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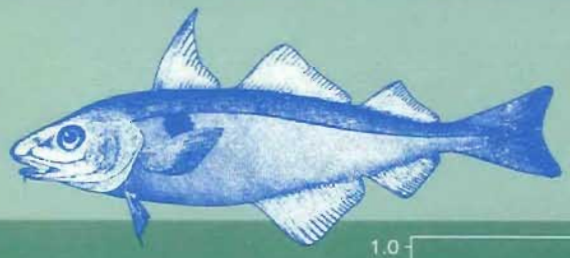
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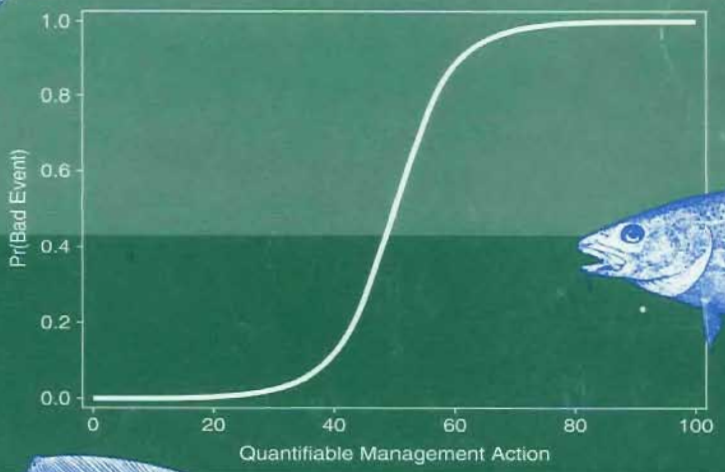
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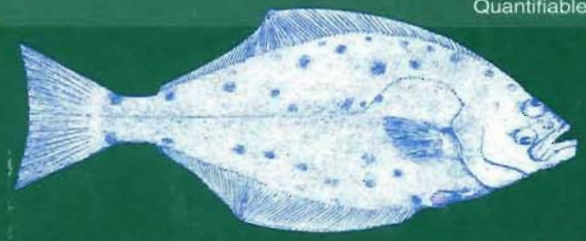
Risk Evaluation and Biological Reference Points for Fisheries Management

Edited by
Stephen J. Smith, Joseph J. Hunt, and D. Rivard









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Risk Evaluation and Biological Reference Points for Fisheries Management

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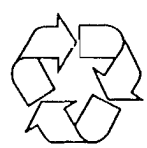
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Abstract/Résumé

Smith, S. J., J. J. Hunt and D. Rivard. 1993. Risk evaluation and biological reference points for fisheries management. Can. Spec. Publ. Fish. Aquat. Sci. 120.

The Canadian Atlantic Fisheries Scientific Advisory Committee (CAFSAC) of the Canadian Department of Fisheries and Oceans sponsored a workshop on Risk Evaluation and Biological Reference Points for Fisheries Management, November 19–22, 1991, in Halifax, Nova Scotia. This meeting was attended by 60 fisheries scientists and economists from Australia, Canada, Finland, France, Iceland, New Zealand, Norway, South Africa, the United Kingdom and the United States. This volume contains 31 papers presented at the workshop in three theme sessions. In the first session the use, track record, strengths and weaknesses of commonly used biological reference points (e.g., F_{med}, $F_{0.1}$, SSB/R) were reviewed. Papers in the second session were focussed on methods for identifying and quantifying the uncertainties associated with using various reference points. Alternative management strategies based on new reference points or suggested strategies for dealing with novel situations (e.g. developing new fisheries resources, management of stock complexes) were the subject of the third session. This volume also includes the reports of four working groups formed to answer specific questions on the application of risk evaluation and biological reference points. These answers were presented by each group in plenary session at the end of the meeting. One common thread which ran through all of the working groups presentations was the emphasis on the scientists' responsibility to evaluate the different management strategies and the managers' responsibility to establish the objectives to be met by these strategies.

Du 19 au 22 novembre 1991, le Comité Scientifique Consultatif des Pêches Canadiennes dans l'Atlantique (CSCPCA), ministère Canadien des Pêches et Océans, a tenu à Halifax, Nouvelle Écosse, un atelier intitulé «Évaluation des risques et points de référence biologiques en gestion des pêches». Soixante chercheurs et économistes du domaine des pêches, en provenance d'Australie, du Canada, de la Finlande, de la France, de L'Islande, de la Nouvelle Zélande, de la Norvège, de l'Afrique du Sud, du Royaume-Uni et des États-Unis, y ont participé. Ce document renferme 31 articles présentés au cours de trois sessions thématiques. La première session a servi à faire le point sur l'utilisation, la performance, les forces et les faiblesses des points de référence biologiques d'usage général (e.g., F_{med}, $F_{0.1}$, BSR/R). Les articles présentés au cours de la deuxième session ont touché au méthodes d'identification et d'estimation des incertitudes associées à l'utilisation de divers points de référence. Les stratégies de gestions fondées sur de nouveaux points de référence ainsi que des suggestions pour faire face à de nouvelles situations (e.g., développement de nouvelles ressources halieutiques, gestion de complexes de stocks) ont fait l'objet de la troisième session. Ce document contient également le rapport des quatre groupes formés afin d'adresser des questions spécifiques sur l'application de l'évaluation des risques et de points de référence biologiques. Les réponses apportées ont été présentées par chaque groupe en session plénière à la fin de l'atelier. Tous les groupes ont soulevé que, bien qu'il en est de la responsabilité des chercheurs d'évaluer les différentes stratégies de gestion, l'établissement des objectifs à rencontrer au moyen de ces stratégies doit être fait par les gestionnaires.

Introduction

This volume contains the proceedings from the Workshop on "Risk Evaluation and Biological Reference Points for Fisheries Management" which was held on Nov. 19–22, 1991 in Halifax, Nova Scotia. The Canadian Atlantic Fisheries Scientific Advisory Committee (CAFSAC) of the Government of Canada sponsored the meeting which was attended by 60 fisheries scientists and economists from Australia, Canada, Finland, France, Iceland, New Zealand, Norway, South Africa, the United Kingdom and the United States (Appendix A).

One of the driving motivations for holding this workshop was to canvass fisheries scientists throughout the world on ways and means of dealing with all of the uncertainties inherent in managing a fishery. The fact that there are uncertainties associated with decision-making in fisheries management does not make this process any different from most of the other decision-making processes in society. Nor is this the first time such uncertainties have been looked at in a fisheries science forum (see for e.g., Shepherd 1991). However, over the last few years there has been a great deal of interest in quantifying and incorporating the basic sources of uncertainties into fisheries management advice from CAFSAC. This interest is based on the assumption that we would present more useful and realistic advice if the uncertainties were recognized. The processes by which the uncertainties would be characterized and incorporated into fisheries advice were loosely termed 'risk' evaluation for the purposes of the workshop.

A review of the many biological reference points currently in use (e.g. $F_{0.1}$, F_{med}, %SPR) was also seen to be timely. Impressive fisheries databases have accumulated over the past 15–20 years. Stock assessment methods have also evolved and advances have been made in quantifying uncertainties in key parameters. Different management approaches have been adopted around the world to suit different realities; thus, an opportunity existed to compare different experiences and investigate alternative approaches. An evaluation of the various reference points was of particular interest in eastern Canada where the groundfish industry had moved from an annual to a multi-year management plan.

The workshop was designed to focus attention on "Biological Reference Points" and "Risk Evaluation" in two ways. The first consisted of the presentation of 38 invited and contributed scientific papers in three sessions (Appendix B).

In the first session the use, track record, strengths and weaknesses of biological reference points were reviewed. Authors of papers in this session were asked to comment on the role that uncertainties may have played in the successes or failures associated with the use of particular biological reference points. One of the major sources of uncertainty common to many of the papers was not one of those usually identified as being important (variability in recruitment, aging errors, etc.). This major source of uncertainty identified was associated with the lack of hard objectives for the management of fish stocks. Many authors pointed out the lack of well-defined social, economic or political objectives often resulted in biological targets being assumed for the fishery. However, if the advice based upon the biological reference point(s) indicated severe conditions for the fishery, other objectives such as maintaining catch levels, reaching agreements amongst member nations in a multinational fishery, etc., became much more important. As a result, there was a general *crie-de-coeur* for an internationally accepted "biological" bottom line for stock depletion which would override any economic, social or political objectives. Proponents of Spawning Stock Biomass per Recruit (SSB/R) based reference points have used recruitment overfishing as this bottom line and define the overfishing threshold as being between 20–30% of SSB/R when $F = 0$ depending upon taxonomic group.

Methods for identifying and quantifying the uncertainties associated with using various reference points were the focus of the second session. The papers in this session fell into two categories. The first category comprised propagation-of-error studies in which measured or assumed uncertainties in the basic data or model assumptions were carried through to estimates of the common quantities from assessment models (e.g., projected catch, yield-per-recruit). Bootstrap or Monte-Carlo techniques were used to study the propagation-of-error due to the complexity of the assessment models being studied. We very quickly ran into a nomenclature problem with the variety of bootstrap and Monte-Carlo methods used, and in Table 1 we offer a key to the definitions of the different kinds of simulation methods used herein.

The second category of papers in this session concentrated on simulation types of approaches to characterize the effects of the uncertainties in terms of the probability or "risk" of some consequence occurring (i.e. meeting a target, exceeding a set fishing mortality, etc.). The key thing to be determined in all such studies was what end event or consequence were we interested in — specifically, what event is 'risky' in the context of fisheries management. In many of the papers the end events were defined in terms of detrimental effects while two papers evaluated properties of the estimates used for management advice. The paper by Rosenberg and Brault was unique in that it was not concerned with the standard sources of uncertainties (e.g., recruitment, M) but with the uncertainties associated with the ability of management to achieve a particular harvest rate in any one year.

The presentations in the third session were concerned with alternative management strategies based on new reference

Table 1. Definition of bootstrap methods used in the workshop.

Method			
Model Assumptions	Form	Definition	Examples
Unconditional	Nonparametric	Observed data are resampled with replacement to form new sets of data which are then used in subsequent analyses. This is the standard bootstrap of Efron (1982).	Pelletier and Gros 1991; Punt and Butterworth (this volume); Smith and Gavaris (this volume).
	Parametric[1]	Parameter estimates from observed data are used to characterize a particular statistical distribution. Random numbers are generated using this distribution to form new sets of data to be used in subsequent analyses. (see footnote 1 below.)	Punt and Butterworth (this volume); Smith and Gavaris (this volume).
Conditional[2]	Nonparametric	The observed residuals from the fit of a model to some observed data are resampled with replacement and then added back onto the predicted values to form new sets of data which are then used in subsequent analyses.	Mohn (this volume).
	Parametric[3]	The observed residuals from the fit of a model to some observed data are used to estimate the variance for the assumed distribution of the predicted data. Random numbers are generated using this distribution to form new sets of data to be used in subsequent analyses.	Gavaris (this volume); Punt and Butterworth (this volume)

[1]: This is basically a Monte-Carlo method which uses the observed data to characterize the parameters of the random number generator. Efron's (1982) parametric bootstrap uses estimates of accuracy based upon properties of an assumed parametric distribution and does not require random numbers being generated.
[2]: The conditional aspect refers to the fact that the residuals are conditional upon a specific model being fitted to the original data and the predicted values are also from this model.
[3]: This approach is similar to that in footnote 1 in that parameter estimates from a fit of a model to observed data are used for the parameters of the random number generator.

points or with suggested strategies for dealing with novel situations.

Management strategies based on life history parameters figured prominently in many of the papers. The first four papers included here from this session concentrated on single species/stock specific applications such as single cohort fisheries or developing fisheries as well as assessment model specific issues. The next three papers evaluated the application of a number of reference points to many species and stocks. The derivation and evaluation of bioeconomic objectives with and without uncertainty being accounted for, was the subject of five papers presented herein. Experimental management methods to reduce the level of uncertainty in managing fisheries was the focus of the next two papers. The first deals with an actual application while the second reviews the literature on the subject. The final paper in this session looks at institutional uncertainties associated with the structure of fisheries management organizations.

In the second part of the workshop, four working groups were formed to answer specific questions on the application of risk evaluation and biological reference points. These answers were presented by each group in plenary session at the end of the meeting and are included in this volume.

Former Assistant Deputy Minister Brian Morrissey set the stage for the working group discussions in his opening address by focusing on the purpose of fisheries management and the role of the Department in achieving this purpose. In particular, he introduced a common thread which ran through all of the working groups' presentations — the emphasis on the scientists' responsibility to evaluate the different management strategies and the managers' responsibility to establish the objectives to be met by these strategies.

The application of Risk Evaluation to fisheries management problems is still in its infancy. This probably has less do with the availability of suitable techniques than the requirement that one needs to be specific about the end event being evaluated. Indeed, many of the presentations in Session I, in which track records of the different biological references were reviewed, raised the issue that the end event, or objective of fisheries management remains ill-defined in practice.

The desire to incorporate social and economic goals (with appropriate reference points) in a rigorous and formal fashion was expressed by all of the working groups. However, the unanimous caveat put on the incorporation of these latter goals by the working groups, was that there must be a biologically defined threshold for minimum stock size. This threshold must be beyond compromise to any social, economic or political objective as it concerns the very viability of the resource.

Acknowledgements

It is inevitable that a project of this magnitude and complexity will owe its successful completion to the assistance and cooperation of a large number of people. We wish to

offer our gratitude and appreciation to the following people who provided their expertise in the following areas.

Organizational

The conceptual and organizational details for the meeting were developed by the Steering Committee, consisting of M. Chadwick, J. Hoenig, J. Hunt, J.-J. Maguire, D. Rivard, A. Sinclair and S. J. Smith. Arrangements for hotel rooms, meeting rooms, documentation, social activities and all of the other sundry items that came up in the course of the meeting were handled with great care and skill by D. Geddes and T. Dugas of the CAFSAC Secretariat. J. Hunt, D. Rivard and A. Sinclair contributed to the smooth running of the sessions as chairpersons.

The working groups were chaired by R. I. C. C. Francis, M. Hildén, E. Pikitch and R. Arnason. They and their respective rapporteurs, namely, R. Stephenson, S. Farlinger, G. Chouinard and B. Leaman are thanked for taking on their duties at short notice and for preparing their reports in such a timely fashion.

Technical

All papers submitted to the editors after the meeting were reviewed by two referees. We acknowledge all those who assisted us in reviewing these papers in Appendix C.

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We dedicate this volume to our wives and children for their understanding and patience with the extra hours that this project consumed.

