	1.419	0.149
0.78	0.500	0.3	No limit	0.497	1.300	0.065
0.24	0.500	0.3	25%	0.682	1.302	0.232
0.46	0.500	0.3	25%	0.612	1.259	0.216
0.78	0.500	0.3	25%	0.514	1.228	0.140
0.24	0.112	0.3	No limit	0.687	1.471	0.182
0.46	0.112	0.3	No limit	0.597	1.415	0.185
0.78	0.112	0.3	No limit	0.420	1.109	0.020
0.24	0.500	0	No limit	0.689	1.009	0.211
0.46	0.500	0	No limit	0.604	0.937	0.304
0.78	0.500	0	No limit	0.493	0.871	0.214
0.24	0.112	0	No limit	0.689	1.007	0.223
0.46	0.112	0	No limit	0.603	0.940	0.329
0.78	0.112	0	No limit	0.419	0.743	0.189

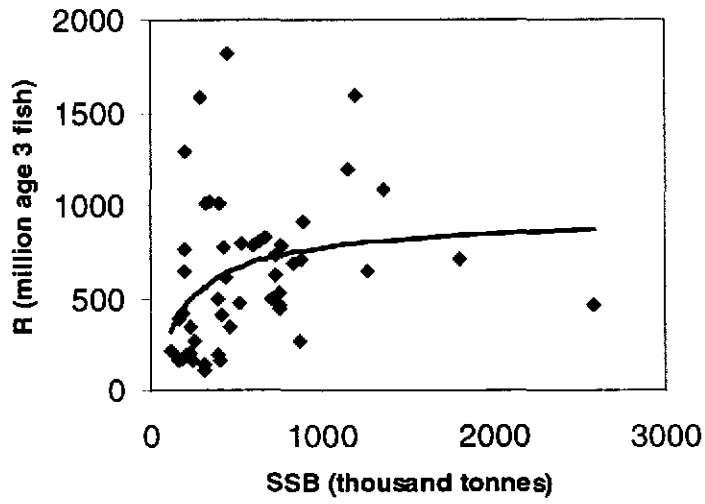
**Table 11.3.1** Simulation results. All biomasses and yields are given in million tonnes.

The lowest fishing mortality gives the highest yield in all cases. Results not presented in the table indicate that the highest yield is obtained for fishing mortalities less than 0.24. At such values, the spawning stock will be quite high. Including density-dependent effects would probably alter this picture. Reducing  $B_{pa}$  from 0.5 to 0.112 million tonnes reduces the yield for  $F = 0.78$ , but has no effect for the other two  $F$ -values. The average yield is hardly affected by the assumption about assessment error, but the variation range of the catch is considerably reduced if no assessment error is assumed. Limiting the year-to-year variability in the catch to 25% also reduces the variation range of the catch, but not as much as when no assessment error is assumed.

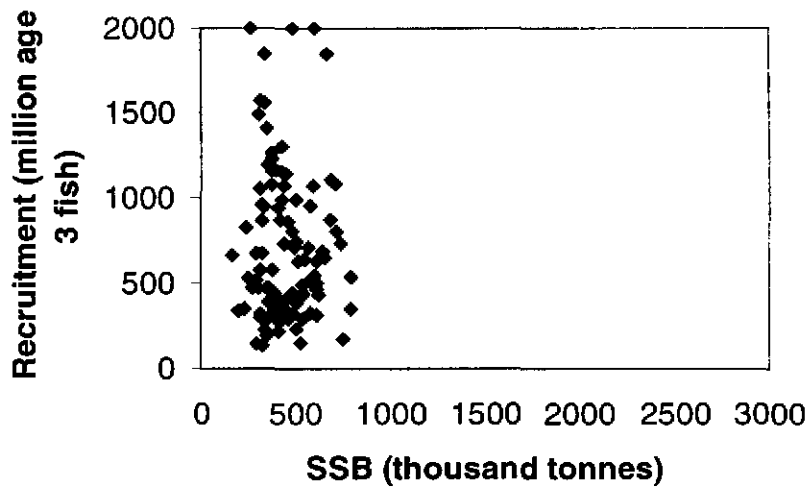
The results give relatively low values of yield compared to historical levels, mainly because the data used in the simulations are taken from a period with high cannibalism (high natural mortality on the youngest age groups) and low growth. The minimum catches are in some cases very low, even at moderate fishing mortalities, and the historical evidence indicate that this should be possible to avoid with a reasonable management strategy.

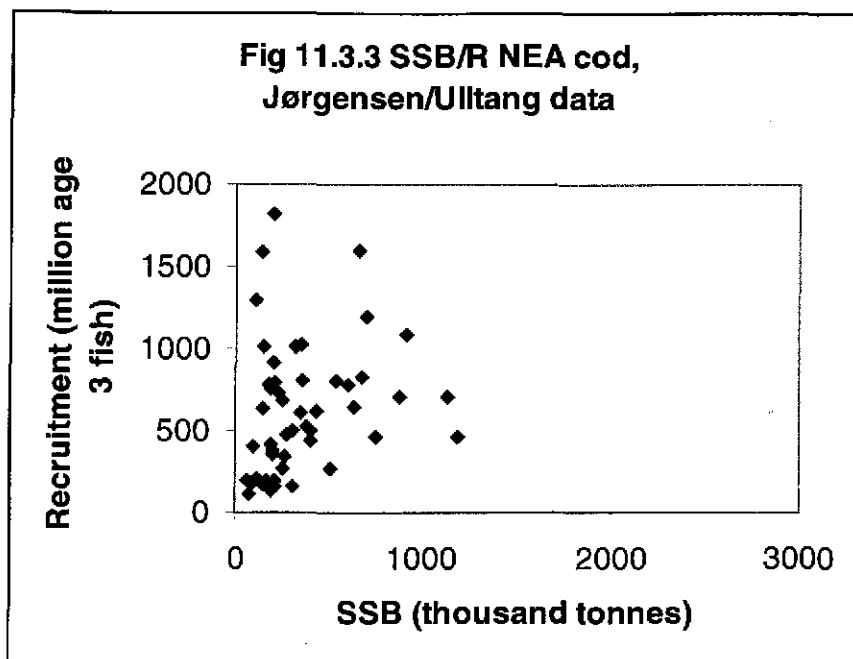


**Fig 11.3.1 SSB/R, NEA cod, 'ICES' time series**



**Fig 11.3.2 Simulated SSB/R relationship**





#### 11.4 Inclusion of multispecies interactions and bioeconomy in studies of reference points

As mentioned in Section 11.2, growth, natural mortality and recruitment are strongly dependent on multispecies interactions. Thus, studies of management strategies will involve modelling of multispecies interactions (in particular cod-capelin-herring interactions), and it will also be useful to include bioeconomic considerations. An overview of the models available for such studies is given in the last COMFIE report.

Harvest control rules are being investigated also for two other systems which are dominated by three species: The Baltic Sea (cod-herring-sprat) (Gislason, WP 10) and Icelandic waters (cod-capelin-shrimp) (Jakobsson and Stefánsson, 1998). These systems are also strongly affected by environmental variations. Co-operation between the groups carrying out such studies should be encouraged.

## 12 ASSESSMENT AND PREDICTION METHODS

### 12.1 Introduction

This section contains a description of modelling efforts underway for improving assessments and predictions in Boreal systems, as well as some concerns regarding the estimation of fishing mortality in VPA and the issue of how to correct some estimation biases in the parameters.

Most assessment methods in current use are based only on single species considerations and emphasise the mortality process, particularly the effect of fishing on the stock numbers. In particular, even from a single-species point of view, these models usually do not take into account spatial variation, variations in natural mortality or dependency of growth on specific factors such as density of the species in question or the abundance of prey. In some cases assessments have been questioned for these reasons and it is clear that more complex models are needed in order to investigate whether these issues are important or not. Since biological interactions may also be quite important, there is a need for increased complexity to describe the processes involved but it has not been demonstrated that this provides better estimates of abundance.

The Bormicon and Flexibest models have been developed to take these processes into account through increasing levels of model complexity. Although the models can potentially explain much more complex ecosystem dynamics in this manner there is as yet no indication of whether they can provide better assessments of the resources.

### 12.2 Bormicon

In the Arcto-Boreal systems off Canada, Iceland and in the Barents Sea it is well-known that growth of cod is quite variable, due to various factors including temperature, predator density (Millar, 1992) and prey abundance (Steinarsson

and Stefánsson, 1999), but the relative importance of each factor varies from system to system. Similarly, total consumption of prey by predators appears to be a considerable fraction of biomass (Bogstad and Mehl, 1997, Pálsson, 1983, Magnússon and Pálsson, 1989 and 1991) and in fact prey mortality seems to be measurably affected by predator abundance in some cases (Stefánsson *et al.*, 1999). These factors have been taken into account in Icelandic waters by using simple bioeconomic multispecies models (Baldursson *et al.*, 1996). In addition to species interactions, it has been seen that cannibalism can also be an important issue for cod in general (Bogstad *et al.*, 1994) and similar *intra*-species concerns apply to other species including herring (e.g., Jakobsson *et al.*, 1993). It is quite common to see spatial variation in growth (WP11) along with migrations between dissimilar regions, where different fishing gear is applied within these regions.

In the Barents Sea these interactions are well known and Bogstad *et al.* (1997) describe a multispecies model which takes many of the above concerns into account. Results from the model have recently been used to advise on capelin catches by taking into account the predation by cod (Anon. 1998).

In light of these variations, it is reasonably clear that a simple virtual population analysis (VPA, Gulland 1964) or even a multispecies VPA (MSVPA, Helgason and Gislason, 1985) is not adequate to describe the processes which are considered important. Further, existing models which do take some of these variations into account need revision. Hence, a proposal was set forward to how these might be modelled (Stefánsson and Pálsson, 1998) and a modelling framework has been developed with the purpose of making a basis for such modelling exercises (Stefánsson and Pálsson, 1997). This model, the BOREal MÍgration and CONsumption model, or BORMICON, forms a basis to model natural and fishing mortality processes, along with the migration, growth and maturation processes. In order to estimate the various unknown parameters in the model, it can incorporate widely disparate data sets such as age readings, total catches, total acoustic biomass, survey indices by age, length distributions or mean length at age in a likelihood function. Details can be found in the user's manual and programmer's manual (Stefánsson *et al.*, 1998a-b).

### 12.3 Flexibest

Work is in progress at IMR in Norway to develop a new assessment tool, the first version of which is designed specifically for Northeast arctic cod. The model can be seen as an extended application of the principle of statistical catch at age models (Fournier and Archibald, 1982) or as a modification of BORMICON (Stefánsson and Pálsson 1997, see also Section 12.2).

The model is designed to provide an alternative to the VPA-tuning models traditionally used for many stocks in the ICES area. The primary purpose was to obtain greater flexibility as to the choice of model assumptions and the use of the data by offering a range of opportunities to formulate relations between the stock and the observations, and the internal structure of the stock and mortality matrices. This would allow both more appropriate use of existing data, and make the stock estimate less dependent on data for which the quality is questionable. Moreover, it should allow for incorporating background information on e.g., growth vs. climate, on fish behaviour, changes in the way the fisheries are performed etc., in model formulations.

As for all statistical catch at age models, the core is a population matrix where initial numbers are specified for each cohort, and the time course of the abundance is specified by mortality models. Modelled catches can be derived from this population and compared with those reported. Furthermore, by making assumptions about the relation between the population abundance and measurements of it (i.e., survey indices etc.), in terms of e.g., catchability models, modelled values corresponding to these observations can be derived.

The Flexibest model differs from most of the statistical catch at age models by describing the population matrix in terms of both length and age, and defining mortalities and catchabilities, as well as the maturity ogive, in terms of length instead of age. The background for this is that the growth rate is known to change considerably over time for this stock, which would make the various ages vulnerable to the fishery to a varying extent, and thus violate the hypothesis of separable fishing mortality. This is illustrated in Figure 12.3.1. Moreover, selection patterns are given separately for each fleet, each being weighted according to their share of the total catch. Growth is modelled assuming a von Bertalanffy growth equation, with the growth rate  $k$  varying from year to year. Fish is moved from one length class to the next according to a transition matrix conditional of the growth rate. The immature and mature fish are treated as separate populations. Most of these elements are in common with the BORMICON model, and large parts of the code have been taken over from BORMICON. The exception is that Flexibest consistently derives catches according to externally modelled fishing mortalities, while BORMICON normally treats the fishery as another predator.