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[1] TeX is a trademark of the American Mathematical Society.
[2] PostScript is a trademark of Adobe Systems Inc.

Biological Reference Points — Some Opening Comments

J. B. Morrissey*

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In preparing for this conference, I asked myself what might be a useful framework to offer at the beginning of this conference which might help position

- the purpose of our activities, and,
- the role we hope to play.

Four questions which came to mind were:

1. What are we trying to achieve in fisheries management?
2. What are the "benefits" sought by our clients?
3. For whom are we trying to achieve these benefits?
4. What is Science's legitimate role in the endeavour?

In looking at these questions, I felt that in order to answer "What are we trying to achieve?", we would have to identify the "benefits" being sought by our clients. This, of course, begged the question "Who are our clients?". In consequence, I felt that the order of the questions should be reversed. That is, we needed first of all to note who our clients were; then we needed to know what benefits they sought from our services; next, we needed to establish which of those benefits we might be able to provide. Having identified the benefits which we might be able to provide, we would be in a position to identify our legitimate role in the endeavour. In doing an analysis of this kind, organizational behavourists often use what is know as a "SOGI" analysis (Society, Organization, Group, Individual). This simply means, that we identify our clients at the level of society, the organization groups in society and individuals. We then ask ourselves, what are the benefits which each would expect a fisheries department, and its scientists to meet.

Society

Society would probably expect its department of fisheries to:

- Ensure the division of scarce resources between competing interest groups in a peaceful manner.
- Ensure that the fishery is prosecuted in an environmentally friendly manner.
- Work for the conservation of fish stocks.
- Ensure the preservation of nature for future generations.
- Ensure the sustainable development of aquatic resources for the benefit of society.
- Ensure prudent profit taking in the fishery (avoid short-term over-exploitation).
- Ensure the above actions are taken with fairness, justice and equity.

Organization

The organization in question is the Department of Fisheries and Oceans. Since a government department is an instrument of society, its role is to meet society's needs. Consequently, a government department could be said to have as its role meeting the needs of society as outlined above. In meeting its needs, society will require that the department respect the following principles of common law:

- Respect the "degree of benefit" principle. This means before taking an action which affects the public, the benefits to society needs to be assessed.
- Respect the "probability of harm" principle. This means that when assessing the degree of benefits, the state also assess the probability of harm. In other words, weigh the benefits against the costs in decisions made in relation to the fishery.
- Respect the "duty of care" principle. This means that when making its decision on the fisheries, the Crown should exercise "a duty of care", or avoid negligence in its decision-making.
- Respect the principle of "reasonableness". This is an important term in relation to the fisheries. In so many stock assessment cases, we deal with probabilities — the science of uncertainty — there is no exact answer. The best that can be hoped for is a range within which the truth may lie. The "reasonableness principle" suggests that the courts will accept that a given stock assessment of biomass is not quite precise, provided that the method used to derive the estimate is "reasonable".

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Groups

The groups who probably have expectations of the Department of Fisheries and Oceans in relation to the nation's management of its fisheries are fishermen's associations, plant workers associations, environmental groups, etc. These groups will probably expect the following benefits from the Crown:

- Ensure cooperation in increasing the "size of the pie". By this is meant, working to ensure that the fish harvest is as large and as sustainable as possible.
- Ensure equity in dividing the pie or "fish harvest" between the various interest groups.
- Ensure participation in the decision-making process as it relates to:
  - stock assessment, and,
  - division of allocations between interested parties.
- Environmental groups will expect that the fishery be managed in an environmentally friendly manner.

Individuals

By individuals are meant people such as individual fishers. The benefits which they will expect from a nation's management of its fisheries are likely to be:

- An adequate return on their investment,
- A stable return on their investment,
- Security of title. By this is meant, a secure right to fish. In legal terms, a property title is considered to be complete if is:

1. **Durable.** For example, a fisheries licence which is owned by an individual but not used for twenty years. It should be just as valid and usable at the end of twenty years as at the beginning. The point is that it cannot be cancelled by a third party. If any party, such as the crown can prohibit this transfer, then the individual's right is incomplete.

2. **Exclusive.** This means that a right to fish is exclusive. Part of that right, such as a fisheries quota, cannot be transferred by a third party (the Crown) to another person.

3. **Transferable.** This means that the owner of a right to fish can transfer this right to a third party. If any party, such as the Crown can prohibit this transfer, then the individual's right is incomplete.

4. **Divisible.** This means that the owner of the right to fish can divide that right and sell or transfer a portion of it to another person. In any party such as the Crown can impede this division, then the fisher's right is incomplete.

5. **Flexible.** This means that the individual is free to use the right to fish in a reasonable manner. Thus, a fisher could contract with another person to catch the fish on his or her behalf. Any reduction in this right would mean that the right to fish is restricted in some way.

The above "SOGI" analysis has attempted to identify the benefits which various segments of society expect from the nation's management of its fisheries. Assuming that the benefits which these parties would hope to get from our management of the fishery have been correctly identified above, the next question is to identify the legitimate role for government in meeting these desired benefits. This involves first identifying the "role of the state" versus the "role of the private sector". The state obviously should only provide those benefits that fall within the state's role. Provided hereunder, is a standard identification of the role of the state used in the political science literature.

The Role of the State

1. **Protect the nation.** From the fisheries point if view, this means protecting the ocean borders of the nation. For example, ensuring that unauthorized foreign fishing does not occur within the 200 mile zone.

2. **Protect property.** This means ensuring that the nation's legislative, judiciary and police force ensure that our property is safe. For example, ensuring that squatters have not occupied our home while we are away at work.

3. **Protection of Person.** This is a function of the legal/police system to ensure that we are reasonably safe in public places.

4. **Protection against market-failure.** From the American Civil War to approximately 1873, many countries aspired to a "laissez-faire" market economy. The Vienna stock market crash of 1873, and the subsequent U. S. stock market crash in 1929, indicated a new role for the state in protecting against certain areas of market-failure. These are:

   - **The prevention of monopolies.** For example, do not give exclusive right to fish to one company.

- **Prevention of externalities.** This means ensuring that one group in society cannot reap private benefits and impose the costs on another group in society. For example, it would be the role of the state to ensure that one company does not harvest fish in a way that could damage habitat for the rest of society.

- **The provision of information.** That is, the provision of information which is necessary to make an informed market-decision but which is not available from sources in the private sector. An example of such information would be stock assessment advice. This allows the nation to make an informed decision on a reasonable harvest.

- **Provision of "public goods".** "Public goods" are goods which are of benefit to the nation, but are not profitable to the private sector to produce. Again, the provision of stock assessment advice could be considered to be a "public good".

From the above, I believe it is clear that protection of sovereignty at the 200 mile zone; the protection of property rights or quasi rights in the fishery; the provision of stock assessment advice; and, the prevention of "externalities" are all valid roles of the state in relation to fisheries management.

Having spoken of:

1. the benefits which clients seek from the management of fisheries, and,

2. the role of the state in meeting these benefits.

it may be worthwhile to spend a moment reflecting on the purpose of fishery science in helping the state fulfill "the role of the state".

Purpose of Science

The purpose of science is often portrayed as being to:

- **Describe.** This means collecting the basic data from which a scientific analysis can be based. In stock assessment, it is the step in which the research vessel data is collected and the "catch and effort" data is collected.

- **Understand.** This is the stage at which the data is analyzed. Trends, patterns or general rules are sought based on the data. In terms of stock assessment, this the stage at which we arrive at an understanding of the state of the stock.

- **Predict.** This is the stage at which we use our understanding of the state of nature and any general rules which we might have been able to develop, to predict how nature may behave at a point in the future. In terms of stock assessment, this is the stage at which we suggest what the harvest might be for the future year.

- **Control.** This is the stage at which we use our knowledge to control the behaviour of nature or, use our knowledge to control the effect which nature may have on us. For example, in stock assessment we might attempt to control, in some small way, the state of the stock, by taking a small harvest in a given year, in the hope of building up the stock for future years.

Conclusions

Since some of the benefits sought by the various client groups in the nation are mutually limiting, this requires a process which ensures the voluntary acceptance of compromises by those affected by the government's decisions. In this context, it is useful to remember that:

An implementable decision = its logic $\times$ its acceptance.

This means that those decisions which the affected parties will allow the nation to implement will have to have:

1. an adequate level of logic, and

2. an adequate level of acceptance.

In some cases, a less than perfect set of logic has to be used in order to gain acceptance by the parties affected.

Possible Courses of Action for the Scientific Community Involved in Stock Assessment:

1. **Describe.** It is suggested that scientists provide clients with an objective and public written description of the state of the stock.

2. **Understand.** It is suggested that scientists ensure that clients understand the status of the stock and the pertinent implications.

3. **Predict.** It is suggested that scientists provide estimates of the implications for the future of the stock of different but reasonable levels of harvest.

4. **Control.** It is suggested that scientists leave control of the decisions on:

    - harvest level (TAC) and

    - time period for which the "TAC" is sought,

to elected officials, in consultation with the various publics. In other words, it is suggested that we in the scientific community recognize, that it is our role to provide objective advice and it is the role of elected officials to make "allocative decisions on the division of scarce resources between competing interest groups in society".

Final Thoughts

- The Act establishing the Department of Fisheries and Oceans, and most other departments in the Canadian Government gives the Minister and not the public servants "the management and direction" of the Department. Consequently, it is the role of the Minister and not of public servants to make policy decisions affecting the fishery.
- It is wise to remember that the purpose of politics is "the allocation of scarce resources between competing interest groups by peaceful means". This is not the role of public servants.
- Finally, it is useful to remember that an implementable decision is a function of both its logic and its public acceptance.

Working Group Reports

Participants at the workshop were assigned to one of four working groups. The following questions were given to each of the groups to help focus discussion on the themes of Risk Evaluation and Biological Reference points. The structure of the discussions was informal — Chairpersons and Rapporteurs were invited to indulge themselves in how they approached the questions and how they documented their discussions.

1. Risk Evaluation

   (a) Quantifying the risks associated with advice as specific reference levels: any approach emerging as being superior (model-based, Monte Carlo, bootstrap, etc)?

   (b) What is an acceptable risk level? Who decides on the reference risk levels? Should managers let the risk float within pre-defined ranges or should acceptable risk levels be established and strictly adhered to and thereby act as a target?

   (c) Is risk analysis a tool for evaluating short-term or long-term options?

2. Biological Reference Points

   (a) Different harvest strategies lead to different fishery characteristics and stock conditions and, consequently, certain reference points may be better suited than others to meet a given set of fisheries management objectives. Typically, what are the social, political, economic and biological objectives of managing groundfish stocks and which reference points would be better suited to meet them?

   (b) Should a reference point be based solely on biological considerations or should they also include socio-economic aspects?

   (c) Reference points based on spawning stock biomass considerations: to be or not to be for (Atlantic) groundfish?

   (d) To what degree should one consider data reliability and the accuracy/precision of stock parameter estimates when choosing a biological reference point (and management strategy) and the appropriate control mechanism (tactics)?

Working Group #1

Chair: R. I. C. C. Francis
Rapporteur: R. Stephenson

The Working Group was composed of 14 people, chaired by Chris Francis. Francis explained that Chairpersons had been chosen purposely "with bias" (in that they were from outside Canada and with different backgrounds) and that they had been encouraged to "indulge their biases". The Working Groups have been purposely assigned members with diverse backgrounds and skills.

In discussing terms of reference for the group, the Chairman stated that it was not his intent to stick strictly to the assigned questions and the definition of Risk — but to focus on those questions which the Working Group members felt that they could answer.

The Working Group began its discussion by surveying the background and special interests of group members. The diversity within the group was reflected in the range of interests — from the definition and calculation of Risk, though integration of biology and socioeconomics in fisheries management, to the biology underlying biological reference points.

From the initial survey, it was obvious that distinction must be made between BIOLOGICAL REFERENCE POINTS and MANAGEMENT STRATEGIES.

This led to a lengthy discussion on MANAGEMENT OBJECTIVES or GOALS versus the STRATEGIES which may be employed to meet such goals. It seems to be difficult to define and separate goals, objectives and strategies. Biological reference points appear to have been promoted from strategies to objectives. There are obviously biological components within a package of societal objectives — but these should operate by way of constraints.

There was considerable discussion about the terms to use, but general assessment on the following scheme:

- MANAGEMENT should have clear OBJECTIVES, (set by senior managers) which are met by way of STRATEGIES.

- OBJECTIVES will contain both biological (e.g., sustainability of target species, multispecies or habitat considerations) and socioeconomic (e.g., maximizing net benefit /or employment, stability of quota etc.) elements; and these are overlapping issues.

- We (in Science) should not presume or establish objectives or socioeconomic strategies, but should be,

1. establishing biological constraints and strategies and
2. using our analytical capability to integrate and evaluate socioeconomic aspects which are given to us.

We imagined the following scheme:

	OBJECTIVES WITH			
STRATEGIES	Socioeconomic CONSTRAINTS (defined for use)	and	Biological CONSTRAINTS (of our definition)	(reference points)
VALUATION	decision on most viable strategy including risk estimation			

- Constraints would include a biological lower limit (the population necessary to meet the objective of a sustained resource).
- There would likely be socioeconomic constraints, as a number of the imagined objectives are not addressed by biological reference points.
- Reference points are considered as part of strategies.

During the discussion, the Working Group touched on a number of related points and issues, including:

- that objectives are often assumed, not specified; and that it would be useful to have clear management objectives;
- that there is a need to define a conservation "bottom line" or threshold, that $F_{0.1}$ doesn't guarantee fulfilment of the conservation objective;
- that there are biological constraints (i.e., what a population can deliver) and beyond that we need strategies for utilization of the resource (including reference points?);
- that it is perhaps inappropriate to think of a common reference point;
- if additional or different reference points are chosen, consideration must be given to their implementation (acceptance);
- that with both biological and socioeconomic constraints (under the model proposed earlier) there may be no feasible region; resulting in the need to stretch constraints.

The Working Group went on to discuss Risk.

The Group suggests a working definition of Risk — "A probability that something (bad) will happen within a given time frame". Risk estimation should not be confused with Risk ANALYSIS which is a formal procedure and includes the use of utility functions, etc.

Risk estimation is considered to be a useful way of translating our uncertainty into terms that managers and clients can understand.

The Working Group addressed the questions on Risk.

Q. 1a) Quantifying the risks associated with advice at specific reference levels: any approach emerging as being superior?

Answers:

- While there has been a diversity of approaches presented and the meeting has shown differences between some; and we have to attempt to understand reasons for these differences — no one approach emerges as universally applicable or superior;
- for assessment purposes, bootstrap may be advantageous (if there is enough data to characterize underlying distributions) in that distributions and associated parameters do not have to be assumed and it may be more easily defended.

Q. 1bi) "What is an acceptable risk level", what is inappropriate? It depends upon the situation.

Answers:

- Risk has several elements, reflecting biological and socioeconomic objectives.
- Some constraints will require more precise/severe risk levels than others (e.g., min. stock size).
- Risk reflects our uncertainty or ignorance. If we are not willing to accept any risk, we will not have chance of gain.

Q. 1bii) Who decides on risk levels? (delete the word reference)

Answers:

- We (in Science) can define probability functions, but it is up to Management (including mangers and users) to define utility functions which will allow risk assessment and decision on risk levels.

Q. 1biii) Should managers let the risk float . . .?

Answer:

- We cannot address this . . . not our business.

Q. 1c Is risk analysis a tool for evaluating short-term or long-term options?

Answer:

- Risk analysis should be of use in both.

The Committee addressed question 2c "Reference points based on spawning stock biomass considerations . . ."

Answers:

- Yes . . .
- Spawning stock biomass considerations are especially important as constraints (threshold, bottom line).

- As reference points, SSB considerations may be of use in defining "comfort" or "target" zones, especially in the long term.

The Committee decided not to address question 2d, as it was complicated and unclear.

The Committee concluded with a discussion of other issues which had come up:

(i) Socioeconomic Analysis

Biology has dominated fisheries management and the Group agreed that there was a need for socioeconomic analysis. At present, economic and social considerations are only being included in an informal way.

The working Group recommends:

1. that there should be more formal socioeconomic analysis, and
2. that socioeconomic analysis be presented jointly with (integrated with) biological evaluation.

(ii) Presentation of Risks to Managers

Risk evaluation is considered to be a useful way of translating our uncertainty into terms that managers and clients can understand. We need to list consequences — as many as possible — and how they fare under a variety of strategies. Only if clear about objectives, will we be able to evaluate the risk of strategies.

Working Group #2

Chair: Mikael Hildén
Rapporteur: Susan Farlinger

$$\text{RISK} = \text{P(BAD EVENT)} \times \text{BADNESS}$$

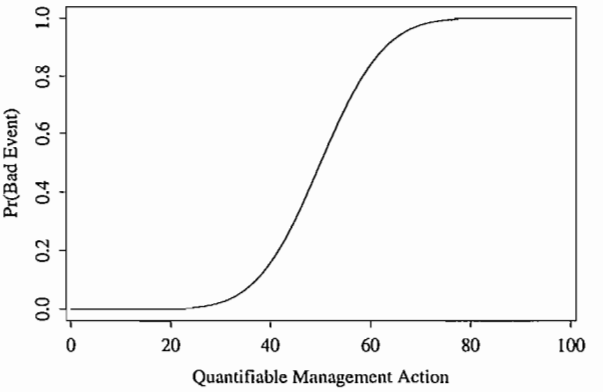
SPELL OUT:

- 'bad' event in appropriate terms/units
- probability of 'bad' event and it's effects (outcome)

i.e., LABEL YOUR AXES CORRECTLY

IT ALL DEPENDS

- ON CONTEXT: are relevant events and probabilities calculated?
- ON WHERE YOU ARE: stock status, fishery ...
- ON OBJECTIVES



The key is the analysis of consequences in probabilistic terms.

- biologists specify biological consequences.
- biological basis for other consequences examined by biologists, but implications should be the responsibility of the respective experts in that discipline.
- GOODNESS or BADNESS is meaningful only in relation to OBJECTIVES.
- consequences should be defined as short or long term and should be a function of stock and/or fishery objectives.

Some Discussion About Reference Points

Reference points are derived from a consideration of objectives, which are biological or translated into biological terms. They are *fuzzy*, and should be expressed in terms of the probability of an effect over a range (for example, true where a TAC is provided in a range where probability is not equal over the range).

Some characteristics of reference points follow.

- Relative reference points may be more stable — explore this.
- There is no argument for fishing higher than F_{max} or F_{msy}.
- Thresholds are essential and should be determined from biological/ecological considerations of sustainability.
- Targets arise from simultaneous considerations of all conceivable objectives, but trade-offs and interactions should be left explicit for consideration and not distilled into a single number or range.

Accuracy/Precision

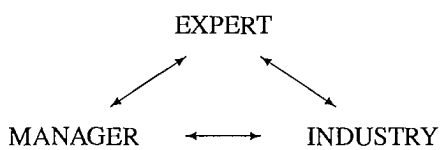
These factors affect possibilities for control, and **the ability to detect the desired effects of control**. They should be explicitly stated (accompanied by means of improvement where appropriate). When examining alternate control variables (effort, catch, ...) accuracy and precision should be considered. Accuracy and precision should be explored for all reference points.

Communication

Communication needs to be developed from a one way message :

EXPERT ⟶ MANAGER ⟶ USER

TO AN
INTERACTION



In order to foster this development, clear means of communication must be used appropriate for different levels of information and involvement. Ways of structuring information and presentations should be fully explored and utilized (i.e., time dedicated to this task).

Working Group #3

Chair: E. Pikitch
Rapporteur: G. Chouinard

Members of Group #3 met on November 21 and 22 to discuss and answer the questions posed by the Steering Committee of the Workshop.

General Discussion

The discussions of Working Group #3 started with some general comments of the topic and questions to be addressed. Participants agreed that fisheries management is a decision-making process which includes uncertainties. When providing advice to fisheries managers, it is not sufficient to merely describe the uncertainties but we should also predict the consequences of the possible management measures; this involves the concept of risk. Although risk was not clearly defined, the participants concluded that it was not the stock assessment scientist role to determine what is or is not an acceptable risk level. This decision should remain with fishery managers. It is, however, the role of stock assessment scientist to provide all available biological information and the implications of various actions so that fishery managers may make an informed decision. Scientist should present the information in a clear and concise fashion understandable by all concerned. Risk curves were considered better than confidence intervals to illustrate the uncertainty in assessments. The Working Group then focused its attention to the questions that were posed.

I. Risk Analysis

a) Quantifying the risks associated with advice as specific reference levels: any approach emerging as being superior (model-based, Monte Carlo, bootstrap, etc)?

The WG concluded that not all methods had been presented or explored in details and that no extensive formal analysis comparing various techniques was presented during the workshop. Furthermore, this aspect has been discussed in statistical circles and is still the subject of debate. Amongst the other methods that were not considered, Bayesian approaches and retrospective analyses were suggested as methods that should be explored.

It was also noted the best method will depend on the type of uncertainty one is trying to estimate. If only estimation error is being considered then bootstrap techniques may be appropriate. However, if estimation error and other uncertainties (process error, model structure, stock structure uncertainty, etc.) are included then the most appropriate techniques are likely to be Monte Carlo simulations and Bayesian methods.

The different methods may produce very different probability distributions of risk. The Working Group considered that management advice is less sensitive to differences in probability distribution generated by various techniques than it is to differences in probability distributions generated by a given method. In other words, the estimate of risk may vary between methods for a given strategy but management advice may be quite similar no matter which method is used.

b) What is an acceptable risk level? Who decides on the reference risk levels? Should managers let the risk float within pre-defined ranges or should acceptable risk levels be established and strictly adhered to and thereby act as a target?

As risk is usually evaluated in relation to some reference point, the discussion focused first on which reference point(s) could be used. It was concluded that an acceptable reference point could be a sort of "warning zone". This point could be defined either as a zone within which there would be a strong chance of stock collapse or, alternatively, a zone within strong chance of the stock being less productive in terms of recruitment. It was clearly stated that this reference point would not be a target level as in the present usage of the terminology but rather a point of reference used to evaluate risk. This zone

should be considered as a zone to avoid. The participants considered that the location of this point should be clearly identified for each species subject to fisheries management. It was suggested that for species with sufficient data that a level equivalent to F_{rep} or the minimum Spawning Stock Biomass (SSB) that produced good recruitment be used. Participants recognized such a data-based approach is useful as it relates directly to the stock in question but that it can also pose problems because it is limited to the range of the observations. For species/stocks where there is little or no data, the location of the warning zone could be established by analogy. In this respect, there is a need to conduct a large scale project to assemble and analyze empirical data to derive general rules that could be used to determine the warning zone. Work presented by Mace during the workshop was considered to be an example of the type of study considered.

The participants felt that it is clearly the role of fishery managers (i.e., government) to decide the acceptable level of risk of entering the warning zone. As for other types of risk related to harvest strategy (e.g., risk of exceeding current fishing) these could be provided and would involve interaction between biologists and fishery managers.

c) Is risk analysis a tool for evaluating short-term or long-term options?

Rather than risk analysis, the Working Group considered that decision analysis was the tool to use to evaluate options. This analysis would necessarily be used for both short and long-term options. Catch options in the short-term need to be assessed while a long time horizon is necessary to evaluate the performance of a harvesting strategy.

II. Biological Reference Points

a) Different harvest strategies lead to different fishery characteristics and stock conditions and, consequently, certain reference points may be better suited than others to meet a given set of fisheries management objectives. Typically, what are the social, political, economic and biological objectives of managing groundfish stocks and which reference points would be better suited to meet them?

Objectives of managing groundfish stocks will vary depending on which expert you choose to listen to (scientist, manager, industry, politician, etc.) and that any harvesting strategy or reference point proposed by the group may not be acceptable to one or more of the other players. The Working Group decided that the question should be reformulated to the following: "How would we go about devising a harvesting strategy?"

The participants concluded that this could be achieved only if there was a high degree of interaction between scientist, managers, industry and other interested parties, at all phases of the process. The most important steps were considered to be as follows:

1. List the objectives/outcomes of the strategy: a list of the objectives and/or results of the harvesting strategy would first be developed. These objectives could be of social, political, economic or biological nature and would be determined by all the interested parties. The Working Group suggested possible objectives/outcomes of the harvesting strategy that could be anticipated:

    - maintain catch constant.
    - reduce variability in catch.
    - maintain effort or employment constant.
    - reduce variability in effort or employment.
    - reduce variability in catch rates.
    - increase net landed value.
    - reduce levels of by-catch or discards.
    - the strategy is adhered to and it's performance can be monitored.
    - the strategy is understandable and acceptable.
    - the probability that the catch will equal the TAC is high.
    - the probability of stock collapse is low.

2. Prioritize the elements: A discussion of the list of objectives/outcomes would occur to rank the elements in order of importance. It should be recognized that ensuring that the stock is not depleted below a critical level (warning zone) would be amongst the first priorities. The establishment of the priorities would require the involvement of scientists and managers.

3. Develop harvest strategy: At the outset, it should be recognized by all parties that the harvest strategy will be long-term, that it will likely be a compromise among objectives and that it will take time and effort to develop it. The development of the harvest policy is seen as an iterative process whereby the role of the scientists would be to generate expected outcomes or expectations for particular policies that satisfy the priorities established in (2). The role of managers, industry and other parties would be to provide feedback to scientists on the properties of each policies so that variations can be explored if necessary. In this way several strategies can be compared and the implications of each would be known by all users. This approach is perceived to convey to users and managers that risks and trade-offs are part of decisions of harvest strategies. Since everyone is involved in the process, it may increase the commitment to the final harvesting strategy and enhance its understanding.

b) Should a referenee point be based solely on biologieal eonsiderations or should they also inelude soeio-eeonomie aspeets?

Soeio-eeonomie aspeets should be ineluded in developing a harvesting strategy. The eritieal level (e.g., warning zone) should be based on biologieal eonsiderations alone however it should be noted that it is not a target level.

c) Referenee points based on spawning stoek biomass eonsiderations: to be or not to be for (Atlantie) groundfish?

The Working Group generally felt that this eoneept is useful and is somewhat similar to F_{rep} or the 'warning zone" whieh were diseussed earlier.

d) To what degree should one eonsider data reliability and the aeeuraey/preeision of stoek parameter estimates when ehoosing a biologieal referenee point (and management strategy) and the appropriate eontrol meehanism (taeties)?

In general, it was eonsidered that the tools are not well suited to management needs. Assessments vary by more than 10% but often the management of a resouree will be re-opened if its status has ehanged by more than 10%. The management strategy and eontrol meehanisms should therefore be robust to data variability.

Working Group #4

Chairman: R. Arnason
Rapporteur: B. M. Leaman

Group #4 attempted to address the individual questions posed by the eonveners. However, we quiekly beeame enmeshed in lengthy diseussions about the merits of partieular referenee points and strategies, as they applied to speeifie situations. The breadth of these diseussions was largely related to the laek of an objeetive framework eoneerning how referenee points should be evaluated, and who assesses the risk of a given deeision. Therefore, we first eonsidered the definition of referenee points and the expression of risk from a more general perspeetive, and then developed some eriteria for the implementation and evaluation of these elements. Our diseussions eoneentrated on the types of referenee points presently in use, the eharaeteristies that make referenee points reliable, and some eonsiderations for their use. We also developed a working definition of risk, again with some suggestions on its presentation and use. Lastly, a number of areas requiring researeh were identified.

Referenee Points

Although there was great diversity of opinion on suitable referenee points, all partieipants plaeed primary emphasis on the need for detailed objeetives to guide the development and evaluation of management strategies. These objeetives must transeend the usual expression of desirable biologieal states and ineorporate the appropriate eeonomie and soeial goals. Indeed, many fisheries management problems arise from eonfliet between the strategies used to address explieit biologieal objeetives, and those needed to address implieit soeial or eeonomie objeetives. These underlying objeetives may not eonfliet direetly, but strategies for their attainment may require mutual aeeommodation.

Not all referenee points were viewed as equally valuable. We identified a number of features that eharaeterize good referenee points and their applieation. Referenee points will be most useful if they are based on well estimated parameters and stable eharaeteristies. For example, a referenee point whieh employs simple eomponents (M, mean age, ete.) may be more useful than referenee points employing parameters subjeet to greater uneertainty (biomass, reeruitment, ete.). Referenee points should be identified with respeet to the proeess involved (e.g., reeruitment, growth), so that managers know what features are being measured and eontrolled. Therefore, the strategies used to implement referenee points should be expressed in terms of the units used in management eontrol (e.g., eateh, mean size). While the best referenee points identify optimal eharaeteristies, it is not imperative that a referenee point identify these optima, simply that an appropriate direetion for neeessary ehange is elearly identified. We noted that developing fisheries represent unique ehallenges for management and require referenee points different from those for well-established fisheries. Information for the ealculation of most referenee points does not exist for new fisheries and managers will have to employ alternative teehniques, such as experimental fishing programs. Lastly, the group stressed that good referenee points should be durable and traetable. This means that they must maintain their value aeross a wide range of stoek abundanees, be measured with faeility, and be interpretable in a timely manner. Points whieh require large amounts of data and whieh are subjeet to signifieant time lags are less desirable than simpler referenees.