Work on the model is still in progress. Specific modelling of predation mortality (cannibalism) based on consumption data is to be implemented. So far, the objective functions relating modelled and observed values are still incomplete and

not well founded statistically, and optimisation of the objective functions is to be improved. Routines for presenting the results are still primitive.

Preliminary retrospective runs (see Figure 12.3.2) indicate that the estimates for the last assessment year are unstable. For earlier years, the results seem quite consistent. Upweighting the Norwegian Barents Sea trawl survey by a factor of 10 did not change this overall picture. The mortalities deviate somewhat from those obtained by the XSA (ICES 1998), which is the standard assessment tool for this stock, but the overall trends are the same, as shown in Figure 12.3.3. It should be noted that some of the data used by Flexibest were revised after the XSA assessment.

The main difference between Flexibest and most other statistical catch at age models is the modelling of fleetwise separable fishing mortalities by length. The yearly fishing patterns at age emerging from the preliminary Flexibest assessment is shown in Figure 12.3.4, indicate that this may lead to quite large year-to-year variations in the perceived selections at age for the intermediate ages.

The results of upweighting some of the data sources are presented in Figure 12.3.5. It is seen that there is much similarity in the trends of the fishing mortality indicated by each survey; but in some periods, the surveys give different pictures.

The model was presented at a workshop in Bergen, December 1998. A more detailed description of the model will appear in the report from that meeting (Ulltang 1999).

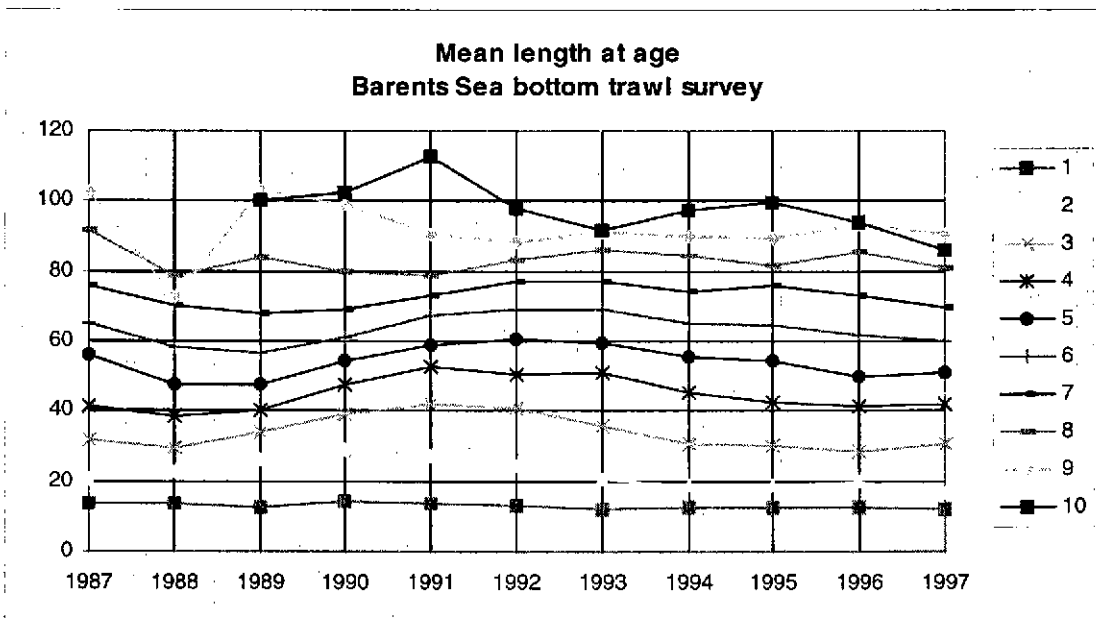
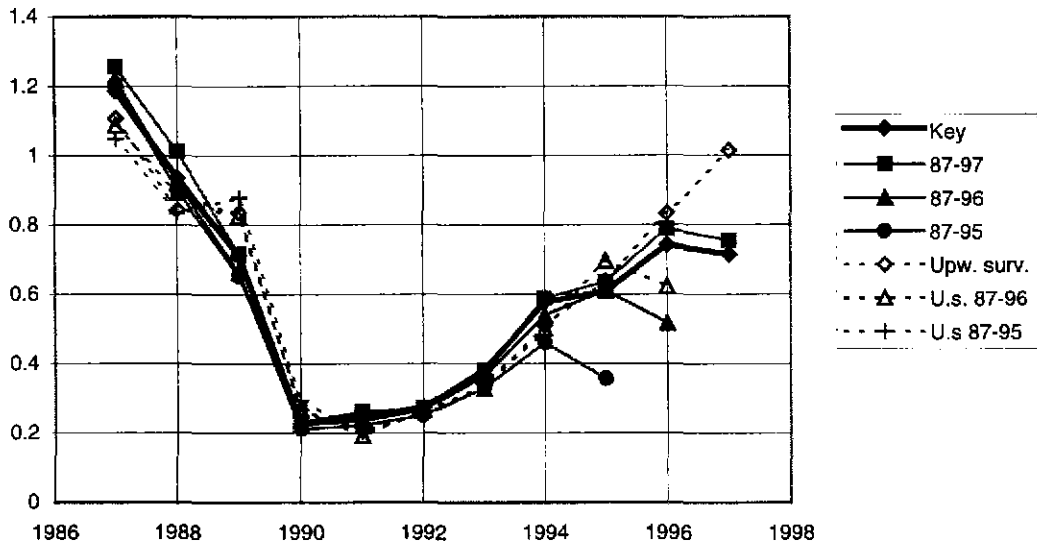


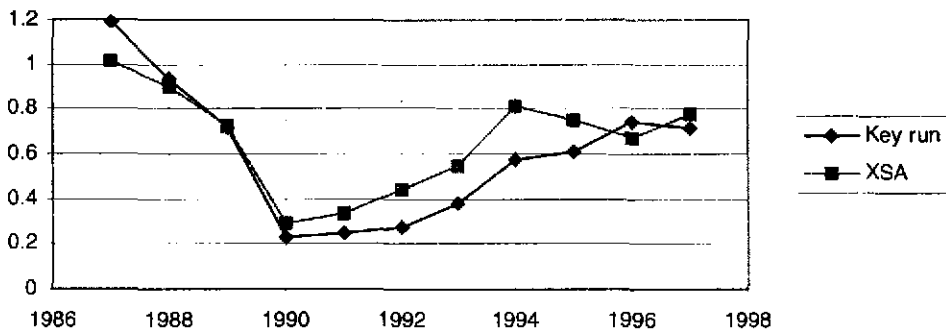
Figure 12.3.1 Yearly mean length at age for North East arctic cod, as observed in the Norwegian bottom trawl survey in the Barents Sea.

**Retrospective analysis  
F 5-10**



**Figure 12.3.2.** Preliminary stock estimate of Northeast Arctic cod using Flexibest. Heavy line: Key run using full data set, including surveys in 1998. Whole lines: Runs using only data from the years indicated. Broken lines: Runs where the survey data were upweighted by a factor of 10.

**F 5-10**



**Figure 12.3.3** Average F 5–10 according to a preliminary run with Flexibest, compared to the most recent assessment using XSA.

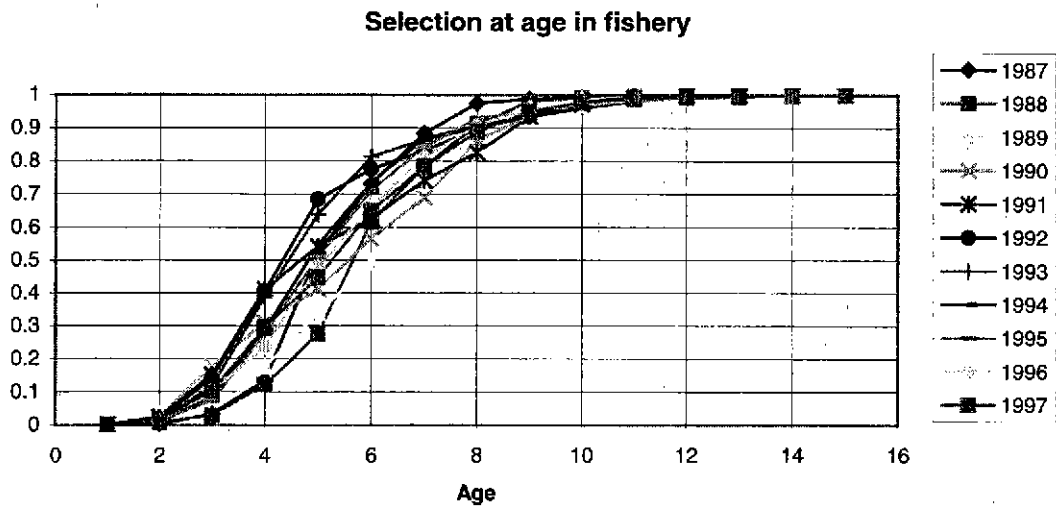


Figure 12.3.4 Yearly selection at age as derived from a preliminary assessment for North East arctic cod using Flexibest. A fleetwise, logistic selection at length with constant parameters was assumed.

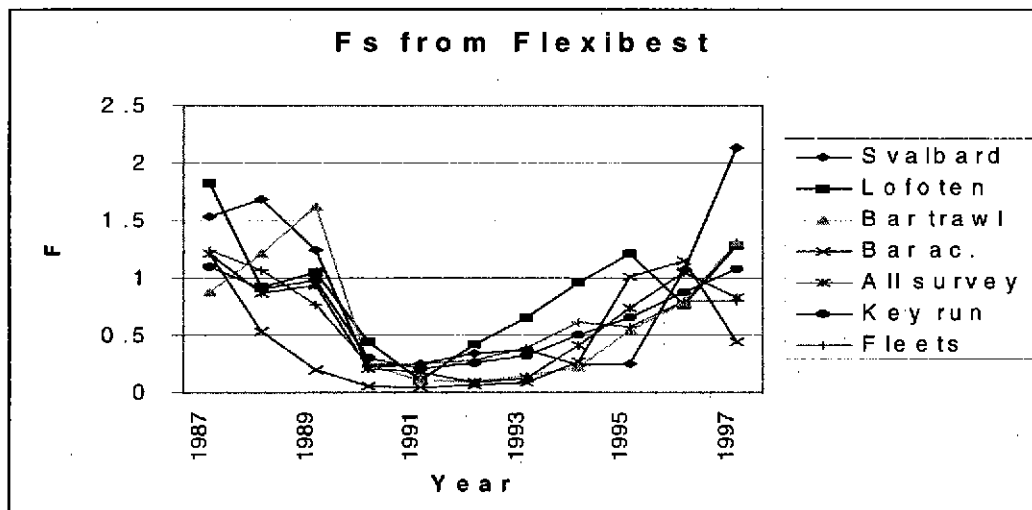


Figure 12.3.5 Fishing mortalities from runs with Flexibest where the weighting of different data sources is varied. In the runs labelled 'Svalbard', 'Lofoten', 'Bar trawl' and 'Bar ac' the survey in question is weighted by a factor 100 compared to the other data sources. In the 'All survey' run, all surveys are weighted by a factor 100. The 'Key run' is essentially the same as in Figure 12.3.3, while in the 'Fleets' run, all surveys and fleets are weighted equally.