The diseussion group eonsidered three main eategories of referenee points for management: (i) status indieators, (ii) targets, and (iii) thresholds. Eaeh of these eategories eontains numerous referenee points, based on either rates or absolute values. We aeknowledged that eeonomic factors play a eritieal role in the performanee and management of fisheries. However, we separated eonsideration of eeonomie and purely biologieal referenee points, in reeognition of the more dynamie frame of evaluation for the former. In particular, the assumed diseount rate for valuation is highly influential and ean be very subjeetive.

i. Status indieators were viewed as eritieal to implementation of any management strategy, since they define the present states and are used to measure progress toward objeetives. They inelude sueh features as stoek biomass (total, exploitable, or spawning), mean eharaeteristies of fish in the harvest or stoek, age at entry, exploitation rate, yield, ete. Eeonomie status indieators inelude sueh variables as rent, employment, indireet revenue, ete.

ii. Target reference points were divided into rates and absolute values. Rate type references include $F_{0.1}$, F_{med}, F_{rep}, $F = M$, etc. They may also incorporate ecological considerations, such as fishing mortality rates which ensure desired predator-prey interactions among target and forage species, e.g., F_{pred}. Absolute targets refer to optimal biomass (e.g., B_{opt}, SSB_{opt}), maximum biomass (e.g., B_l, B_{max}), or catch (e.g., OY, C_{max}) levels. Economic targets include F_{mey}, $F_{rent} = 0$, B_{mey}, B_{max} employment, etc.

iii. Threshold reference points were regarded as danger zones to be avoided or points of warning, the approach to which should trigger remedial actions to change the existing trajectory of the stock or fishery. It was recognized that these references incorporate some uncertainty in either definition or impact. For example, minimum spawning stock biomass per recruit (SSB/B_{min}) is a threshold beyond which there may be strong concern that successful recruitment is imperiled. This uncertainty reflects an incomplete knowledge of the factors controlling recruitment.

No clearly superior reference point was identified by the discussion group. In part, this was because few sensitivity analyses of reference points have been conducted, and the records for application of any given reference are seldom lengthy or unobscured by other management interventions. Therefore, the group recommends that the evaluation of reference points move from the traditional consideration of equilibrium simulations, to fully stochastic scenarios. We also recommend that standard assessment output should include stock status relative to specific or generally useful reference points.

Several considerations for the use of reference points emerged from group discussions. The use of a reference point which is integrated across disciplines (e.g., biology, economics) is not advisable during the initial stages of developing a management strategy. Such integration carries necessary trade-offs and the nature of these trade-offs needs to be explicitly recognized in the development of suitable reference points. Integrated references may be superior but their internal compromises must be apparent to all participants. It is important to gain agreement on the use of particular references with all the participants in the management program. In particular, agreement on the appropriate courses of action dictated by the reference should be gained, prior to the need for these actions. The application of strategies based on a reference should also continue over a biologically meaningful time frame, e.g., at least one generation period. Finally, we should strive to minimize the number of interventions created by management programs, so that the effectiveness of decisions based on a particular reference may be interpreted.

Risk and Uncertainty

The majority of the discussion on risk assessment and the expression of uncertainty concerned the problem of who assesses the risk of a given decision. We believe, along with most other discussion groups, that risk is a combination of probability and consequence. It is the probability component that represents the expression of uncertainty, and that is most directly related to stock assessment activities. The consequence component is clearly multi-dimensional, and is conditioned on the objectives of those affected by the decision. There may also be competing alternative decisions, dependent on the clients' objectives and reference points. We therefore developed working definitions of risk and the client group(s) to which it applies.

"Risk": Risk is the probability of some relevant event occurring, conditioned on the appropriate reference point and client group(s), as a result of a given management decision.

"Client group(s)": Those who may influence the management decision chosen, and those who will be influenced by it.

A consensus of the group was that the valuation of risk was the legitimate role of fishery managers, in consultation with the client groups. The role of fisheries scientists is primarily to provide the probabilities of stock changes and the effects on yield, associated with particular management actions, but not to assess the acceptability of those actions. However, we stress that this role does not obviate the requirement for the scientist to clearly indicate when deleterious stock changes are likely to result.

The group did not identify any emergent superior technique for expressing risk. However, several important features of risk expression were identified. The major issue was viewed to be the evaluation of alternative decisions or courses of action. In this regard, the absolute accuracy of risk expression was viewed as less important than the provision of an ability to distinguish the appropriate choice from competing alternatives. The majority of the group also believed that the expression of uncertainty, in the normal form of confidence intervals, does not convey the risk associated with a decision appropriately. Rather, risk should be expressed as an increasing or cumulative probability. Thus, the approach to threshold reference points should be expressed as increasing penalty functions. Lastly, the group noted that risk will eventually require valuation in both ecological and socio-economic frameworks, due to species interactions and collateral effects on dependent fisheries.

Research Needs

The assessment and expression of risk are still in the initial stages for most fisheries. Several lines of research should be continued or initiated.

1. Continue model-based, Monte Carlo, and bootstrap-type investigations for the expression of uncertainty.
2. Conduct additional sensitivity analyses of the underlying assumptions that are used in the expression of probabilities.
3. Develop simulation models to evaluate the performance of decision rules and trigger points (e.g., 50% rule) presently in use.
4. Develop programs to implement experimental fisheries and adaptive management for the evaluation of alternative strategies. For developing fisheries, the learning period may be reduced through the use of programs involving Bayesian priors for particular strategies. For developed fisheries, experiments can be employed to reduce key uncertainties.

Session I
Biological Reference Points

Reference Points for Fisheries Management: The Western Canadian Experience

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Leaman, B. M. 1993. Reference points for fisheries management: the western Canadian experience. p. 15–30. *In* S. J. Smith, J. J. Hunt and D. Rivard [ed.] Risk evaluation and biological reference points for fisheries management. Can. Spec. Publ. Fish. Aquat. Sci. 120.

The use of *F*-based reference points in the management of marine fisheries on the west coast of Canada has been developed most extensively for groundfish stocks. Herring management relies on a fixed harvest rate policy, combined with a cutoff level for harvest prohibition at low stock biomass. The management of the diverse Pacific coast shellfish fisheries is based on a mixture of size limits, spawning escapement goals and fixed exploitation rates. *F*-based policies for groundfish generally take the form of $F = M$, $F_{0.1}$, or F_{opt}. The late recruitment age and longevity of many Pacific groundfish constrains the implementation and evaluation of traditional *F*-based management programs. For many species, cohorts have passed the age of critical size prior to entering the fishery, and the $F_{0.1}$ level may be either inappropriate or difficult to determine. Given the short history of management for Pacific groundfish (circa 1977), this also means that the results of some management programs are only recently being expressed in stock dynamics. Highly variable recruitment, previous histories of overexploitation, variable adherence to management targets, and short time-series of biological data have also hampered evaluation of these programs. Successful management needs to be based on: joint development of goals and objectives by management and industry; matched management tools and objectives; comprehensive programs encompassing all sources of fishing mortality; and, the use of reference points and stock indices that can be understood readily by client groups. It is also important that assessment advice be expressed in term of the risk of particular consequences, rather than the simple statistical error of estimates.

L'utilisation de points de référence fondés sur la valeur *F* pour la gestion des pêches marines de la côte ouest du Canada a été développée plus largement pour les stocks de poisson de fond. Le hareng est géré selon une politique établissant un taux de capture fixe, combinée à une interdiction de capture lorsque la biomasse du stock descend au-dessous d'un certain seuil. La gestion des différentes pêches aux mollusques et crustacés de la côte du Pacifique s'appuie sur un ensemble de mesures : limites de taille, objectifs d'échappées des géniteurs et taux fixes d'exploitation. Les politiques fondées sur la valeur *F* pour les poissons de fond s'expriment en général sous la forme de $F = M$, $F_{0.1}$ ou F_{opt}. L'âge de recrutement tardif et la longévité de nombreux poissons de fond du Pacifique sont une source de problèmes pour la mise en oeuvre et l'évaluation de programmes classiques de gestion fondée sur la valeur *F*. Dans le cas de nombreuses espèces, les cohortes ont dépassé l'âge de la taille critique avant d'entrer dans la pêche, et le niveau $F_{0.1}$ peut être inapproprié ou difficile à calculer. Étant donné que la gestion des poissons de fond du Pacifique existe depuis peu de temps (1977 environ), ce n'est donc que tout récemment que les résultats de certains programmes de gestion ont été exprimés sous forme de dynamique du stock. Un recrutement très variable, des conditions antérieures de surexploitation, un respect variable des cibles de gestion et une courte série chronologique de données biologiques ont également gêné l'évaluation de ces programmes. Une bonne gestion doit s'appuyer sur une élaboration commune des buts et des objectifs par les gestionnaires et l'industrie; des outils et des objectifs de gestion compatibles; des programmes globaux regroupant toutes les sources de mortalité par pêche, et l'utilisation de points de référence et d'indices du stock que les groupes clients peuvent facilement comprendre. Il est également important que l'avis d'évaluation soit exprimé en fonction du risque de conséquences particulières, plutôt que par la simple erreur statistique des estimations.

Management of most marine fish on the west coast of Canada is relatively recent. Management programs for offshore species, in particular, required the regulatory and enforcement regime that came only with the extension of Canada's zone of exclusive fisheries jurisdiction in 1977. The capabilities thus provided did much to halt the excessive exploitation of stocks by foreign distant-water fleets and transfer harvesting control to the domestic industry (Fig. 1; Stocker and Leaman 1990). However, many of these offshore species are characterized by slow growth and late ages at recruitment (10–20 yr), so that perturbations introduced by large harvests in the 1960's and 1970's were still being expressed in stock dynamics through the 1980's. We are only now beginning to see some of the effects of management programs instituted after the extension of jurisdiction. Therefore, the time is ripe to examine the performance of some of these programs and their attendant reference points. The workshop provided a forum for this examination as well as for discussions of how best to express the lessons learned, in the advice provided to fishery managers. This paper reviews the

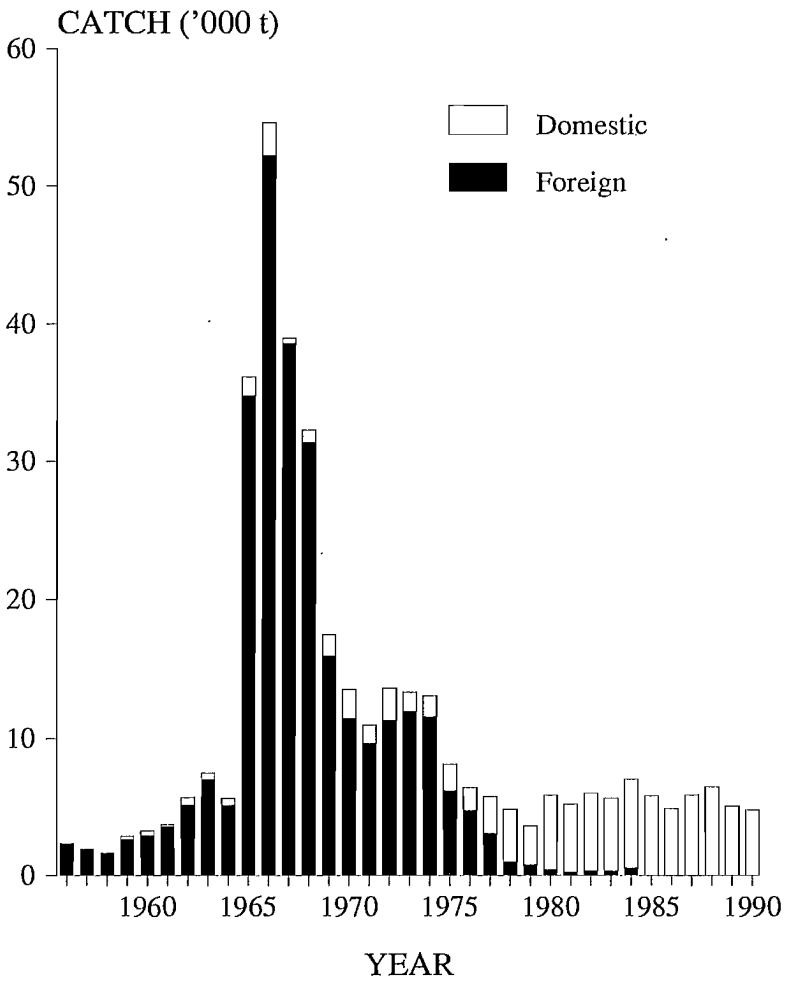


Fig. 1. Catches of Pacific ocean perch by foreign and domestic fleets off the British Columbia coast, 1956–1990.

western Canada experience in marine fisheries management and the use of particular biological reference points. The focus of evaluation will be on groundfish species but historical aspects of management programs for herring and invertebrates will also be presented.

The evolution of management strategies is a complex process and I focus here on only a subset of the issues governing the choice of a particular strategy, and its evaluation. To facilitate this examination, the workshop posed the following statement on biological reference points as background.

> Reference points can either be targets to be achieved or danger zones to be avoided. Fisheries management measures are supposed to help achieve the targets or avoid the danger zones.

To address the validity of this statement, the workshop concentrated on answering the following questions.

1. How many different fisheries management strategies have been used and what were their main characteristics?
2. Have the targets been achieved and/or the danger zones avoided?
3. What was the role of fisheries management measures in achieving/not achieving targets and/or avoiding danger zones?
4. What were the main reasons for success?
5. What were the main reasons for failure?
6. What role did uncertainties (about stock status, effect of management measures, catch statistics) play in successes and failures?

In addition to addressing these questions, I will attempt to identify the lessons learned from previous management programs and how they might be applied to the design of future programs.