## 12.4 Estimating Fishing Mortality on the Oldest Age

### 12.4.1 Introduction

Virtual population analysis requires a fishing mortality rate for the oldest age group in each year to initialize the backward solution of these cohorts. The oldest age group may be either an individual age class or a plus group. Consider the notation  $F_{A,t} = \alpha_t F_{A-1,t}$ , where A represents the oldest age group. Estimation of  $\alpha$  for all t is equivalent to estimation of fishing mortality on the oldest age. Alternatively, a common  $\alpha$  may be estimated for a block of years, which is equivalent to assuming that the exploitation pattern at age for age group A relative to age group A-1 is constant over the years in that block. Another approach is to assume known fixed values for  $\alpha$  for all t or for a block of years.

In general, simultaneous estimation for the oldest age group of abundance index calibration coefficient, natural mortality and all the  $\alpha$ 's is problematic because these parameters are highly correlated. Often, the natural mortality is assumed known for all age groups, including the oldest age group. Notwithstanding this simplification, simultaneous estimation of  $\alpha$ 's and abundance index calibration coefficient may still be problematic.

## 12.4.2 Simulation Experiment

Results from a simulation experiment were presented in WP14 demonstrating the benefit of having a recruitment index relative to having an index covering a range of ages when estimating alphas. This simulation experiment considered observation errors in both catch at age as well as the tuning indices and focused on the importance of composite versus individually aged indices. This work was extended during the meeting to focus on three questions: 1) Is it better to use a plus group or to use an oldest true age? 2) Does the inclusion of an index for the oldest age improve the results? 3) How does the VPA respond to estimating extra alphas, the correct number of alphas, or only one alpha when the alphas do in fact change.

### 12.4.2.1 Methods

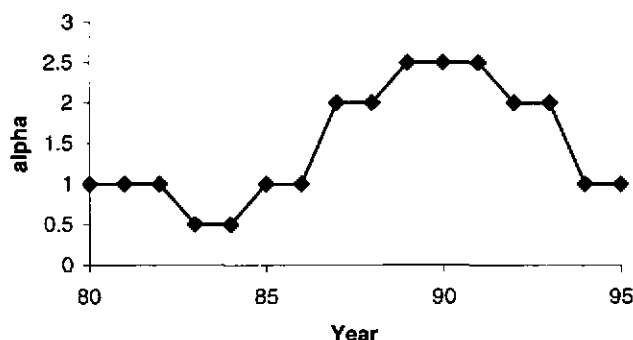


Figure 12.4.1 Pattern of  $\alpha$  used to simulate data.

A simulated population was created with the following characteristics. There were sixteen years, denoted 1980 to 1995, and 50 ages. The age data were either summed to form a 10 plus group or else ignored data for ages 11 and older. The 10 plus group contained a fairly large portion of the total biomass of the population (up to 35%). Natural mortality was 0.3 for all ages and years. Lognormal noise was added to a Shepherd (1982) stock recruitment equation so that resulting recruitment varied between 7 and 20 million fish annually. Weight at age followed the von Bertalanffy relationship exactly with the true plus group weight input to the VPA if needed. The fishing mortality matrix was constructed so that early in the time period  $F$  was low and focused on young fish and then increased both in level and age of full selectivity over time. Catch at age was input correctly to the VPA. Indices were created as the total biomass at the start of the year individually for ages 1 to 10 or 10 plus. Lognormal error was added to the tuning indices with a CV of 0.2 to create 50 Monte Carlo simulations. The alphas varied by year as seen in Figure 12.4.1.

The VPA used was Fadapt (Restrepo 1996), modified to allow blocks of alphas to be estimated. The scenarios examined considered estimating an alpha for every year, seven  $F$  ratios were estimated in blocks that match the true values, or a single  $F$  ratio was estimated for years 1980 to 1994. The terminal year  $F$  values estimated for all scenarios were ages 4, 8, 9 and 10, thus estimating the 1995  $F$  ratio independently of all others. The correct selectivity pattern for the terminal year was entered in the VPA. Results were measured in terms of the management benchmarks: the oldest age (10 or 10 plus) abundance in the terminal year, the depletion of the oldest age group ( $N_{oldest,95}/N_{oldest,80}$ ),  $F_{0.1}$  in the terminal year, and the total allowable catch (TAC) in 1996 under  $F_{0.1}$  for each Monte Carlo simulation.

### 12.4.3 Results and Discussion

Management benchmarks for the 50 Monte Carlo simulations of the 12 scenarios are summarised in Figure 12.4.2. The box for each scenario shows the inner 50<sup>th</sup> percentile while the whiskers give the range of observations from the 50 VPAs. Note in the Figure that the true values are given separately for the cases when the last age is a plus group and when the last age is a true age, denoted  $Y_{true}$  and  $N_{true}$ , respectively. When a plus group is used, the estimation of alphas for all years improves the results relative to when only seven years are estimated in terms of both location and spread in general. However, when the last age is a true age, the estimation of alphas for all years causes large amounts of bias in the results while the estimation of seven alphas produces relatively good estimates. Estimating a single alpha when there are in fact many causes large amounts of bias in the results, especially when the oldest age is a true age. The

use of a plus group appears to stabilise the estimation of alphas in VPA, at least when there are many fish still remaining in this group. This large number of fish in the oldest age group may account for the differences seen here and those presented in Myers and Cadigan (1994) who found that alphas were poorly estimated and recommended estimating only a single alpha. The inclusion of additional data in the form of a tuning index for the oldest age improves the results by decreasing the inner 50<sup>th</sup> percentile range and having an average closer to the true value in almost every case.

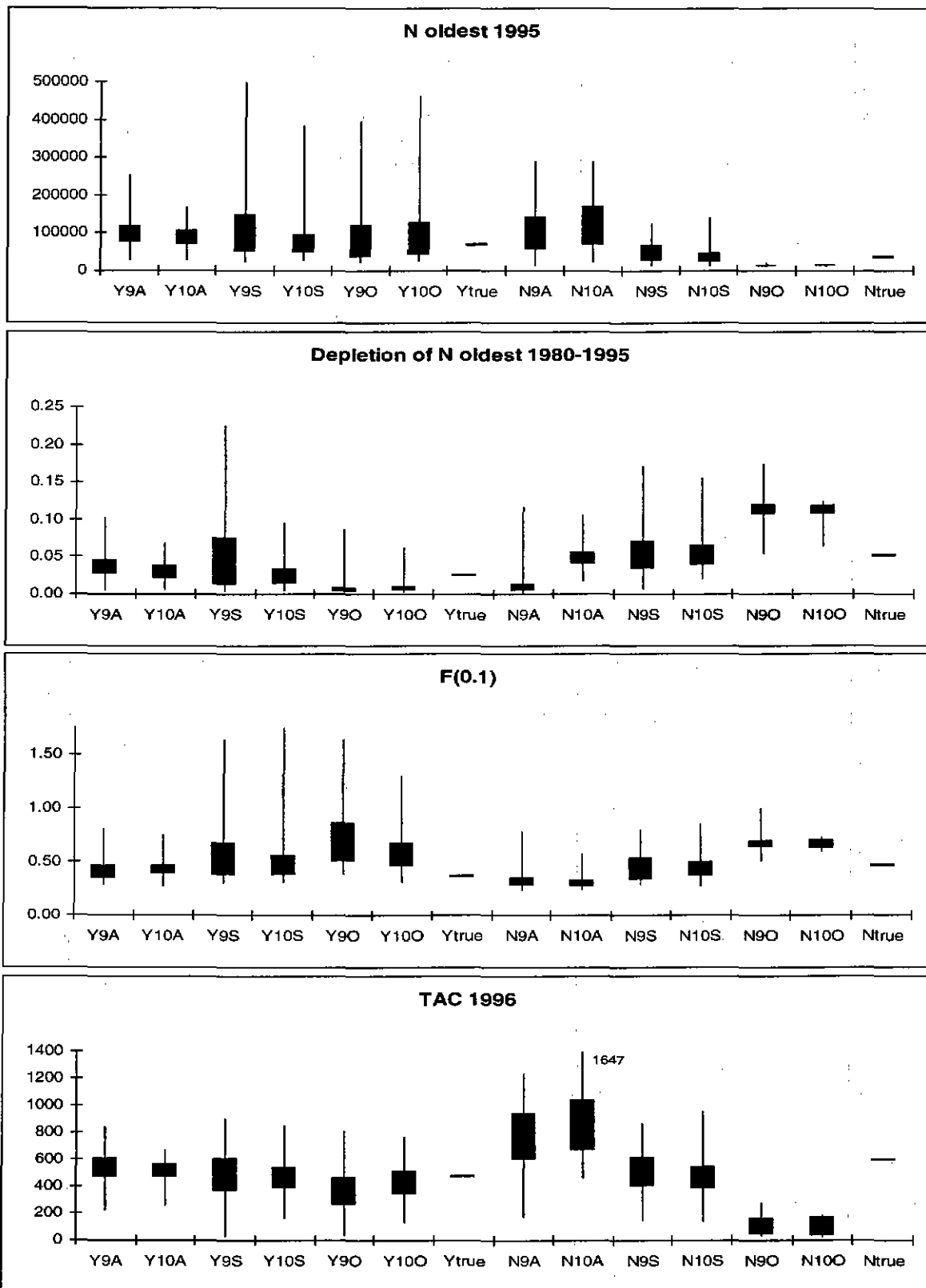


Figure 12.4.2 Monte Carlo results of simulation experiment. Y = plus group, N = last true age (Ytrue and Ntrue are true values for plus group and last true age simulations, respectively); 9 = age-specific indices to age 9; 10 = age 10 or 10+ index also used; A = all years alphas, S = seven alphas, O = one alpha.



#### 12.4.4 Georges Bank Haddock Example

The alpha ratio method was applied in WP 15 to data for eastern Georges Bank haddock where the oldest age group was 9+. The results from 3 models were compared where a common  $\alpha$  was estimated for the entire time period used, 1986–97, while  $\kappa$ , the calibration coefficients for the abundance indices, were a) freely estimated for age 9+, b) were constrained to be equal for ages 8 and 9+, and c) were constrained to be equal for ages 7 to 9+. These results were extended to examine the estimation of annual  $\alpha$ 's and to constrain  $\kappa$  over more ages.

The results in Table 12.4.1 show that the estimate of  $\alpha$  when  $\kappa$  was freely estimated for age 9+ had large variance and bias and was confounded with  $\kappa$ . The statistics for  $\alpha$  improved considerably when the estimate of  $\kappa$  was constrained to be equal for ages 8–9 and the estimate of  $\kappa$  displayed a better pattern over ages. Constraining  $\kappa$  to be equal over more age groups did not result in any further significant effect. To estimate individual annual  $\alpha$ 's, it was necessary to constrain  $\alpha$  to be constant for 1993–95. The individual estimates of  $\alpha$  were very imprecise. Their precision was not appreciably improved even when  $\kappa$ 's were constrained on older ages.