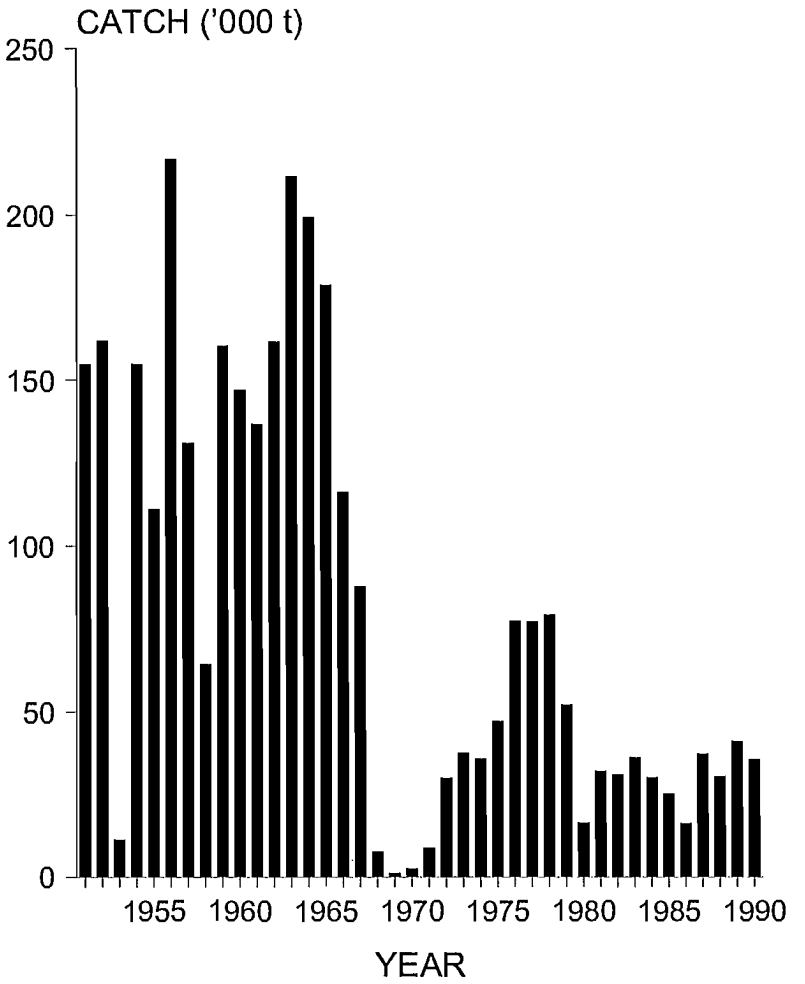


Fig. 2. Catches of Pacific herring in British Columbia waters, 1951–1990.

Types of Management Strategies

Herring

Historically, Pacific herring (*Clupea harengus*) was the object of a reduction fishery. However, high exploitation rates in this fishery and a series of poor recruitment years resulted in a stock collapse and closure of the fishery in 1968 (Fig. 2). Following the closure, the stock recovered and the fishery was re-opened in 1972. Pacific herring is a littoral and subtidal spawner, and undergoes an annual migration from offshore waters to spawn in British Columbia's coastal bays and inlets. The primary fishery is now a roe fishery on adults, when they are in these nearshore, pre-spawning aggregations. Minor fisheries are conducted for food and bait, and there is a small specialty fishery for spawn-on-kelp. Details of the assessment methods and management process for herring are presented by Stocker and Leaman (1990) and Haist and Schweigert (1991). The primary herring management objective is to ensure adequate spawning biomass. A target harvest rate (20% of spawning biomass) is applied to pre-spawning aggregations as long as the forecasted/estimated spawning biomass is above a pre-determined level, called the CUTOFF level (Table 1).

The CUTOFF level for estimated spawning stock biomass has been established as 0.25 times the estimated long-term unfished spawning biomass level. The target harvest rate "... is based on an analysis of stock dynamics, which indicates this level will stabilize both catch and spawning biomass while foregoing minimum yield over the long term." (Haist and Schweigert 1991, p. 36), although the 20% figure was in use prior to this analysis. As noted by Haist et al. (1986), "The exact population level defining the CUTOFF level is a matter of acceptable probabilities, rather than guarantees." Two independent assessment models (spawn escapement, age structured) are used in conjunction with recent-year recruitment trends, pre-season estimates of recruitment, and in-season acoustic surveys to estimate spawning biomass.

Table 1. Characteristics of the assessment and management program for Pacific herring off the west coast Canada.

PACIFIC HERRING STOCKS
- independent assessments with separable age-structured analysis and spawn escapement models (Haist and Schweigert 1991).
- Spawning Biomass (SB_i) calculated from weighted average of assessment models plus estimated recruitment of first-year spawners.
- harvest strategy uses fixed exploitation rate (μ_{opt}) and CUTOFF for SB_i:
$CATCH = \mu_{opt}(SB_i)$ if $SB_i \gg CUTOFF$
$CATCH = SB_i - CUTOFF$ if SB_i only slightly above CUTOFF
$CATCH = 0$ if $SB_i < CUTOFF$
- CUTOFF level is 25% of estimated unfished SB.
- μ_{opt} is 0.20.
- sensitivity analyses of μ_{opt} and bootstrap examination of input parameters.
- well-developed review and advisory structure (Stocker and Leaman 1990)

Other agencies on the Pacific coast follow similar management strategies. In Washington state, an F_{opt} level was originally determined to be equal to the rate of natural mortality, but was set subsequently at less than this rate due to ecological considerations about variability in herring recruitment, and its role as a forage fish (Trumble 1983). In Alaska, the 20% exploitation rate was apparently adopted as a compromise between proponents of either 0% or 40% harvest rates (V. Haist, Pacific Biological Station, Nanaimo, B.C.; pers. comm.).

Invertebrates

Fisheries for the numerous invertebrate species in British Columbia are characterized by an equally numerous set of biological reference points and management strategies, which are not generally defined in terms of target fishing mortalities. While fishing mortality or harvest guidelines are used widely in the herring and groundfish fisheries, invertebrate management is characterized by more static controls, such as minimum size limits and spawning period or area closures (Bernard 1982; Table 2). However, in two instances (shrimp, geoducks) management strategies address a target reference point (MSY/REC, MSY).

The control measures for many of the bivalve mollusc fisheries (minimum size limits) are intended to provide some guarantee of at least one spawning opportunity for recruiting individuals. Although this control criterion is well-defined, there is presently no documentation on whether such controls address any optimality conditions. The lack of formal evaluation of optimal fishing mortality strategies and targets is often related to the recent and explosive development of some of these fisheries. In many cases, fisheries have been established well in advance of any knowledge base for management. The development of valid age determination methods has been a significant problem for some of these fisheries. The consequent lack of knowledge about age composition, growth, and recruitment places severe limits on insight into population dynamics and yield. Field surveys are becoming a more widely used tool to estimate exploitable biomass. An exception to

Table 2. Characteristics of assessment and management for invertebrate fisheries off the west coast of Canada.

PACIFIC INVERTEBRATE SPECIES
1. Crustacean fisheries (Bernard 1982).
(a) Crabs (*Cancer* spp.):
- minimum size limit
- non-retention of females
- soft-shell stage closures
- savings gear (escape rings)
(b) Shrimps (*Pandalus jordani, P. borealis*)
- abundance surveys and μ_{opt}, based on yield per recruit
(c) Prawns (*Pandalus platyceros*)
- spawner index survey to set cohort escapement
- savings gear
- minimum size limit
- alternative management areas
2. Mollusc fisheries.
(a) Intertidal clams (many genera)
- minimum size limit, to allow at least one spawning
- restrictions on repetitive digging of beaches, to reduce juvenile mortality
- seasonal closures to limit pre-recruit mortality
(b) Geoducks (*Panope abrupta*)
- MSY management based on μ_{opt} and an estimate of B_∞ (Harbo and Fulton 1987)
- IQ management system
- rotational area closures
(c) Abalone (*Haliotis kamtschatkana*)
- minimum size limit to allow 2–3 spawnings
- season limitation
- quotas based on surveys and CPUE index
- IQ management system
3. Echinoid and holothuroid fisheries.
(a) Urchins (*Strongylocentrotus* spp.)
- minimum and maximum size limits
- quotas based on surveys and recruitment indices
- rotational area closures
(b) Sea cucumbers (*Parastichopus californicus*)
- "precautionary" quotas based on best understanding of biology

this generality is the fishery for geoducks, where management addresses maximum sustainable yield (MSY) through an optimal exploitation rate (μ_{opt}) times virgin biomass, where the μ_{opt} is defined as the rate of natural mortality (Harbo and Fulton 1987). Virgin biomass estimates were derived through density-area calculations based on surveys in unfished areas. Geoduck fisheries are also managed through an individual quota system and a program of rotational closures for harvesting areas has been instituted.

Crustacean fisheries off the west coast of Canada have the best knowledge base for management, among the invertebrate fisheries. Numerous measures such as minimum size limits, savings gear, soft-shell closures, and sex-specific retention controls act to preserve the breeding stock of female Dungeness crabs. Prawn stocks are monitored to permit minimum escapement of females from a cohort (these species are protandric hermaphrodites), directed to achieve MSY under yield per recruit theory. In addition, alternative management measures are being tested in several areas. Offshore shrimp fisheries are monitored by regular abundance surveys and yields are set through yield-per-recruit analysis.

Urchin and sea cucumber fisheries are the most information-poor of the Pacific invertebrate fisheries. Minimum size limits are in effect for urchins and some quotas are calculated on the basis of exploitation histories and observed recruitment rates. Sea cucumber population dynamics and stock levels are poorly known and management might be best described at this stage as "remedial incremental adjustments", as more information is gained.

Groundfish

The groundfish stocks of the Pacific coast were those affected most strongly by foreign distant-water fleets during the mid-1960's. Since exploitation by these fleets was well in excess of sustainable levels, management programs have had to address the approach to optimum stock biomass from the over-exploited side of the yield curve. The need for rehabilitation of these stocks resulted in recommended yields below the maximum levels that biomass of existing stocks are estimated to support. Excess domestic fleet capacity was therefore directed to previously unfished stocks and species. This process resulted in a suite of management challenges and a corresponding mix in the strategies and reference points used (Table 3).

Management of the groundfish fisheries generally employs F-based reference points to establish target harvest levels. The two most widely applied references are $F_{0.1}$ and F_{opt}, although the latter has several derivations. In addition, a number of F-independent measures are in place, including target spawning biomass and minimum size limits. For developing fisheries or fisheries which have operated historically at low exploitation rates, MSY reference points are still used. The Pacific coast has also seen the development of a several alternative management approaches, involving joint quotas for assemblages of rockfishes, habitat-based productivity estimation, and experimental fishing programs (Fargo and Leaman 1991).

One of the major difficulties constraining the use of traditional reference points for Pacific groundfish is that many species have passed the critical age or size (corresponding to maximum cohort biomass) prior to entry into the commercial fishery (Fargo and Leaman 1991). Rockfishes, which constitute over 50% of the domestic trawl landings, are typical of this phenomenon. For example, Pacific ocean perch (*Sebastes alutus*), achieves maximum cohort biomass at $\approx$ 8 yr but the point of 50% recruitment to the fishery does not occur until ages $\geq$ 11 yr (Archibald et al. 1983). This late recruitment appears to be behaviourally based, since fish are physically vulnerable to the fishing gear at ages $\geq$ 6 yr. Under a straightforward biomass maximization argument, F should be set very high to maximize yield per recruit, since no gain of cohort biomass will be gained through delayed harvest. When growth rates are low, simple market-size requirements may preclude such fisheries. Clearly, such an approach also fails to account for the life history strategy of such species, and alternatives are required. This requirement has fostered the development of some management programs which use reference points based on biological parameters, rather than those derived from analytic models. The most common strategy used is to calculate yield as $F_{opt} \times B_i$, where F_{opt} is generally taken as being equal to M (Gulland 1961, Francis 1986). Analytical studies, including sensitivity analyses of stock-recruitment parameters, have shown that M is generally less than F_{MSY} for rockfishes. For example, Archibald et al. (1981) estimated M to be 0.05 for *S. alutus*, while F_{MSY} was estimated to be 0.06 (Archibald et al. 1983). Lingcod quotas are also calculated using this philosophy.

Pacific cod (*Gadus macrocephalus*), although occasionally contributing up to one-third of the annual domestic trawl landings, has seldom been subject to management off British Columbia. In part, this has been because recruitment was perceived to be largely independent of stock biomass, and fishing effort responded directly to abundance. The latter tended to restrict fishing effort when abundance was low, and reduced the risk of recruitment overfishing. Recently, however, assessment biologists have concluded that fishing effort application lags population abundance by about one year, so that recruitment overfishing is more likely, and that environmental influences do act on spawning stock biomass to affect recruitment. In concert, these factors have generated the first quotas for major Pacific cod stocks. Quotas are based on the $F_{0.1}$ reference point (Foucher and Tyler 1991).

Management programs for flatfishes and sablefish are based on the $F_{0.1}$ or $F_{0.05}$ reference points (Fargo 1991, Saunders and McFarlane 1991). A size limit to decrease juvenile mortality is used for both of these species or groups. Both management programs attempt to project stock dynamics based on recruitment modelling. Recruitment is projected at several deterministic levels for sablefish and projected on the

Table 3. Characteristics of assessment and management for groundfish fisheries off the west coast of Canada.

PACIFIC GROUNDFISH SPECIES

1. General (Fargo and Leaman 1991)
   - all recommendations expressed as range from low risk – high risk
   - some experimental harvest programs (Leaman and Stanley 1993)
   - multi-year assessment and recommendation framework
   - some IQ management programs
   - quarterly quotas and variable trip retention limits
2. Pacific cod (*Gadus macrocephalus*)
   - $F_{0.1}$ strategy based on VPA and environmentally-driven recruitment model
   - variable seasonal closures to protect juveniles (Foucher and Tyler 1991)
3. Flatfishes (*Eopsetta jordani, Lepidopsetta bilineata, Parophrys vetulus, Microstomus pacificus*)
   - $F_{0.1}$ strategy based on VPA and yield per recruit modelling (Fargo 1991)
   - annual trip limits for some species
   - minimum size limit to decrease juvenile mortality
   - environmentally-driven recruitment model for some species
4. Sablefish (*Anoplopoma fimbria*)
   - $F_{0.1}$ and $F_{0.05}$ management strategy based on VPA and several deterministic recruitment scenarios (Saunders and McFarlane 1991)
   - minimum size limit to decrease juvenile mortality
5. Pacific hake (*Merluccius productus*)
   - separable SPA assessment model (Saunders 1991)
   - target spawning biomass strategy based on probabilistic sampling of recruitment series
   - F levels set through hybrid strategy with fixed F_{opt} when $>$ CUTOFF and variable F and when spawning biomass $<$ CUTOFF, i.e.,

     $$F_i = F_{opt}\left(SB_i/SB_{opt}\right)$$

   - low, moderate, and high risk levels set at $F = M$, F_{MSY}, and $F_{0.1}$, respectively
   - Canada-U.S. trans-boundary stock management
6. Rockfishes (*Sebastes* spp.)
   - $F = M$ and F_{opt} strategies
   - F estimated from length frequency analysis, CPUE analysis, depletion methods, and separable SPA
   - variable trip limits and quarterly quotas to prolong fishery
   - assemblage and experimental management programs
   - some habitat-based assessment
   - some sensitivity analysis of population parameters and SRR's (Leaman 1991)
7. Lingcod (*Ophiodon elongatus*)
   - minimum size limit to decrease juvenile mortality
   - seasonal closure to protect nest-guarding males
   - size-based mortality estimation
   - $F \leq M$ strategy for management

basis of an environmentally-driven model for some of the flatfishes.