**Table 12.4.1.** Statistics for alpha ratios and calibration coefficients.

	Est.	rel. err.	rel. bias	Est.	rel. err.	rel. bias	Est.	rel. err.	rel. bias	Est.	rel. err.	rel. bias	
$\alpha$ (year)													
all years	1.96	0.63	0.21	0.49	0.21	0.02	0.47	0.20	0.02	0.40	0.20	0.02	
Calibration coefficients (age)													
1	0.14	0.25	0.02	0.13	0.26	0.03	0.13	0.26	0.03	0.13	0.27	0.03	
2	0.37	0.24	0.01	0.36	0.25	0.02	0.36	0.25	0.02	0.36	0.25	0.03	
3	0.71	0.23	0.01	0.69	0.24	0.02	0.69	0.24	0.02	0.67	0.25	0.03	
4	0.64	0.23	0.01	0.61	0.24	0.02	0.61	0.24	0.02	0.59	0.25	0.03	
5	0.75	0.23	0.01	0.71	0.24	0.02	0.71	0.24	0.03	0.69	0.25	0.03	
6	0.61	0.23	0.01	0.57	0.24	0.03	0.56	0.24	0.03	6–9	0.68	0.25	0.03
7	0.92	0.24	0.01	0.82	0.26	0.03	7–9	0.85	0.25	0.03			
8	0.87	0.24	0.00	8–9	0.91	0.26	0.03						
9	3.81	0.47	0.03										
	Est.	rel. err.	rel. bias	Est.	rel. err.	rel. bias	Est.	rel. err.	rel. bias	Est.	rel. err.	rel. bias	
$\alpha$ (year)													
1986	0.60	1.53	1.07	0.22	1.30	0.66	0.22	1.27	0.72	0.23	1.31	0.80	
1987	0.56	1.51	0.87	0.29	1.08	0.30	0.30	1.04	0.37	0.33	0.99	0.36	
1988	1.62	1.63	1.39	0.63	0.86	0.17	0.64	0.84	0.23	0.60	0.83	0.22	
1989	0.61	1.44	0.84	0.22	0.86	0.12	0.22	0.84	0.19	0.21	0.81	0.17	
1990	0.65	1.62	1.27	0.22	0.78	0.08	0.23	0.78	0.13	0.21	0.75	0.12	
1991	2.56	1.95	2.93	0.82	0.58	0.04	0.85	0.58	0.01	0.68	0.59	0.00	
1992	0.51	1.30	0.51	0.25	0.90	0.15	0.25	0.88	0.23	0.25	0.83	0.21	
1993–5	0.54	1.10	0.69	0.17	0.46	0.00	0.17	0.46	0.03	0.15	0.45	0.02	
1996	1.13	1.21	0.56	0.54	0.92	0.26	0.55	0.91	0.32	0.47	0.92	0.32	
1997	6.05	1.24	0.80	2.45	0.52	0.11	2.52	0.52	0.09	2.05	0.52	0.08	
Calibration coefficients (age)													
1	0.14	0.26	0.00	0.13	0.26	0.03	0.13	0.26	0.02	0.12	0.27	0.02	
2	0.37	0.25	0.00	0.35	0.25	0.03	0.35	0.25	0.02	0.34	0.26	0.02	
3	0.70	0.24	0.00	0.66	0.24	0.03	0.66	0.24	0.02	0.63	0.26	0.02	
4	0.63	0.24	-0.01	0.58	0.24	0.03	0.58	0.24	0.02	0.56	0.25	0.02	
5	0.74	0.24	-0.01	0.66	0.25	0.04	0.67	0.24	0.02	0.63	0.25	0.02	
6	0.60	0.25	-0.02	0.52	0.26	0.05	0.52	0.24	0.02	6–9+	0.57	0.26	0.02
7	0.93	0.28	-0.02	0.75	0.29	0.08	7–9+	0.74	0.27	0.04			
8	0.93	0.32	0.00	8–9+	0.71	0.32	0.15						
9+	2.42	1.01	0.33										

The population abundance estimates did not change much for any of these options except for age 9+ (Figure 12.4.3). This insensitivity seems consistent with the observation by Myers and Cadigan (1994).

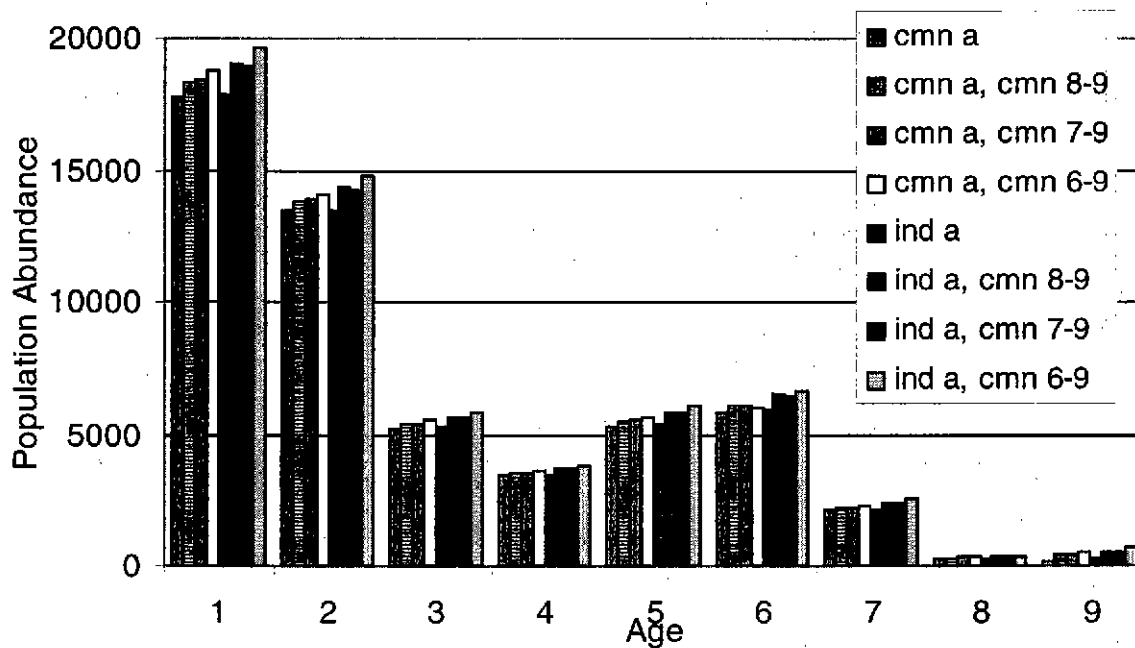


Figure 12.4.3 Population abundance estimates at the beginning of 1998 for various options of estimating  $\alpha$  and  $\kappa$ .

#### 12.4.5 Catchability at the Oldest Ages - Laurec-Shepherd tuning and XSA

Estimates of fleet or survey catchability at the oldest ages in a VPA based assessment are directly dependent on the terminal population or F values used to initialize the underlying VPA. Catchability at the oldest age cannot be determined without the use of additional constraints. At successively younger ages the rigidity imposed on the estimates of population abundance by convergence of the VPA reduces the dependence and the catchabilities are considered to be increasingly independent of the terminal estimates.

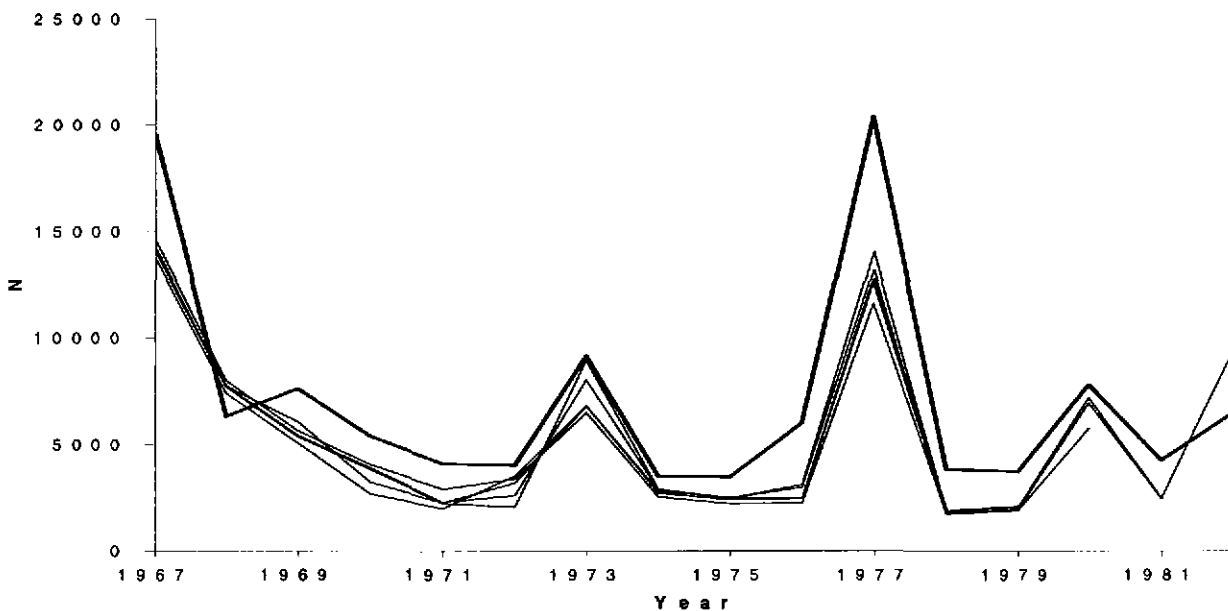
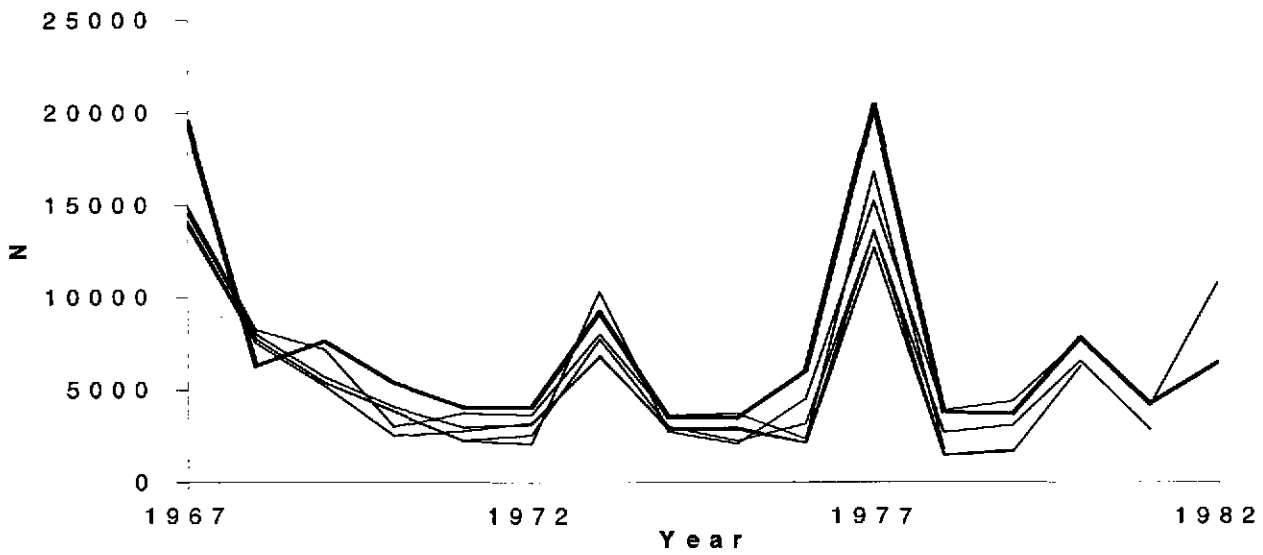
The Laurec-Shepherd (Pope and Shepherd, 1985) *ad hoc* tuning procedure introduces an additional constraint by making the assumption that F on the oldest age could be calculated as a proportion of the average of the F of the preceding ages in the same year. The proportion and the range of ages are user defined.

Extended survivors analysis (XSA Shepherd, 1992) uses an alternative approach to reduce the number of parameters estimated by imposing the constraint that fleet catchability is constant (independent of age) above a certain age. The age (constant for all fleets) is user-defined with a default imposed to ensure that, at the very least, the catchability of the oldest true age is fixed to that of the preceding age. For each fleet, the catchability value estimated at the specified age, is used with the fleet CPUE data to derive population abundance estimates for all older ages.

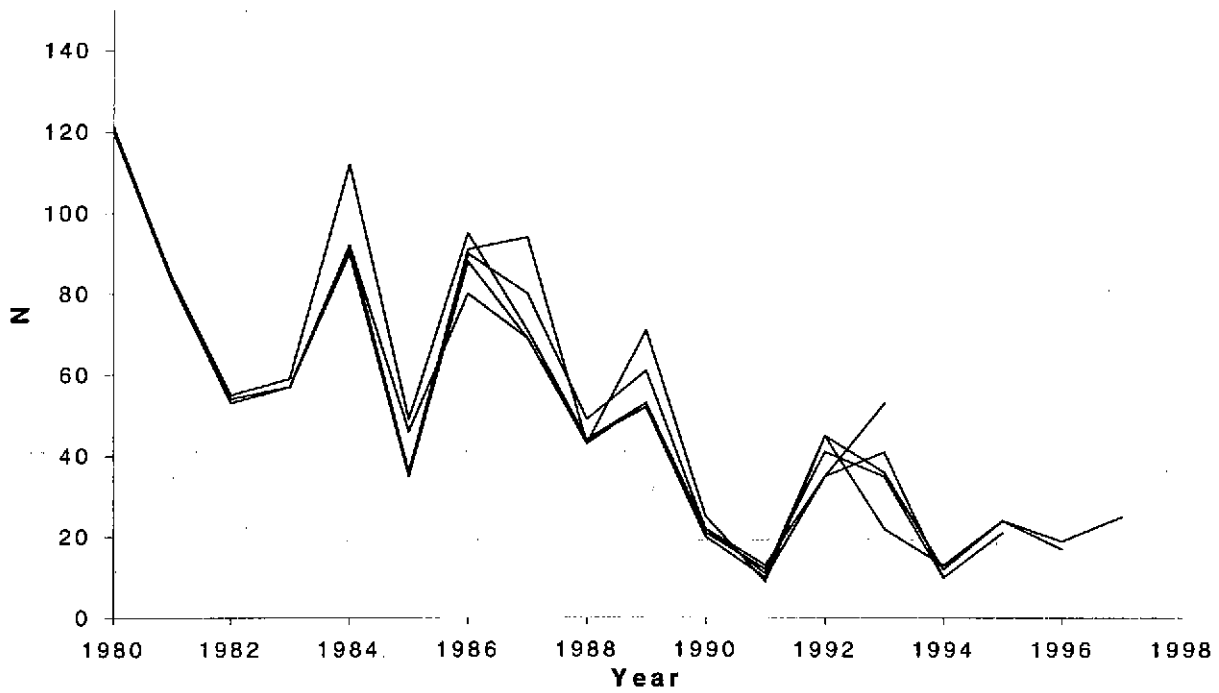
A development explored at the Working Group was the use of a geometric mean of the catchability of a specified range of older ages, rather than the estimate from a specific age. This should introduce greater robustness to the time series of catchabilities. The utility of such an approach was examined by retrospective plots of estimates of population abundance at the oldest age derived from two data sets taken from the North Sea cod and the Sim5 simulated population, ICES(1988b). The results are presented in Figures 12.4.4a,b and Figs 12.4.5a,b. They show that, for the data sets examined, there is an improvement in the consistency of the estimates derived from the XSA models fitted in consecutive years. However, as illustrated in Figures 12.4.4 a,b, the XSA estimates for the oldest age of the simulated data set may be consistent but they may also be biased when compared to the "true" population values (the bold line) used to generate the simulated data; this could result from model mis-specification.

Future modifications to the XSA model which should be investigated are the use of other weighting options for estimating the mean catchability at the oldest ages and the potential for setting the age ranges, at which the catchability models are fitted within XSA, independently for each fleet.

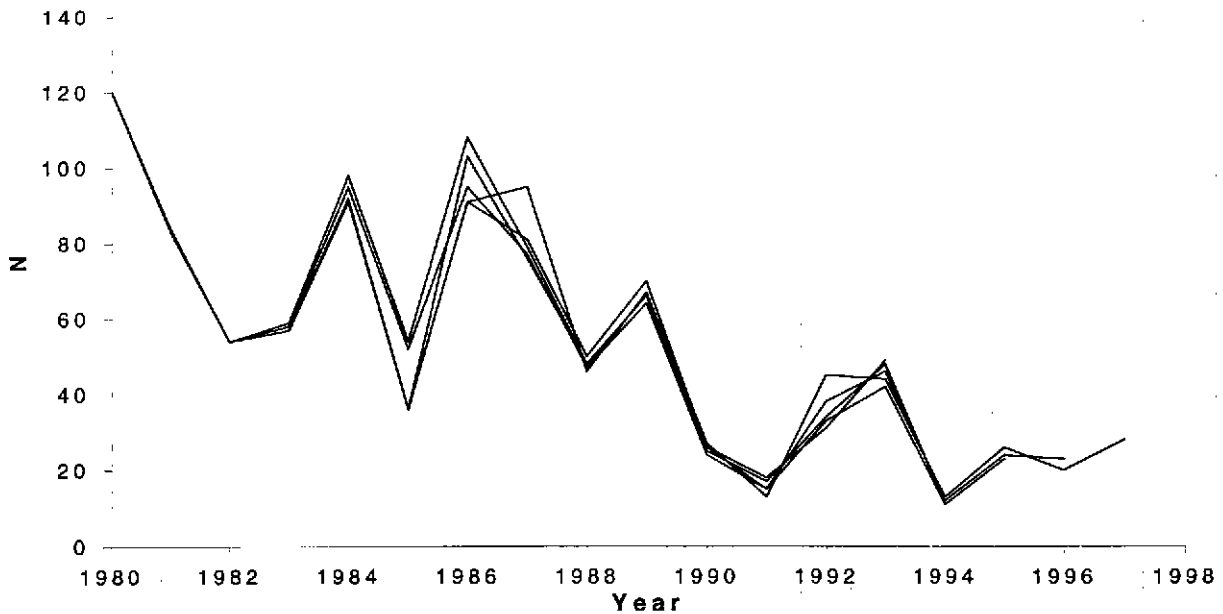
**Figure 12.4.4 a** Estimates of population abundance at age 11 derived from XSA applied the sim5 data set. Catchability at ages 9 and 10 is held constant at the value estimated for age 8. The bold line indicates the "true" population estimates used to generate the catch at age data.



**Figure 12.4.4 b** Estimates of population abundance at age 11 derived from XSA applied to the sim5 data set. Catchability is a geometric mean of ages 8 - 10. The bold line indicates the "true" population estimates used to generate the catch at age data.



**Figure 12.4.5 a** Estimates of population abundance at age 11 derived from XSA applied a North Sea cod data set. Catchability at ages 9 and 10 is held constant at the value estimated for age 8.



**Figure 12.4.5 b** Estimates of population abundance at age 11 derived from XSA applied to a North Sea cod data set. Catchability is a geometric mean of ages 8 - 10.

#### 12.4.6 Additional possibilities for estimation

Plus-group catch-at-age data are often excluded in the estimation phase of many assessments, e.g., when using XSA within ICES working groups. Once stock numbers and fishing mortality rates are determined for the all "true" ages, plus-group catch-at-age is appended by assuming some known relationship between  $F$  on the oldest "true" age and  $F$  on the plus-group. Although estimation in some of the methods above did indeed include the plus-group, an alternative approach (which may have different properties) was also discussed.

A two-class modified DeLury modelling approach (Collie and Sissenwine 1983; Conser 1991), in which the oldest "true" age is taken as "recruits" and the plus-group is taken as the remaining population (other than recruits) may lend itself nicely to this problem. Future research on methods for estimating  $F$ 's and  $N$ 's on the oldest "true" age and the plus-group may be worth exploring with modified DeLury modelling.

In some circumstances, accumulated fish in the plus group can result in unrealistically high population numbers. Modeling plus group dynamics with higher  $M$  values may alleviate this problem.

### 12.5 Bias Correction <sup>(Z)</sup>

#### 12.5.1 Introduction

Fisheries management decisions are based on point estimates and/or confidence statements about interest parameters such as spawning stock biomass or fishing mortality rate. These interest parameters may be computed for the projected time period conditional on management actions, e.g., what would be the resulting fishing mortality rate during the coming fishing season for a particular choice of quota. Fisheries assessment models, including VPA calibration models, statistical catch at age models and production models, can be complex and may involve non-linear relationships. Further, few data are typically available. This means that point estimates of interest parameters may be biased and confidence statements for the interest parameters from standard methods may not be accurate.

Statistical inference has relied on asymptotic properties, normal theory assumptions and specialised theoretical calculations to derive reliable sample estimators of population interest parameters. When sample sizes are small, the underlying distribution is unknown or the sample estimator is not an unbiased estimator of the population interest parameter (e.g., for non-linear models, least squares estimators are biased), the standard approach may not be reliable. The bootstrap (Efron 1982) was introduced as a way to automatically obtain accurate statistical properties of estimators and to make accurate confidence statements for interest parameters. The bootstrap addresses estimation bias. There may be other sources of bias arising from model mis-specification, e.g., unreported catches, temporal trends in assumed constant natural mortality or errors not independent, which are not addressed by the bootstrap.

Let  $\eta$  represent the interest parameter and let  $\hat{\eta}$  represent its estimator. Statistical properties and confidence statements for the interest parameters are derived from the bootstrap replicate estimates  $\hat{\eta}^b$ . The replicates are computed by applying the estimation to the bootstrap samples. Parametric bootstrap replications are obtained if the bootstrap samples are generated from a known parametric distribution that best fits the data. Nonparametric bootstrap replications are obtained if bootstrap samples are generated by random sampling with replacement from the empirical distribution, i.e., the observed data. Model conditioned bootstrap replications are obtained if bootstrap samples are generated by sampling the residual distribution (parametric) or the observed residuals (nonparametric) and adding these to the model predicted values.

The results of a Monte Carlo experiment from an age structured analytical fisheries assessment model using data from eastern Georges Bank haddock (Gavaris and Van Eeckhaute 1998) were reported in WP/1. The parameter estimates and the predicted values of the ln abundance index from an initial application of the assessment procedure were considered to be the 'truth'. Random Gaussian error with a mean of zero and a standard error of 0.7 was added to the 'true' ln abundance index to generate the 100 Monte Carlo trial data sets. Each of the data sets was analysed using the assessment procedure. Spawning stock biomass in the terminal year,  $SSB_t$ , was used as an example of the interest parameter for fisheries management. The experiment compared the accuracy of probability coverage between two bootstrap methods, the percentile and the bias corrected percentile. During the meeting this work was extended to compare results with those obtained from two additional methods, the shifted percentile and the bias corrected shifted percentile. The Monte Carlo results were also used to summarise and compare properties of point estimators. The findings are reported below and will be referred to as Monte Carlo experiment 1.

A similar experiment to WP/1 was reported in WP/3. The principal differences were in the underlying data generation and in some of the tests performed. The data generation is a standard stock projection with fixed recruitment. Two sorts of simulations were performed: 1) adding lognormal noise to the abundance index used in tuning, and 2) changing the model specification (either in terms of discarding practice or partial recruitment pattern) halfway through the simulated data series. Bias was estimated from bootstrapping. Bias corrections are from the bias-corrected percentile method (Efron, 1982) and adjustments using the mean and median of the bootstrap distributions, (Prager, 1994). The bootstrap estimates are also corrected as would be done in the determination of confidence levels for estimates of survivors from SPAs.

The simulations with noise added to the tuning data showed that in many cases the bias corrected point estimate was further from the truth than the point estimate. There were not sufficient replicates to infer whether or not bias correction improved the situation on average.

Comparisons was made of three types of bias correction for the cumulative frequency distribution; percentile bias correction, shift (median) and shift (average). In the one Monte Carlo trial data set, using survivor estimates from Scotian Shelf cod, the median shift and percentile method behaved similarly while the correction based on the average was considerably larger. Again more work is required.

Simulations wherein model mis-specifications were modelled as changes in discarding and partial recruitment displayed retrospective patterns. In general for both cases bias correction moved the estimate away from the true value. It is concluded that the residuals should be analysed and/or a retrospective analysis be performed before considering bias correction.

A bias corrected median shift percentile method and bias corrected percentile distributions from a Scotian Shelf cod assessment were compared. Although insufficient to generalise, the bias corrected median shift correction appears to work fairly well.

The work in WP/3 was extended during the meeting to include more replicates. As opposed to 10 replicates with bootstrap distributions defined with 1000 samples in the WP, 100 replicates with bootstrap estimates from 500 samples were performed. The results are summarised below and will be referred to as Monte Carlo experiment 2.