Perhaps the most elaborate basis for management of the Pacific coast groundfish is that for Pacific hake (*Merluccius productus*) (Saunders 1991). The primary management objective is to protect the optimal spawning stock biomass. This level is defined as 0.1% of the long term female spawning biomass in the unfished state, as assessed through simulation. The strategy adopted to achieve this goal is a hybrid strategy of variable exploitation rate when the spawning stock is below the optimum and a fixed exploitation rate when the stock is above the optimum. The exploitation rate choice has been examined by simulation and is expressed as low, medium, and high risk. The risk levels correspond to the percentage of years the spawning stock drops below the optimum level. The F levels associated with these risk levels are $F = M$, F_{MSY}, and $F_{0.1}$, and correspond to 10%, 20%, and 30%, respectively. It is noteworthy that these simulations identified the $F_{0.1}$ reference as a high risk level.

For a number of groundfish stocks we have little biological or historical fisheries knowledge. These stocks (mainly some rockfishes and flatfishes) are managed on the basis of changes in indices, particularly CPUE or mean size. Catches are adjusted to maintain or reverse trends in the performance of these indices. This is the "remedial incremental adjustment" strategy noted for the invertebrate fisheries. A new modification of this approach is the calculation of fishery productivity by habitat area (Yamanaka and Richards 1992). This approach relates estimates of observed sustainable fishing to the amount of habitat available within historically-fished areas. The resulting standard is used to estimate productivity for new areas, on the basis of their habitat areas.

There have also been a number of attempts at assemblage and experimental management for rockfishes. Assemblage management assigned grouped quotas on the basis of single species assessments, modified by observed temporal or areal availability patterns (Leaman and Nagtegaal 1987). This program was applied to several rockfish stocks off the west coast of Canada during 1983–1985. Two experimental management programs involving periods of controlled overfishing, and unrestricted fishing followed by closure, have also been implemented (Leaman and Stanley 1993).

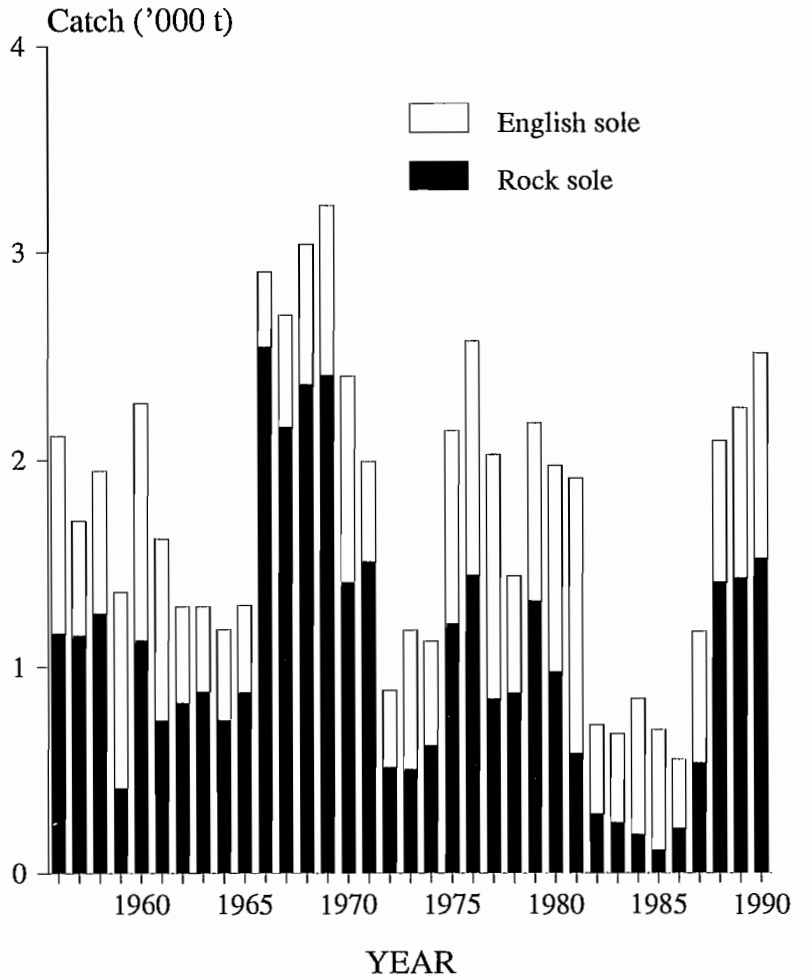


Fig. 3. Catches of English and rock soles in Hecate Strait, 1956–1990.

Finally, one of the major groundfish stocks (sablefish) is managed under an individual vessel quota (IVQ) system. This system allocates proportional shares of the available yield to licence holders, on the basis of historical performance of their licence and characteristics of their vessel. It permits licence holders to harvest their proportion of the overall quota at their own pace and allows control of catch to attain maximum price.

Achievement of Groundfish Management Objectives

The management objectives for some groundfish stocks off the west coast of Canada have been achieved, in the sense that the quotas used to achieve the target reference points have been adhered to (Fargo and Leaman 1991). Flatfish management objectives, in particular, have been achieved consistently for most species and stock productivity has been maintained. These stocks are managed using the $F_{0.1}$ reference point and landings have been controlled through annual quotas and trip landing limits. The primary area of productivity for flatfishes, Hecate Strait, has had relatively stable yields of English and rock soles since the 1950's (Fig. 3). During periods when quotas more restrictive than the long term average have been required, they have been implemented successfully and stock trajectories have been modified. However, the management program based on trip limits is sensitive to changes in effort patterns and may have multispecies effects. For example, fisheries yields for most flatfishes have been well controlled because the fisheries are directed at single target species. The flatfish fishery which has had the poorest record of management, although good by the standard of most groundfish, is that for English sole (*Parophrys vetulus*). This species is caught in conjunction with Pacific cod, which has a higher yield, hence quota overruns are created for English sole.

The record for many stocks is not as laudable as that for the flatfishes. The shortcomings of management programs for these stocks have two levels: failure to achieve targets and failure to avoid danger zones. Consistent and serious occurrence of the former inevitably leads to the latter. The severity of the problems created by these failings is dependent

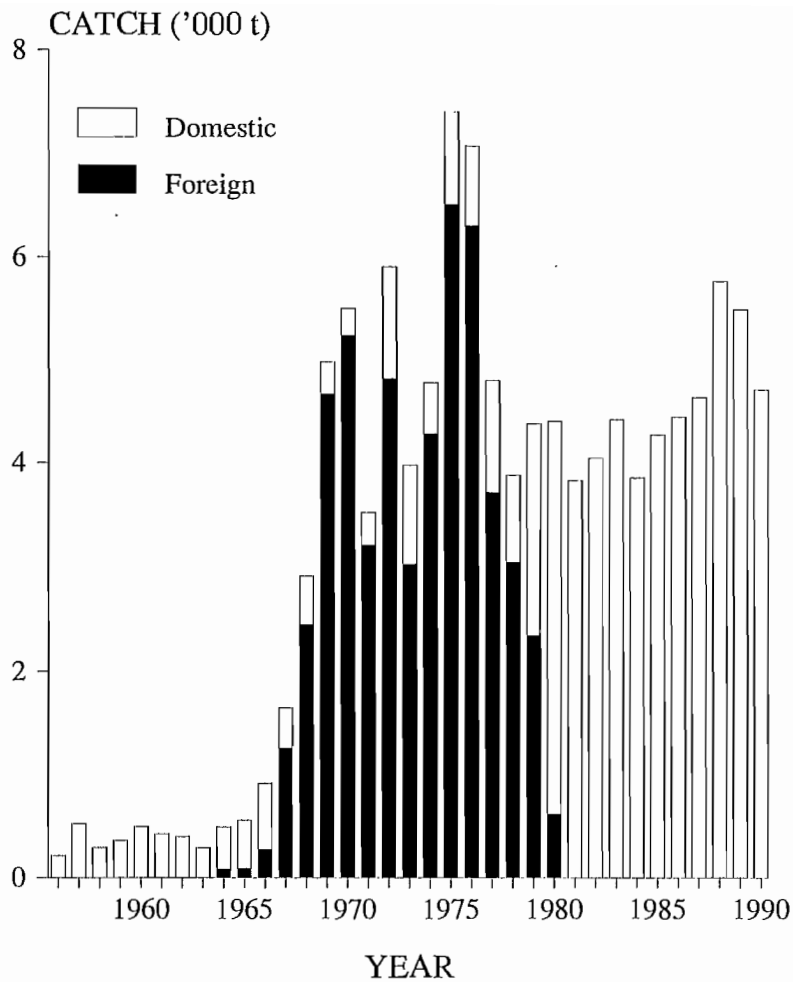


Fig. 4. Catches of sablefish by foreign and domestic fleets off the British Columbia coast, 1956–1990.

largely on the underlying biology of the target species. Highly dynamic stocks, such as Pacific cod and some flatfishes, are more resilient to departures from targets than slow growing species, such as the rockfishes. In addition, the absolute levels of yield for the target stocks influence the absolute magnitude of management precision. The precision of quota management is dependent largely on how landings, in the units of vessel trips rather than in units of tonnage caught, are controlled. Since vessel trips have relatively consistent average landings, control of the number of additional trips directed at stocks with small quotas, as the limit is approached, will have a greater effect on the precision of management than for stocks with larger quotas. Lastly, the management framework for each fishery also influences the precision of management. Management based on IVQs can be achieved with greater precision than for "Olympic" style fisheries.

For sablefish, harvest of the resource has moved from primarily foreign vessels to exclusively Canadian vessels (Fig. 4). While this transition has permitted greater revenue for Canadian fishermen, it has also decreased the control of harvest. Harvest control during foreign fishing was achieved through bilateral negotiations and put into effect by the client country on its own fishermen. From the Canadian perspective, this meant that control of the harvest, and penalties for poor control, involved only one entity — the foreign client. With a fully domestic fishery, control of harvest put into effect through each individual fishermen, and is therefore less precise. In large part, this accounts for the consistent quota overruns in the sablefish fishery. The IVQ management framework has largely eliminated these overruns, since its inception in 1990. The minimum commercial size limit for this fishery works effectively because capture mortality in this trap fishery is low and there is only a small recreational fishery. Similarly, the minimum commercial size limit for lingcod also controls pre-recruit mortality, but the historical lack of a similar limit for the large recreational fishery, the absence of harvest control for some gears, plus predation by an increasing population of marine mammals, hampers realization of some management goals.

The record of achievement of management goals for the ten major rockfish species has been mixed. For major stocks

Table 4. History of stock management for slope rockfishes (*Sebastes aleutianus, S. alutus, S. reedi*) off the west coast of Canada, 1978–1990.

	YEAR AND QUOTAS (T)												
STOCK	1978	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990
Southwest Vancouver Is.													
S. alutus Recommended	0	50	600	500	500	500	150–500	0–500	0–500	0–200	100–200	100–200	100–200
Adopted	0	50	600	500	500	500	500	300	C	100	150	150 (C)	0 (C)
Catch	–	125	430	547	508	752	551	243	242	542	308	279	290
Northwest Vancouver Is.													
S. alutus Recommended	n/a	n/a	n/a	n/a	250	250	250	250–500	250–500	250–500	200–600	200–600	200–600
Adopted	–	–	–	–	250	250	250	350	C	350	400	400 (C)	400 (C)
Catch	–	3	–	–	–	86	193	313	1046	451	492	994	877
NW Vancouver Is./QCSd													
S. reedi Recommended	n/a	100	100	250	250	200–500	200–500	200–500	200–500	250–750	250–750	250–750	250–780
Adopted	–	–	–	–	250	250	250	350	C	350	375	500 (C)	500 (C)
Catch	17	10	28	5	228	628	458	717	1208	1170	575	984	969
Queen Charlotte Sound													
S. alutus Recommended	2000	2000	2000	1500	1000	1100–2000	1100–2000	1050–1800	1050–1800	600–1150	700–1000	700–1000	700–1000
Adopted	2000	2000	2000	1500	1000	1250[1]	1050[1]	1200[1]	C[1]	850	850	850 (C)	850 (C)
Catch	1240	1431	1531	1481	856	1874[1]	1318[1]	1540[1]	1831[1]	1646	1186	1173	1409
Moresby Gully													
S. alutus + Recommended	n/a	n/a	n/a	1800	2000	2000–2800	2000–2800	1900–2800	1900–2800	1900–2800	2160–3500	2160–3500	2160–3500
S. reedi Adopted	–	–	–	1800	2000	2250	2250	2250	C	2250	3250	3350 (C)	2780 (C)
Catch	–	–	–	2217	3626	2422	2393	2199	729	2027	3432	1682	1547
Queen Charlotte Islands													
S. alutus + Recommended	1500	1400	1400	1400	1400	900–2500	900–2500	900–2200	900–2200	900–2200	900–1700	900–1500	900–1500
S. reedi + Adopted	1500	1400	1400	1400	1400	1400	1400	1400	C	1400	1400	1250 (C)	950 (C)
S. aleut. Catch	3518	1420	1398	1532	1309	1498	1383	1482	1475	1408	1410	1050	1025
Langara Spit													
S. alutus + Recommended	0	10	400	200	200	F	F	F	F	F	200–270	200–270	200–270
S. aleut. Adopted	0	10	400	200	200	F	F	F	F	F	F	F	F
Catch	–	241	87	207	411	419	2413	2404	3159	1310	1556	2061	1678
Coastwide													
Adopted	3500	3460	4400	5400	5600	5900	5700	5850	5000	5300	6425	6500	5480
Catch	4775	3230	3474	5989	6938	7679	8709	8898	9690	8554	8959	8223	7795

C = Coastwide quota and fishing limits
F = Free fishing experiment
[1] Includes *S. reedi*

depleted previously by foreign distant-water fleets (primarily *S. alutus*), there has been no indication of rehabilitation, although reconstruction goals and trajectories have been identified (Archibald et al. 1983). Quotas for continental slope rockfishes (*S. aleutianus, S. alutus, S. reedi*) have generally been exceeded, often by substantial amounts (Table 4). Quota management for continental shelf rockfishes (*S. brevispinis, S. flavidus, S. pinniger*) has been more successful but the limited knowledge base used to establish some of these quotas has resulted in downward revisions as more information is acquired (Stanley 1991). In this regard the danger zones have not been avoided, although this has resulted from incomplete knowledge rather than a failure of the reference point. The domestic fleet has also transferred effort to species which were less desirable previously, in order to maintain the volume of landings. Some of these species (*S. proriger, S. entomelas*) either do not presently have quotas, or have not been the subject of stock assessments due to resource limitations. Given the sensitivity of these long-lived species to fishing pressure, it is probable that some stocks of these species will be overexploited before management controls are applied.