### 12.5.2 Point Estimation

Point estimates of fisheries management interest parameters are often biased. The bootstrap estimates of variance and bias for an interest parameter are defined as:

$$\bar{\eta} = \sum_b \hat{\eta}^b / B$$

$$\text{var}_B(\hat{\eta}) = \sum_b (\hat{\eta}^b - \bar{\eta})^2 / B - 1$$

$$\text{bias}_B(\hat{\eta}) = \bar{\eta} - \hat{\eta}$$

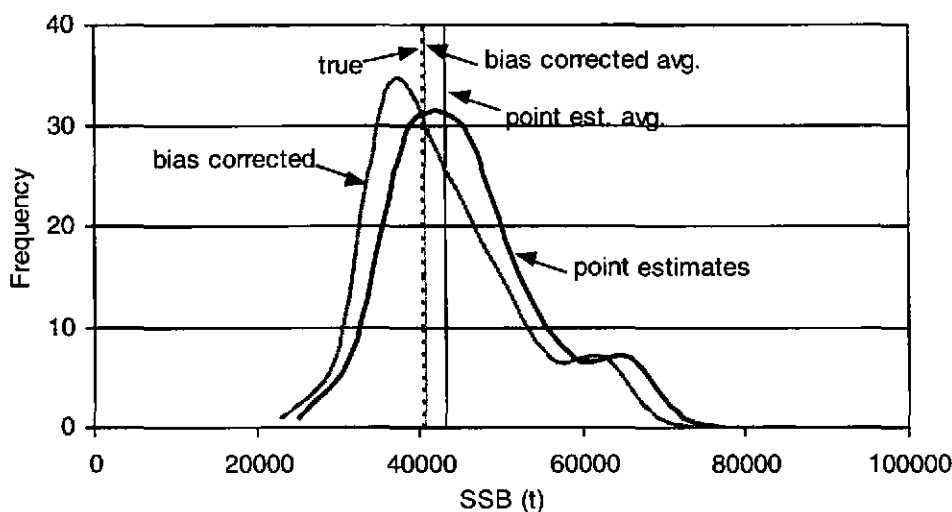
The superscript *b* indexes the B bootstrap replicates. Efron and Tibshirani (1993) emphasise that there is a wrong tendency to think of the average of the bootstrap replicates as a bias corrected estimate. If the point estimator is biased, the average of the bootstrap replicates would adjust in the wrong direction. An obvious bias corrected estimator is:

$$\hat{\eta}_{BC} = \hat{\eta} - \text{bias}_B(\hat{\eta})$$

Bias correction may reduce bias but can increase the variance of the estimator. Therefore, bias correction of point estimates is not always prudent. Efron and Tibshirani (1993) suggest that the bias may be safely ignored if it is less than 25% of the standard error.

For each trial data set in Monte Carlo experiment 1, the bias of the interest parameter was estimated from the 600 bootstrap replicates. The estimates of bias ranged between about 1,100 t and 5,100 t with an average of about 2500. The "true" SSB was about 40,000 t. The bias corrected estimates were compared to the point estimates. Figure 12.5.1 shows the frequency distribution of the two estimates from the 100 Monte Carlo trial data sets. The average for each of the estimators is compared to the true value. The expected value of the bias corrected estimator is closer to the true value. Applying the bias correction only reduced the magnitude of the deviation from the "truth" roughly half the time (48 of the 100 trials). However, the root mean squared deviation from the "truth" was about 10% smaller for the bias corrected

estimator. Comparing the estimate of bias to the respective estimate of standard error for each of the 100 trial data sets in this example, the bias ranged from about 20% to 40% of the standard error.



**Figure 12.5.1** Comparison of bias corrected estimates with the simple biased estimator of spawning stock biomass from 100 Monte Carlo trial data sets.

The results from Monte Carlo experiment 2 for point estimation are shown in Table 12.5.1. Although for ages 4 to 7 the bias correction was in the direction of the truth, it tended to over-correct with the mean overcorrecting less than the median. Age 2 is not estimated as a free parameter but is specified by the model and in this instance the bias correction was in the wrong direction. Age 3 survivors are very poorly estimated, based on one observation and the bias correction took the point estimate away from the truth. Given the wide range of the 100 realisations compared to the amount of bias shown in Figure 12.5.2, a larger sample size should be used. It was noted that the bootstrap coefficient of variation of survivors increased with age. This is contrary to the expected pattern and probably reflects model mis-specification. Therefore, these results should be regarded as preliminary.

**Table 12.5.1** Average results of 100 RV CV = 0.3 simulations with 500 bootstraps for bias estimation. The estimates are of survivors at age. Resampling of residuals was on an age by age basis. Bias M and Corr M are the bias and corrected point estimates using the median while Bias A and Corr are based on the average. SE(%) is the average standard error of the bootstrap replicates.

Age	Point	Median	Bias M	Corr M	Bias A	Corr A	SE(%)	True
2	728	775	48	680	34	693	10	795
3	596	558	-38	635	-74	670	31	595
4	371	347	-24	395	-22	393	66	361
5	176	163	-14	190	-13	189	63	183
6	82	76	-7	89	-6	88	65	82
7	40	39	-1	40	3	37	81	37

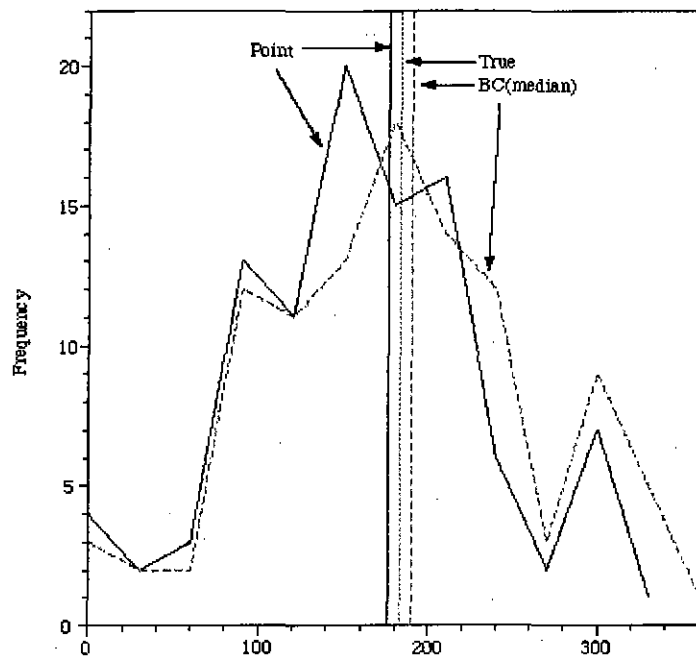


Figure 12.5.2 Point and bias corrected distributions and their means from 100 Monte Carlo trial data sets.

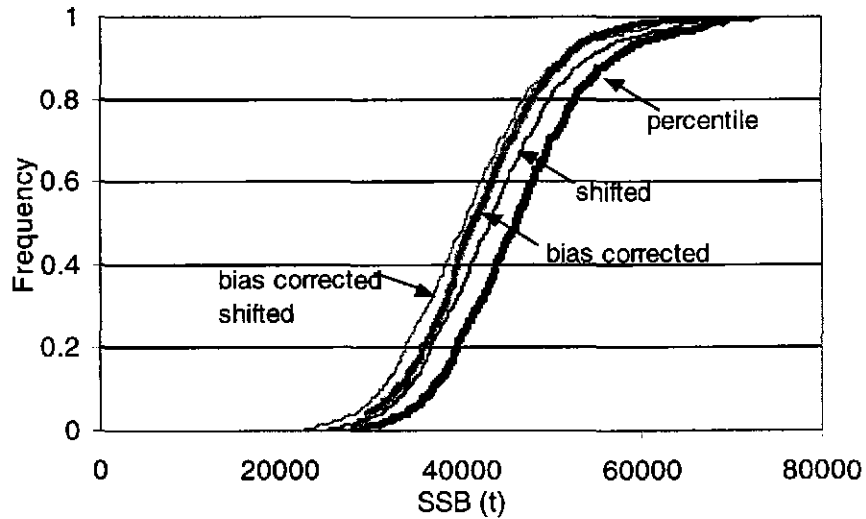
### 12.5.3 Confidence Statements

Increasingly, fisheries management decisions are taking uncertainty and risk about the interest parameters into consideration. For example, a catch quota is selected on the basis of an acceptable risk that the resulting biomass will not decrease below an established threshold level. Risk analyses are founded on the cumulative frequency distributions of interest parameters. There are several methods to construct cumulative frequency distributions from bootstrap replicates. The quality of confidence statements can be judged on the basis of correctness and accuracy (Efron 1992, Efron and Tibshirani 1993). A confidence statement is said to be correct if the candidate confidence point matches an ideal or exact confidence point,  $\hat{\eta}[\alpha] = \hat{\eta}_{exact}[\alpha]$ . Exact confidence points can only be computed in certain parametric situations. Therefore, we cannot assess correctness in the fisheries assessment problem. A confidence statement is said to be accurate if the confidence point achieves the desired probability coverage,  $\Pr(\eta \leq \hat{\eta}[\alpha]) \approx \alpha$ . We compared the probability coverage of four methods applied to a Monte Carlo experiment of 100 trial data sets. The four methods were:

- percentile- simple empirical cumulative frequency distribution constructed from the bootstrap replicates (Efron 1982)
- bias corrected percentile- adjustment of the percentile cumulative frequency distribution to account for discrepancy between its median and the point estimate, the adjustment is not constant (Efron 1982)
- shifted percentile- constant shift of the percentile cumulative frequency distribution so that its mean equals the point estimate (Noreen 1989)
- bias corrected shifted percentile- constant shift of the percentile cumulative frequency distribution so that its mean equals the bias corrected point estimate

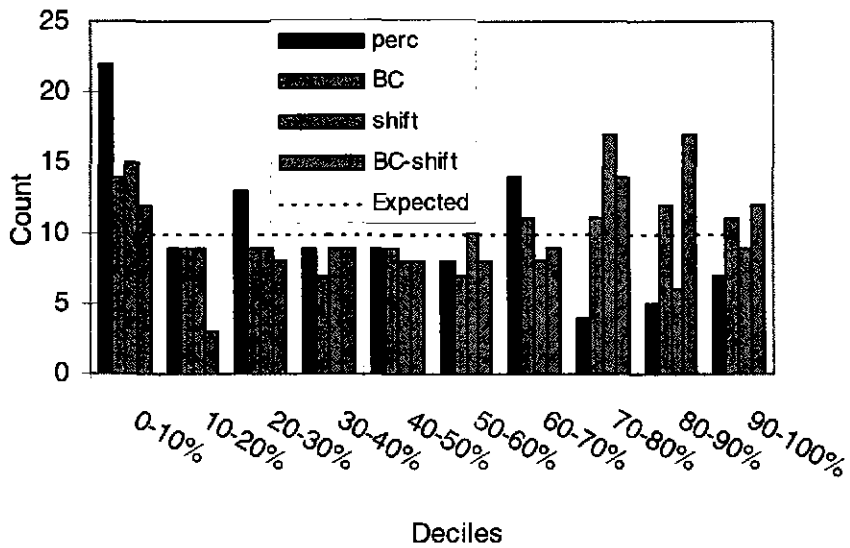
The relationship between the cumulative frequency distributions from these four methods for one of the Monte Carlo trial data sets is displayed in Figure 12.5.3 as an example. Though the distances between cumulative frequency distributions will vary from data set to data set, the relative ordering should generally be the same.



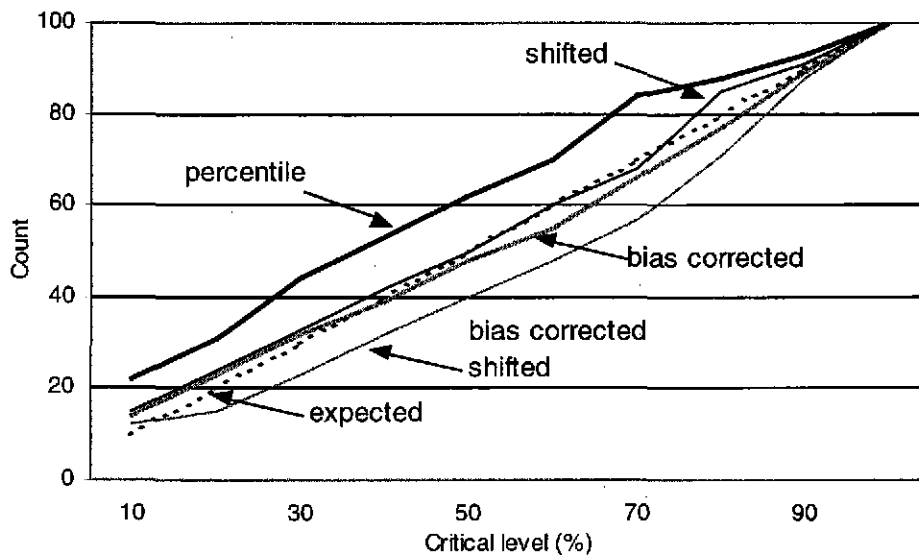


**Figure 12.5.3** Comparison of cumulative frequency distributions from 4 bootstrap methods for one of the Monte Carlo trial data sets.