Stocks of the major inshore rockfishes (*S. caurinus, S. maliger*) in the inside waters of Vancouver Island are in a depleted state, largely as a result of the explosive growth of fisheries, prior to management control. This situation may be regarded as a failure of the reference point, because it could not be measured or indexed with sensitivity sufficient to avoid danger zones. The use of habitat-based productivity quotas has only recently been adopted and it is too early to evaluate its success. On its own, this method is likely to be effective only to the extent that past productivity in reference areas will reflect productivity in areas to which they are applied. However, it does represent a useful tool for control of developing fisheries.

Management of Pacific hake has generally achieved the goal of maintaining the optimum spawning stock biomass (Saunders 1991). This achievement is particularly noteworthy because it involves joint assessment, harvest, and management of this transboundary stock by the United States and Canada. As noted by Stocker and Leaman (1990) this achievement was facilitated by the coincidence of extended fisheries jurisdiction and the occurrence of several strong cohorts in the stock. Scientists of the two countries have generally reached agreement on the available yield from the stock. However, allocation issues between the two countries have become one of the dominant features of stock management, as the national fleets have supplanted foreign fishing and achieved the capacity to harvest all of the yield quickly.

The success of assemblage management programs, in terms of achieving management goals, was mixed (Leaman and Nagtegaal 1987). Management by assemblages did achieve greater adherence to species quotas than had been achieved with single species management, for some but not all assemblages (Fig. 5). In those instances where improvements were not noted, the system performed at least as well as the traditional method. Assemblages were defined by cluster analysis of commercial catch statistics, rather than through ecological linkages, and simply attempted to capture the reality of what species were actually caught together. As such, their usefulness in the long term would be limited by the stability of the ecological relationships in effect at the time.

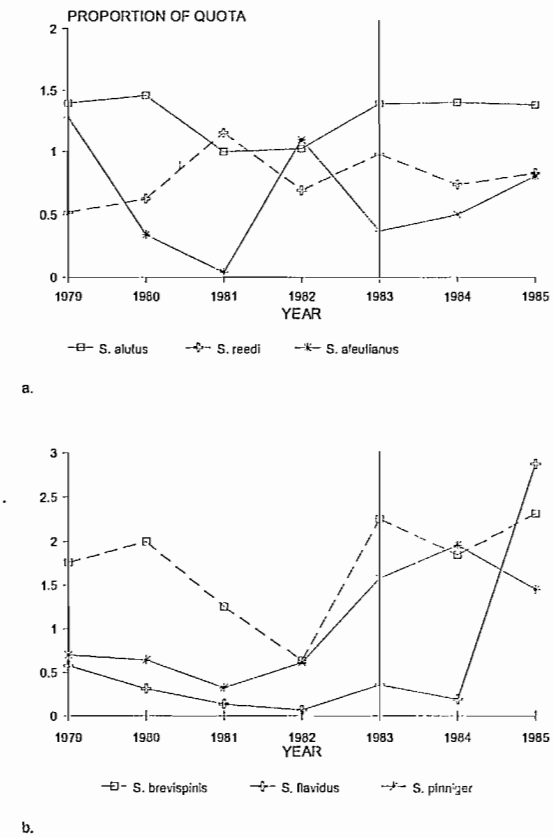


Fig. 5. Comparison of catches during single-species and assemblage management of several rockfish species off British Columbia. Single species management was in effect during 1979–1982, while species were managed as two assemblages during 1983–1985. Catches are expressed as proportions of the assigned quota. a. Slope rockfish management off the west coast of the Queen Charlotte Islands. b. Shelf rockfish management off the northwest coast of Vancouver Island (after Leaman and Nagtegaal 1987).

The opportunity for iterative improvement of the assemblage management approach, particularly through seasonal adjustments of proportions, was abbreviated when a simple trip limit violation charge against a fisherman resulted in a judgment that DFO had not legally defined the assemblages in its regulations. However, the merits of assemblage management are once again being investigated under a proper regulatory framework. An evaluation of the other experimental management programs implemented on the west coast is presented elsewhere in this volume (Leaman and Stanley 1993).

Role of Fisheries Management

Groundfish assessment biologists and fisheries managers on the west coast of Canada work very closely together in the development of the Groundfish Management Plan. Managers participate in the meetings of the Groundfish Subcommittee of the Pacific Stock Assessment Review Committee (PSARC), where the stock assessments are presented and subjected to peer review by internal and external scientists. In turn, assessment biologists participate in meetings of the Pacific Region Groundfish Advisory Committees (GAC's), where industry representatives work together with DFO staff to develop the final Management Plan. While cooperation among these parts of the management team in developing the Management Plan and establishing targets for the fishing year is good, several factors hamper the achievement of longer term goals.

Fisheries off the west coast of Canada are dominated by shorter-lived species such as salmon and herring. The relatively recent development of groundfish management, with its unique requirements for very long-term management policies, has required a concomitant development of management goals and objectives. However, harvest capacity developed much more rapidly than management control, and was well in excess of available yield. This mismatch in time scales led to the establishment of some implicit industry objectives, which were not mirrored in those of management. For example, the groundfish industry desired year-round employment and large average landings, which were incompatible with existing harvesting capacity and yield levels. Therefore, one management goal is to reduce fishing effort to levels commensurate with yield, a much more difficult goal to achieve than one of increasing effort. In addition, a strategic initiative yet to be resolved is whether some species will be managed as a collection of single species quotas, or as an assemblage whose composition may be dynamic. Such a strategy must consider the potential ecological shifts and economic impacts that may result from differential species' and area productivity within the assemblages.

Fisheries management has generally reacted quickly to nascent problems of quota overruns, although delays in implementation of management measures have meant that F_i was often greater than F_{opt}. The delayed development of long term goals has also resulted in a mismatch of management tools and stock productivity. For example, the stock structure of rockfish species is complex, and there may be as many as 30 different management units. Such complexity, when coupled with different exploitation histories, variable species' productivity and quotas, limited enforcement capability, and multispecies catches, has created serious optimization problems for the *Sebastes* group. Some quotas have been consistently undersubscribed while others were exceeded. In addition, there were strong economic incentives to falsify records of catches for similar species, to avoid some quota restrictions. The result of these factors has been an inability to achieve stock-specific management for this group. The response of the GAC to this problem was to institute a series of coastwide trip limits, so that enforcement was localized to ports of landing. This measure also tended to remove incentives to spread fishing effort in relation to stock yields. Instead, vessels fished those areas offering sufficient return that were nearest to ports. It is a simple fact of rockfish stock distribution that those stocks with the highest yield potential are the farthest from major ports. The use of coastwide trip limits as a management tool implies that long term fisheries productivity is not dependent on optimization of area or stock-specific productivity. If additional measures to balance yields among stocks are not instituted, this mismatch of area productivity and management tools will have long term consequences for these stocks. These consequences may be acceptable in the context of overall management but they must be anticipated, acknowledged, and evaluated by the participants.

Management has limited ability to restrict the increases in fishing power associated with technological change. Thus, even with limited entry programs in place, fishing power of the fleet will inevitably increase. Quotas will be subscribed quickly and the effectiveness of licence limitation will be lost. The typical response to this problem by both industry and management is to institute measures which reduce the rate at which quotas are caught, e.g., to decrease the efficiency of the fleet through restrictive trip limits. This approach treats the symptom (shorter fishing seasons) rather than the problem (too much fishing effort). The long-term solution to this problem is to tailor fleet capacity to available yield, or to implement alternative management frameworks that permit the fleet to regulate its own effort, such as IQ-type programs. However, these alternatives may not always be possible within multispecies fisheries and profound changes in the structure of these fisheries may be necessary.

Allocation issues between fleet components or geographic areas are also common to west coast groundfish management, similar to most other areas of the world. Differential fishing capability between vessel classes, areas, and gears generates conflicting demands on managers. Bycatch mortality of species by one gear type restricts yield available to directed fishing by other gears. A large boat/small boat conflict in seasonal fishing capability (hence access to quotas) has resulted in accommodations to both groups, further increasing the complexity of the management plan. Solution of these problems will be a test of the creativity and flexibility of all members of the management team. Allocation issues between the U.S. and Canada also exist and have begun to affect the Pacific hake fishery. In 1991, a failure to reach agreement on the proportional allocation between the two countries resulted in overharvest of the estimated sustainable yield. Both countries wish to avoid a repetition of this occurrence and the search for a mutually satisfactory solution is underway.

A final role of fisheries management is the development and promulgation of the regulations necessary to enforce the provisions of the management plan. In several instances, notably the lack of a legal definition for rockfish assemblages,

management programs have failed because necessary regulations either did not exist, or were found to be legally invalid. This regulatory process is an ongoing adjunct of the development of effective management programs, and can often be a critical factor in their success.

Reasons for Success and Failure

Successful stock management is based on a clear and shared understanding of management goals among the industry participants and managers. The instances of management success for groundfish fisheries on the west coast (flatfishes, sablefish, hake) are associated with well-stated objectives and clear goals for management actions. The relative success for these species is also related to the discreteness of their fisheries. Fisheries which either use gear that is highly selective for the target species, or exploit characteristics of the species behaviour to increase the selectivity of the gear, have been more manageable than fisheries which produce additional mortality on non-target species. In this vein, comprehensive management of removals by all gears has been a hallmark of success for species such as herring, halibut, and sablefish. A relatively uniform fleet capability (e.g., through vessel size) also enhances management, since opportunity costs for the resource users are approximately equal. Diverse fleet capabilities creates the need for equally diverse management measures. For example, the International Pacific Halibut Commission employs trip limits scaled to five different vessel size classes for some of its fisheries (IPHC 1991).

The biological nature of the target species is an integral component of management success. Highly productive species, such as Pacific cod and some flatfishes, are more forgiving subjects for fisheries than longer-lived species. The reasons for this are operational as well as biological. Species with rapid dynamics present more opportunities for the iterative process of application and refinement of management programs. For example, a cohort of rock sole may contribute to a fishery for only 4–5 yr (Fargo 1991) whereas Pacific ocean perch cohorts are only 50% recruited to the fishery at age 11 yr, and may contribute for another 20–30 yr (Archibald et al. 1983). The window of strategic opportunity for management, which includes both remedial action and response of the population to it, has been estimated at > 20 yr for rockfishes (Leaman 1991), whereas it would be approximately 8–10 yr for rock sole. This shortened period of management learning can be crucial to gaining credibility and cooperation with industry. Similarly, the sensitivity of indices used to monitor fisheries is related to species' life histories. Where annual recruitment is a small proportion of biomass and species are aggregating, indices such as CPUE are likely to be of limited value in detecting population changes. A low age at recruitment will enhance the sensitivity of indices used in stock monitoring.

A lack of exploitation prior to the implementation of management can be a major factor in management success, since participants are able to generate management goals jointly. This can help to avoid the development of competing goals by different sectors of the industry. However, even a previous history of over-exploitation can be a valuable asset to management, when the present participants were also accountable for that exploitation. Industry then has a collective memory of the consequences of excessive fishing mortality and can condition their expectations with this experience. This has been the case for some of the flatfish and herring fisheries of the Pacific coast, where stock declines have occurred in the memory of present participants. If management actions (or even fortuitous events coincident with management actions) have resulted in stock recoveries, the credibility of management is enhanced and participants are more receptive to subsequent management adjustments.

Economic issues have also played a role in management success for Pacific coast groundfish. Manageability of species is often related both to the ease with which species can be segregated into market categories, thus in the catch statistics used for monitoring and assessment, and to the magnitude of the economic incentive to misreport catches. Species that are well recognized and segregated in the market place are more likely to be reported accurately than several cryptic species of similar value. For example, there are several red-coloured slope rockfishes that do not have unique market value, but which have significantly different quotas and productivity, and are caught together. A common feature of rockfish management is an increase in the reported landings of these secondary species after the quotas for the major species are fully subscribed. This scenario is highly unlikely without misreporting or discarding (hence high mortality) of the major species. Indeed, it is this unavoidable mortality that is the incentive for assemblage management proposals.

The reasons for failure of some management programs for west coast groundfish are generally the mirror images of the reasons for success listed above. The characteristics of these shortcomings are: a lack of stock or species management objectives (individual species vs. assemblages); mismatched management tools and management objectives (coastwide trip limits vs. stock-specific quotas); competing industry goals leading to complex management programs (seasonal allocations to accommodate different vessel sizes); lack of comprehensive management framework for all gears harvesting a given species (uncontrolled incidental mortality in non-target fisheries); and, insensitive reference points and indices of stock status (late recruitment, longevity, and aggregation effects on CPUE).

For the rockfishes, which are the mainstay of the Pacific trawl fishery, a major factor in management is the over-exploitation by large, foreign factory trawlers which occurred in the 1960's. The present generation of fishermen did not participate in that fishery and many are unconvinced that fishing effort by the smaller domestic trawlers can produce fishing mortality sufficient to impact the productivity of stocks. The generation of experimental management programs for two stocks of Pacific ocean perch (Leaman and Stanley 1993) was

largely a result of joint efforts by industry, managers, and biologists to evaluate the impact of fishing effort by the domestic fleet on rockfish stocks.

Comparison of the successes and failures of groundfish management for the west coast of Canada, with those in other areas of the world, is hampered by the short history of management off the west coast. Some aspects of this management performance were reviewed by Stocker and Leaman (1990). For those stocks where some comparisons can be made, management appears to have been as effective (or ineffective) as it has been in other areas of the world. Stocks of long-lived species (e.g., rockfishes) which were over-exploited previously by foreign distant water fleets, exhibit population dynamics that are still dominated by interventions from the mid-1960's and early 1970's. None of these stocks has been rehabilitated, although most have been maintained. The precision of quota management for these stocks has been poor and the value of the reference points used cannot be accurately evaluated. Total rockfish production has actually increased through discovery of the few stocks not exploited previously and by landing of species not previously marketable. Performance of rockfish management off the west coast of the United States has been even poorer than that off Canada (Pacific Fishery Management Council 1991). Stocks of rockfishes exploited traditionally have continued to decline and quota restrictions are severe, relative to catching capacity. These declines in rockfish production have been mitigated somewhat, as in Canada, by the development of markets for non-traditional species.