We would expect the cumulative frequency distributions from 10 of the 100 Monte Carlo trial data sets to contain the 'true' *SSB* between the 20<sup>th</sup> and 30<sup>th</sup> percentile confidence limits for example, and similarly for each of the ten deciles from 0% to 100%. The results from the bias corrected percentile method more closely approximate the expected counts compared to the percentile method (Figure 12.5.4a). These same results can be displayed on a cumulative basis (Figure 12.5.4b). The bias corrected percentile method had the most accurate probability coverage while the percentile and bias corrected shifted percentile methods perform poorly.



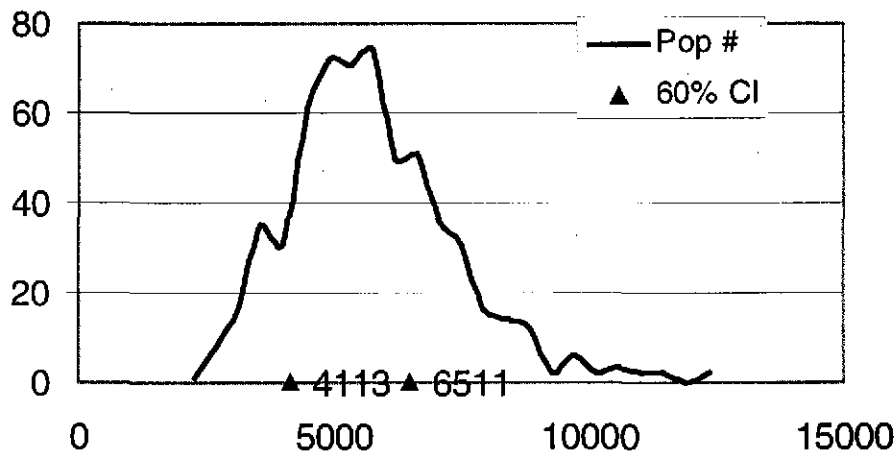
**Figure 12.5.4a** Observed counts for four bootstrap methods compared to the expected count for each of the 10 deciles between 0% and 100%.



**Figure 12.5.4b** Observed cumulative counts for four bootstrap methods compared to the expected cumulative count for critical levels from 10% to 100%.

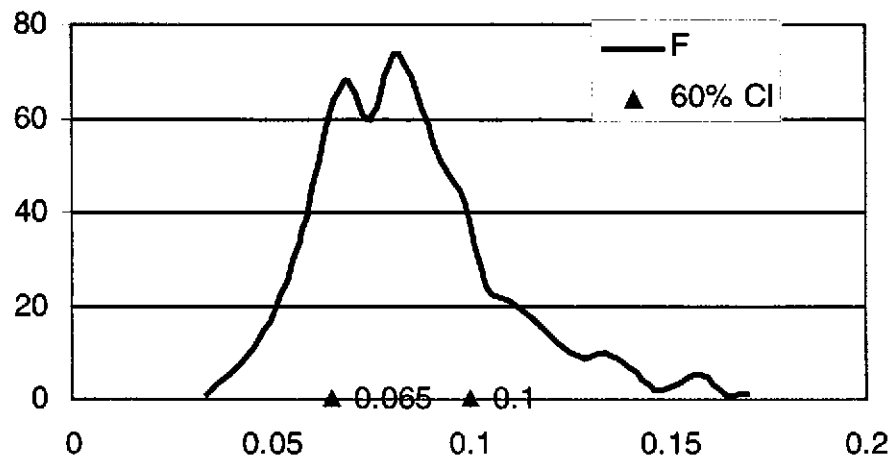
The percentile and bias corrected percentile methods also have some additional desirable properties that are not shared by other two methods. They are *transformation-respecting*. That is, for any monotone parameter transformation, the confidence limits of the transformed quantity is obtained by simply applying the transformation to the original confidence limits. The percentile methods are also *range-preserving*. That is, if the estimation method invokes a range restriction on the interest parameter, percentile confidence limits will obey the same range restrictions.

The *transformation respecting* property is important in fisheries management where there is often interest in population abundance and in the corresponding fishing mortality rate. We would like to have confidence statements about population abundance to be consistent with those about fishing mortality rate. Figure 12.5.5a displays the frequency distribution of population abundance of age 6 at the beginning of 1998 from 600 bootstraps of the first Monte Carlo trial data set in the first experiment. The confidence limits for a 60% confidence interval are indicated.



**Figure 12.5.5a** Frequency distribution of population abundance indicating 60% confidence interval.

The corresponding frequency distribution for fishing mortality rate on age 5 during 1997 is shown in Figure 12.5.5b. The catch at age 5 during 1997 was 483 and the assumed natural mortality was 0.2. It is clear, by applying the catch equation, that the 60% confidence limits for fishing mortality correspond to those for the population abundance.



**Figure 12.5.5b** Corresponding frequency distribution of population abundance indicating 60% confidence interval.

The *range-preserving* property is also important in fisheries management to ensure that we do not obtain negative confidence limits for intrinsically positive quantities like spawning stock biomass. Such negative confidence limits can undermine client credibility.

#### 12.5.4 Conclusions

It is important to emphasise that the methods examined address estimation bias. Model mis-specification may also introduce bias and result in more severe deviations. Model diagnostics should always be examined to detect important violations of assumptions and pathologic cases. The consequences of applying techniques for correcting estimation bias to pathologic assessments may not be predictable.

For any given fisheries assessment, the decision to correct (or not correct) a point estimate of interest may not be entirely straightforward. While the bias correction appears to be beneficial on average (Figure 12.5.1), it may in fact overcorrect or undercorrect any given estimate – perhaps giving a “corrected” estimate with even greater deviation from the “true”. A useful guideline for bias-correcting point estimates may be:

- if the bootstrap distribution will be corrected in order to determine confidence intervals or risk profiles (e.g. via Percentile BC or the Shift Method), it will generally be more consistent to correct the point estimates as well;
- if not, then use the 25% of standard error rule of thumb, i.e., ignore the bias when it is less than 25% of the standard error for the parameter of interest, otherwise, correct point estimate

The results of the Monte Carlo experiments are more informative regarding methods for confidence statements. The percentile method had poor probability coverage as did the bias corrected shift percentile. The performance of the bias corrected percentile method with respect to accuracy of probability coverage appeared to be superior though the shift percentile was almost as good. Recognizing the *transformation-respecting* and *range-preserving* properties of the bias corrected percentile method, this approach seems preferable to the others tested here.

The issue of how many bootstrap replicates are needed for reliable results in the fisheries assessment problem was not thoroughly investigated. Some results from Monte Carlo experiment 1 suggested that about 600 replicates were needed in that case to get stable results. It would appear that the practice of using about 200 replicates may not be sufficient.

## 13 RESEARCH RECOMMENDATIONS

- The bias corrected and accelerated percentile method has theoretical properties which are superior to the methods examined. Computation of the acceleration constant has not been investigated for model-conditioned nonparametric bootstrap therefore it has not been implemented for the fisheries assessment problem.
- Investigate the implications of bias correction for medium term projections which employ joint bootstrap/Monte Carlo simulation procedures.

### 13.1 A method to evaluate temporal changes in technical efficiency of fishing fleets

A working document was presented to describe a method to evaluate temporal changes in technical efficiency of fishing fleets (WP 7). The annual assessments of Northern Europe fish stocks are often carried out with the underlying assumption that catchability, that is the coefficient of proportionality between effective and nominal fishing effort, remains constant over time. This assumption allows considering CPUE (Catch per Unit of Effort) as a good indicator of stock abundance (Mendelsohn and Cury 1989). However, evidence is accumulating that the catchability of commercial fleets is subjected to non-stochastic variations over time (Crecco and Overholtz 1990; Swain *et al.* 1994). Hence, variability in CPUE should be attributed to both biomass and catchability fluctuations. A major issue is to be able to distinguish catchability variability and stock fluctuations in catch and effort data. A method is hereby suggested to derive a stock-independent index of time trend in catchability from catch and effort data. This index, referred to as index of technical efficiency is calculated relatively to a reference sub-fleet, which is selected with regards to its stability over time.

It was noted that a selection process which is based on minimum variance in CPUE may select a subfleet which does not have CPUE proportional to stock size.

#### 13.1.1 Glossary

alpha, beta

$v$	fishing vessel,
$r$	reference fishing vessel,
$V$	number of fishing vessels within one fleet,
$R$	number of reference fishing vessels within one fleet,
$t$	month,
$T$	duration of the fishing period (months),
$E_v(t)$	fishing effort for vessel $v$ , in month $t$ (days),
$C_v(t)$	catch for vessel $v$ , in month $t$ (tonnes),
$U_v(t)$	CPUE (Catch Per Unit Effort) for vessel $v$ , in month $t$ (tonnes/day),
$U_V(t)$	CPUE (Catch Per Unit Effort) for all the $V$ vessels belonging to one fleet, in month $t$ (tonnes/day),
$N(t)$	stock biomass in month $t$ (tonnes),
$f[N(t)]$	any function of $N(t)$ ,
$q_v(t)$	catchability coefficient for vessel $v$ , in month $t$ ( $\text{day}^{-1}$ ),
$q_V(t)$	catchability coefficient for all the $V$ vessels belonging to one fleet, in month $t$ ( $\text{day}^{-1}$ ),
$\text{Var}\{X(t)\}$	variance of a variable $X(t)$ , calculated over the $T$ -month fishing period,
$\text{Cov}\{X(t), Y(t)\}$	covariance between two variables $X(t)$ and $Y(t)$ , calculated over the $T$ -month fishing period,
$I(t), J(t)$	indices of technical efficiency for one fleet, in month $t$ .

#### 13.1.1.1 CPUE model

A model is required to describe variations of CPUE in terms of catchability and stock abundance. A general representation of CPUE is

$$U_v(t) = \frac{C_v(t)}{E_v(t)} = q_v(t) f[N(t)] \quad (1a)$$

$$U_v(t) = \frac{\sum_{v=1}^V C_v(t)}{\sum_{v=1}^V E_v(t)} = q_v(t) f[N(t)] \quad (1b)$$

The main assumption underlying Equations (1a) and (1b) is that CPUE can be split into one stock-independent ( $q_v(t)$  or  $q_v(t)$ ) and one stock-dependent ( $f[N(t)]$ ) components.

### 13.1.1.1.1 Index of technical efficiency

The technical efficiency of any fleet is defined as the ratio between the catchability of all  $V$  vessels and the catchability of a sub-fleet of  $R$  reference vessels. For several reasons, we calculate and analyse the natural logarithm of this ratio. First, ratios are rarely linear or gaussian in distribution. Calculating the natural logarithm of this ratio, that is the difference between the log-catchabilities, provides a series, which is expected to have better statistical properties with regards to normality. Second, taking the logarithm of the original series decreases the weight of marginal values in the analysis. An index of technical efficiency  $I(t)$  of a fleet relatively to the reference sub-fleet may then be expressed as

$$I(t) = \ln\left(\frac{q_v(t)}{q_r(t)}\right)$$

$I(t)$  cannot be calculated directly. However, it may be possible to derive another index of technical efficiency,  $J(t)$ , whose temporal variations are consistent with fluctuations in  $I(t)$ .

We are first selecting, amongst the  $V$  vessels, a sub-fleet of  $R$  reference vessels, whose catchability coefficient has remained constant over the past  $T$  months. Consider the situation where there is only one reference vessel  $r$  in the sub-fleet ( $R = 1$ ). If such a vessel exists, then it must satisfy

$$\text{Var}\{\ln(q_r(t))\} = \text{Var}\{\ln(q_r)\} = 0$$

In practice, due to the very dynamic character of fishery systems, such a vessel is unlikely to exist. However, it is possible to look for one vessel  $r$ , which complies with equation (2)

$$\text{Var}\{\ln(q_r(t))\} = \text{Min}\{\text{Var}\{\ln(q_v(t))\} \mid 1 \leq v \leq V\} \quad (2)$$

We then calculate

$$\text{Var}\{\ln(U_v(t))\} = \text{Var}\{\ln(q_v(t))\} + \text{Var}\{\ln(f[N(t)])\} - 2 \text{Cov}\{\ln(q_v(t)), \ln(f[N(t)])\}$$

Assuming that  $q_v(t)$  is stock-independent, we then have

$$\text{Var}\{\ln(U_v(t))\} \approx \text{Var}\{\ln(q_v(t))\} + \text{Var}\{\ln(f[N(t)])\}$$

and

$$\text{Min}\{\text{Var}\{\ln(q_v(t))\} \mid 1 \leq v \leq V\} = \text{Min}\{\text{Var}\{\ln(U_v(t))\} \mid 1 \leq v \leq V\} - \text{Var}\{\ln(f[N(t)])\}$$

The vessel  $v$ , whose catchability minimises  $\text{Var}\{\ln(U_v(t))\}$ , also minimises  $\text{Var}\{\ln(q_v(t))\}$  and hence,  $r$  is from now on defined as the vessel whose catchability minimises  $\text{Var}\{\ln(U_r(t))\}$ .

In practice, the above methodology may not be applicable as such to real data, for two reasons. First, the reference vessel should be chosen amongst a pool of vessels that have been fishing regularly within the  $T$ -month period. Second, the temporal dynamics in the catchabilities of 2 vessels could be very different, while their characteristic variance  $\text{Var}\{\ln(q_v(t))\}$  is similar. While both vessels could potentially be chosen as the reference vessel, the temporal dynamics of  $I(t)$ , calculated for each of them, could be completely different. Hence, calculating  $I(t)$  relatively to only one reference vessel may bring up undesirable features, specific to this vessel and make the interpretation of  $I(t)$  less tractable. As a result of these two points, a selection of vessels, which have accumulated the largest number of days fishing in the  $T$ -month period, is picked up in each fleet. These vessels are then sorted in ascending order, with respect to  $\text{Var}\{\ln(U_v(t))\}$ . For each fleet and each species, a reference sub-fleet will then be made up of the vessels with the lowest variance in Log-CPUE ( $\text{Var}\{\ln(U_v(t))\}$ ). The number of vessels to be included in each reference sub-fleet will be determined by comparing the variance of Log-CPUE across vessels, for each fleet and each species. It is assumed from now on that

$$\text{Var}\{\ln(q_R(t))\} \ll \text{Var}\{\ln(q_v(t))\}$$

The new index of technical efficiency  $J(t)$  is then calculated for each fleet by

$$J(t) = \ln \left( \frac{U_v(t)}{U_R(t)} \right)$$

## 14 CONCLUSIONS AND RECOMMENDATIONS

### 14.1 Precautionary approach

The WG recommends that it is important to consider precautionary management rules for fisheries which are controlled by technical measures only in addition to TAC-controlled fisheries.

The WG recommends that the variability of the selectivity of commercial fleets, uncertainty in growth, maturity and natural mortality should be taken into account when creating a risk averse management rule.

The WG recommends that there is a need to establish a theoretical framework to evaluate combined technical measures (e.g., area closure, mesh size, landing size) and that this needs to be integrated with economic analyses.

The WG recommends that Biological Reference Points (BRPs) should be recomputed when there are persistent and substantial changes in the biological or fishery parameters. How often the quantities need to be recomputed when the underlying causes are changes in weight, fecundity, selectivity and survivorship probably depends on the annual signal:noise ratio in the data.

### 14.2 Inclusion of multispecies interactions and bioeconomics in studies of reference points

The WG recommends that to the extent that growth, natural mortality and recruitment are strongly dependent on multispecies interactions, studies of management strategies should involve modelling of multispecies interactions and preferably bioeconomic considerations.

The WG recommends increased cooperation among the various multispecies modelling groups. This could in part be accomplished by exchanging a common model or data description analogous to the exchange of fishery data to compare VPA programs.

The WG recommends that the ICES demersal WG (WGNSSK) takes up the following issues in the assessment of North Sea plaice and sole:

- Further evaluate the implication of multi-species assessments of North Sea flatfish fisheries
- Evaluate the implications of including 0-group estimates in the VPA assessments rather than using RCT3 for recruitment estimation
- Further evaluate the implications of not including discards in the traditional VPA assessments
- Further explore model-mis-specification using different assessment models e.g., selection pattern
- Review the COMFIE report with regard to the comprehensive fishery evaluation of North Sea plaice and sole
- Explore the possibilities of incorporating economic analysis in the assessment process.

The WG suggests that ACFM consider requesting ICES WG to attempt estimation of prices and costs in the respective fisheries, utilising whatever expertise is required.

The WG recommends that the information on So. Gulf of St. Lawrence cod be combined into a complete review of current knowledge for the next meeting of COMFIE.

### **14.3 Recommendations about future work and operation of the WG**

Multispecies considerations are needed for the development of 'comprehensive' analysis. Coordination is needed with Multispecies Assessment Working Group, or perhaps a successor, to avoid duplication of effort.

The future work of the Group should include review and synthesis of current research leading towards a comprehensive description of designated fisheries (including modelling of biological, management, technological, economic and marketing interactions and effects) and usable for the provision of management advice for strategies in the medium and long term.

The Group should also advise ACFM/Resources Committee on research activities presently required to achieve this goal, and benefits immediately attainable therefrom.

The Group should also address methodological issues in fisheries assessment and management in the context of comprehensive fishery evaluation.

## **15 BACKGROUND MATERIAL AND WORKING DOCUMENTS PRESENTED TO THE WORKING GROUP**

WP1: Gavaris, S. Comparison of confidence statements for a fisheries assessment problem from two bootstrap methods.

WP2: Withdrawn

WP3: Mohn, R. Simple tests of bias correction in SPAs using simulated assessment data or to BC or not to BC.

WP4: Cadigan, N. G. and Myers, R. A. A comparison of lognormal and gamma maximum likelihood estimators of a sequential population analysis.

WP5: Sinclair, A. F., Hanson, J. M., Swain, D. P., Currie, L. Size selective mortality of cod in the Southern Gulf of St. Lawrence.

WP6: Sinclair, A. F., Swain, D. P., Hanson, J. M. Disentangling the effects of size selective mortality, temperature and density on the growth of cod in the Southern Gulf of St. Lawrence.

WP7: Marchal, P., Lassen, H., Hovgård, H. and Hartman, M. Identifying and analysing temporal variations in fisheries technical efficiency.

WP8: Rijnsdorp, A. D., Dol, W., Hooyer, M. and Pastoors, M. A. Effects of fishing power and competitive interactions among vessels on the effort allocation on the trip level of the Dutch beam trawl fleet.

WP9: Rijnsdorp, A. D., van Maurik Broekman, P. L. and Visser, E. G. Competitive interactions among beam trawl vessels exploiting local patches of flatfish in the North Sea.

WP10: Gislason, H. Single and multispecies reference points for Baltic fish stocks.

- WP11: Taylor, L., Stefánsson, G. Comprehensive assessment of haddock (*Melanogrammus aeglefinus*) in Icelandic waters.
- WP12: Bell, E. Harvest control rules in the absence of age based assessments.
- WP13: Pastoors, M. What do we call comprehensive anyway?
- WP14: Legault, C. M. Does the age range of indices affect F ratio estimation?
- WP15: Gavaris, S. On estimating F for the oldest age group.

## 16 REFERENCES

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## 17 APPENDIX I: NOTATION

### 17.1 Biological reference points

Biological reference points have been discussed in several reports of the "Methods Working Group" (Anon 1983, 1984, 1993) and in the previous COMFIE reports (Anon 1996, 1997). Caddy and Mahon (1995) review the literature on reference points and provide commentary on various problems related to their implementation. Reference points are most commonly stated in terms of fishing mortality rates or biomass. In this section, we provide a brief overview of the biological reference points which are discussed in later sections of this report. These reference points are commonly derived from analyses of yield per recruit ( $Y/R$ ) and spawning stock biomass per recruit ( $SSB/R$ ), and from age-structured production models.

$F_{0.1}$ : fishing mortality rate at which the slope of the yield per recruit curve as a function of fishing mortality is 10% of its value near the origin.

$F_{max}$ : fishing mortality rate which corresponds to the maximum yield per recruit as a function of fishing mortality.

$F_{low}$ : fishing mortality rate on an equilibrium population with a  $SSB/R$  equal to the inverse of the 10th percentile of the observed  $R/SSB$ .

$F_{med}$ : fishing mortality rate on an equilibrium population with a  $SSB/R$  equal to the inverse of the median observed  $R/SSB$ .

$F_{high}$ : fishing mortality rate on an equilibrium population with a  $SSB/R$  equal to the inverse of the 90th percentile of the observed  $R/SSB$ .

$F_{x\%}$ : fishing mortality rate on an equilibrium population with a  $SSB/R$  of  $x\%$  of the  $SSB/R$  for the corresponding unfished population.

$B_{MSY}$ : biomass corresponding to maximum sustainable yield as estimated from a production model

$F_{MSY}$ : fishing mortality rate which corresponds to the maximum sustainable yield as estimated by a production model.

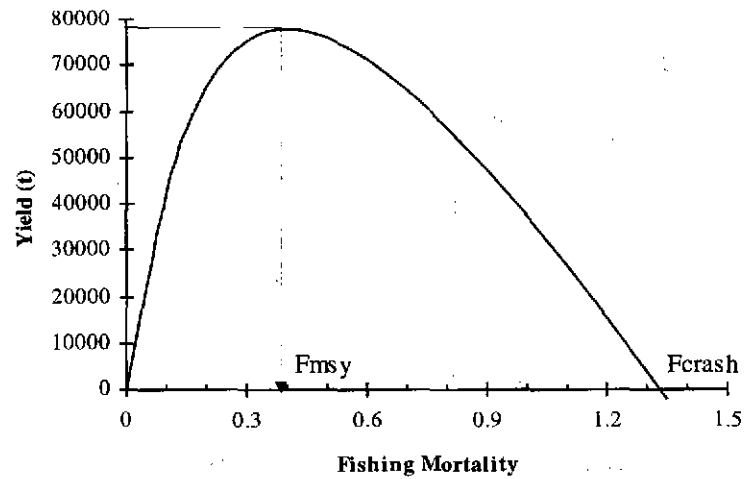
$F_{crash}$ : fishing mortality which corresponds to the upper intersection of the yield and fishing mortality relationship with the fishing mortality axis as estimated by a production model.

$F_{loss}$ , the replacement line corresponding to the Lowest Observed Spawning Stock (LOSS).

$B_{50\% R}$ , the level of spawning stock at which average recruitment is one half of the maximum of the underlying stock-recruitment relationship.

$B_{90\% R, 90\% Surv}$ : level of spawning stock corresponding to the intersection of the 90th percentile of observed survival rate ( $R/S$ ) and the 90th percentile of the recruitment observations.

Estimates of these quantities may be conditional on the stock-recruitment relationship assumed.



**Figure 2.1** Equilibrium yield as a function of fishing mortality determined from an age-structured production model.

In addition to these analytical reference points, the Minimum Biologically Acceptable Level (MBAL) refers to a critical value of spawning stock biomass. Issues related to the calculation and interpretation of MBAL have been discussed elsewhere (Anon 1991, Anon 1993).