For the major gadid species off the west coast of Canada, Pacific cod, there has been no history of management and quotas were imposed in 1992 for the first time. Data presented in Foucher and Tyler (1991) show the west coast cod stocks undergoing substantial fluctuations in abundance, which the authors attribute to factors external to the fishery. In contrast, Garrod (1988, p. 213) provided good evidence that changes in abundance of similar proportions for gadid stocks in the North Atlantic were ". . . heavily influenced by exploitation.", although multispecies and environmental effects were also believed to be important. While it could be argued that unmanaged Pacific cod stocks have performed as least as well as managed Atlantic cod stocks, it is important to note that fishing effort off the Pacific coast has been partially restricted, by fishing licence limitation, since the 1970's. The recent introduction of the first quota for Pacific cod was strongly influenced by increasing effectiveness of fishing effort brought on by gear and technological innovation.

The problems of restricting total fishing mortality for stocks under international management, which have resulted in overexploitation, are also being visited on the west coast Pacific hake stock. While the total yield from this stock is not in dispute between U.S. and Canadian scientists, there is continued disagreement over an objective and equitable allocation formula for this total. This disagreement has resulted in an over-subscription of the most desirable yield level in 1991 and 1992. These problems appear to be a common feature of such management in other areas of the world (Gulland 1988).

Role of Uncertainty

Fisheries stock assessment is largely an interpretive process based on attempts to find consistent answers from analyses of different data sets, all of which are subject to degrees of uncertainty. The types of uncertainty or error are categorized generally as measurement uncertainty and process uncertainty. Most observations of fisheries available to assessment biologists have been filtered by some form of measurement error, including the estimation of fishing effort, catch by species and age group, etc. The interpretation of observations is also influenced by uncertainty in the description of the underlying processes of stock and fishery dynamics, such as the relation of stock and recruitment, vulnerability and availability to fishing gear, etc. These effects are visited on both research and commercial data sources and impact the management of stocks in two ways. First, they create legitimate uncertainty in the estimates of stock biomass and yield, so that the assessment biologist may be unable to provide advice with the precision requested by the manager. Second, the industry may not have ready access to an index which provides a reliable correlate of stock condition or changes to it. Indices which the biologist finds reliable may result from extensive post-collection processing of the only data available to the industry. For example, a fishing master may interpret CPUE in its raw form while the assessment biologist may subject the same data to extensive standardization for vessel horsepower, electronic aids, net type, season, etc. This standardization may yield an interpretation which is quite different from that based on the raw data. This different interpretation of the "same" data often provides the basis for disagreement between industry and the management agency over stock trajectories and yield.

Uncertainty has played a significant role in the management of those species with slow dynamics and longevity. The major feature of this has been the long time delays between the imposition of fishing mortality effects and their manifestation in the indices used to monitor the stock. For example, many species of rockfishes and bivalves may have a very protracted recruitment and be 10–20 y old before they are fully recruited to the fishery. Consequently, high fishing mortality may reduce abundance substantially before recruitment declines, or the age composition is truncated (Harbo et al. 1983; Leaman 1991). This delayed insight into the effects of fishing contributes to uncertainty in the choice of management action. Because the history of stock management for many groundfish species has been one of rapid over-exploitation, managers of developing fisheries have generally opted for what industry may view, based on their high initial catch rates, as a conservative approach. In many instances this approach is consistent with appropriate long-term yields but industry has understandably questioned the information base upon which it may be based. In these instances, reference is often made to previous experiences with the same or similar species but this is seldom satisfactory to all parties. The impact of this

uncertainty is increasing pressure on managers for new analyses and "interim" measures. Once instituted, such measures may be difficult to rescind, particularly when they are in effect over several years.

Industry has also become more knowledgeable about where uncertainty exists and what the impacts of uncertainty at various stages of the assessment and monitoring process are. This has had overwhelmingly beneficial effects on the quality of management. For example, if we consider a simple model of optimum biological catch, fishing rate and biomass:

$$C = F_{opt} \times B_{opt},$$

there is potential uncertainty associated with each of the three variables. Historically, industry may have directed its primary criticism at the estimated biomass level. However, as industry knowledge and sophistication has increased, so has the requirement for better documentation and analyses of all parts of the yield calculation. This has stimulated assessment biologists to re-examine the basis for some components of the assessments, and has resulted in the modification of some sampling programs to provide data of higher quality (e.g., Stanley 1990). A second beneficial aspect is that the acknowledgement of uncertainty by all parties has encouraged some joint approaches to reducing the uncertainty of various estimates, including experimental fishing programs (Leaman and Stanley, this volume).

The recognition and acknowledgement of uncertainty improves the process of providing useful management advice. However, it also has the potential for negative effects, if it is not presented in an appropriate form. Biologists often spend considerable time presenting the statistical error of a particular estimate, in an attempt to convey the idea that quantities are not known with absolute certainty. Managers have become accustomed to seeing confidence intervals or error bars around a point estimate, such as in Fig. 6a. The intervals shown in this figure are centered on the point estimate, and extend approximately two standard errors to either side of the point estimate. If we assume a normal distribution for the errors then, on average, the interval will cover the parameter value with 95% confidence.

Two features of the intervals in Fig. 6a merit comment. First, the lower bounds of the intervals for three of the species are less than zero. This unhappy result can be avoided by either choosing an alternative distribution for the errors (one that recognizes zero as a lower bound for the potential observations, such as the negative binomial), or obtaining confidence bounds via a resampling procedure (e.g., bootstrapping). An incorrect procedure used commonly is to truncate the interval, calculated using the normality assumption, at zero. This yields an interval of unknown coverage properties and bias. The second feature of the intervals in Fig. 6a is that there is no reason to believe that a value 100 t less than the point estimate is any less plausible than a value 100 t greater than the point estimate. However, the impact of decisions based on these two alternatives, in terms of the risk to future stock yields, may be vastly different.

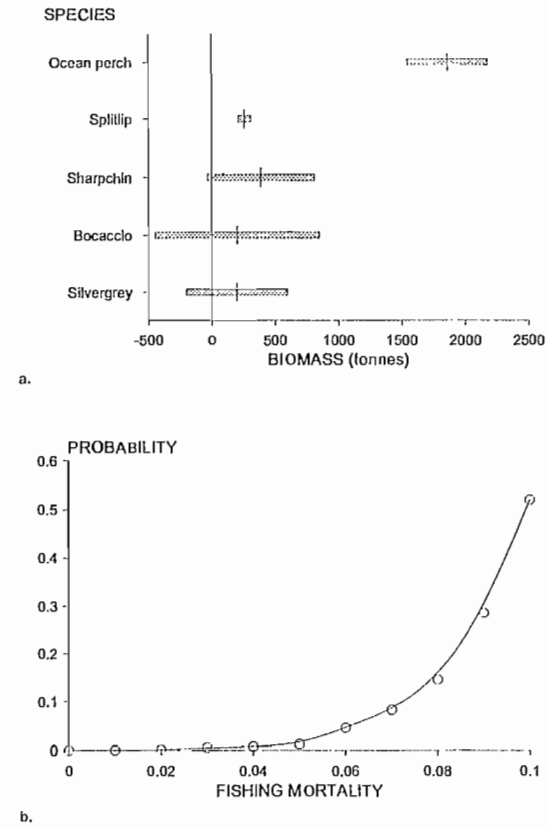


Fig. 6. Two forms of expressing uncertainty. a. Typical mean and 95% confidence intervals for rockfish biomass estimates from a trawl survey off the west coast of Vancouver Island (after Leaman et al. 1988). b. Probability of stock biomass decline as a function of fishing mortality, for a Pacific ocean perch stock in British Columbia waters (after Archibald et al. 1983).

The risk curve (Fig. 6b) presents uncertainty in a different way. It combines the distribution of estimates from a confidence interval with a utility function describing the consequences of action taken, when using values from one end of the interval to the other. It attempts to show that the distribution of risk does not mirror the distribution of uncertainty about the parameter in question, but is (in the case of Fig. 6b) highly skewed compared with the uncertainty. While much research on the estimation of these consequences is required, management will be best served if the advice from assessment biologists both conveys the uncertainty of estimation, and is expressed in the language of management actions and their consequences.

Discussion

Management of marine fisheries on the west coast of Canada, while seeing the development of several detailed and innovative programs, has not yet resulted in a comprehensive management framework for all species. Many species of invertebrates and groundfish have either short or no history

of assessment on which to base management decisions. In some instances the fisheries themselves are recent in origin or of limited importance to the total fishery yield, but there are several major fishery resources for which assessment and management are yet in their initial stages. For those fisheries having a history of management, the reference points employed have mirrored those used elsewhere in the world. The $F_{0.1}$ and $F = M$ strategies have occupied prominent places in the suite of references used, but no clearly superior choice from among this suite has emerged. Indeed, there has been variable interpretation of the meaning of even the common $F_{0.1}$, with some analyses supporting it as a conservative strategy and others concluding it represents a high-risk strategy. The primary cause for this variation concerns how the future dynamics of the stock are assumed to occur. Postulated stock-recruitment relationships that are highly compensatory support optimistic interpretations of this reference point.

The simple difference in the interpretation of a common and well-studied reference point such as $F_{0.1}$ highlights the need for management advice that reflects the uncertainty associated with the yield estimates provided. Managers and advisory groups must be able to assess this uncertainty in formulating the management plan. The initial attempts to quantify or qualify these risks have supplied only the biological component of the risk assessment, in the form of probabilities of potential stock trajectories under various fishing mortalities. Future development of this biological component must make increased use of computer intensive techniques to ensure that alternative hypotheses of stock dynamics are fully explored, and their probabilities reflected in the yield recommendations. However, it will be equally important that the analyses necessary to quantify the economic consequences of management decisions also be conducted. Such estimations carry the implicit assumption that management goals and objectives for the fishery have been developed, which may not always be true. These estimations are likely to be as uncertain as those for the biological aspects of stocks, due to the necessary assumptions about discount rates and changing opportunity costs, but managers and industry should consider the combination of biological and economic performance as an appropriate measure of long-term success.

As the changing technology of fish capture continually increases fishing power, management of fish stocks will require matching increases in sophistication. Managers will be best able to achieve this if they can assess the consequences of their decisions thoroughly. The expression of uncertainty in management advice must therefore involve a translation of the statistical uncertainty of various parameter estimates, and the uncertainty about the nature of population and fishery dynamics relationships, into an estimation of the risk associated with a given management decision. This translation represents one of the major challenges to future fisheries management.

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Reference Points for Fisheries Management: the Eastern Canadian Experience

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The implementation of various reference points in Eastern Canada since 1977 is reviewed, with particular attention to the evolution of the $F_{0.1}$ reference on Atlantic groundfish stocks. The expected benefits of a strict application of $F_{0.1}$ are discussed, in terms of catch levels, stock abundance, fishing mortality levels, and principal characteristics of the fisheries (catch rates, fish size, etc.). The actual trends in these quantities are then examined in light of the changes in stock dynamics (growth and recruitment), changes in implementation of target reference points (e.g., 50% rule), and changes in management approach (straddling stock constraints, bycatch regulations, cod-haddock-pollock management plan, multi-year plan, etc.) that have occurred throughout the 1980's. *F*-based strategies have been implemented in Eastern Canada in the context of "management by quota" and, in that context, a gradual erosion of the scientific data required for stock assessments has been observed. This is particularly true for straddling stocks and in areas with overcapacity of the fishing fleets. For certain stocks of the Scotian-Shelf, of the Grand Bank and of the northeastern Gulf of St. Lawrence, the implementation of $F_{0.1}$ has been compromised by a number of factors (e.g., pursuit of other management approaches, non-reporting of catches, etc.) and the overall effect has been that fishing mortality remained high and that no significant improvements have been detected in the total catches. However, for the most important groundfish stocks, the pursuit of $F_{0.1}$ led to a substantial reduction of the fishing mortality in comparison to the early 1970's.

Cet article examine l'adoption de divers points de référence dans l'Est du Canada depuis 1977 et accorde une attention particulière à l'évolution de la référence $F_{0.1}$ pour le poisson de fond de l'Atlantique. On y discute des avantages prévus d'une application stricte de $F_{0.1}$ en matière des niveaux de prise, de l'abondance du poisson, des niveaux de mortalité par pêche et des principales caractéristiques des pêches (taux de prise, taille du poisson, etc.). On examine ensuite les tendances réelles de ces valeurs à la lumière des modifications de la dynamique de stock (croissance et recrutement), des modifications apportées lors de la mise en application des points de référence cibles (p. ex. règle du 50 %), et des modifications des méthodes de gestion (restrictions quant aux stocks qui chevauchent la zone de 200 milles, règlements quant aux prises fortuites, plan de gestion morue-aiglefin-goberge, plan pluriannuel, etc.) qui sont apparues au cours des années 1980. Des stratégies reposant sur la mortalité par pêche (*F*) ont été mise en application dans l'Est du Canada, dans le cadre de la «gestion par contigentement». De cette situation a découlé une érosion graduelle des données scientifiques essentielles à l'évaluation des stocks. Ceci a été particulièrement noté chez les stocks qui chevauchent la zone des 200 milles et dans les zones où il y a une trop grande capacité des flotilles de pêche. Dans le cas de certains stocks de la Plate-forme néo-écossaise, du Grand Banc et du nord-est du golfe du Saint-Laurent, la mise en pratique de $F_{0.1}$ ont compromis divers facteurs (p. ex. adhésion à d'autres méthodes de gestion, non enregistrement des prises, etc.). À cause de tels facteurs, la mortalité par pêche est demeurée élevée et aucune amélioration notable du nombre total de prises n'a été notée. Toutefois, en ce qui a trait aux plus importants stocks de poisson de fond, le respect de $F_{0.1}$ a découlé en une réduction apprecéciable de la mortalité par pêche par rapport au début des années 1970.

In the 1960's and the early 1970's, management controls aimed primarily at limiting the capture of small fish by regulating the characteristics of the commercial fishing gears (e.g., mesh size regulations for trawls and gillnets). These gear limitations proved insufficient to protect the resources in the face of the massive increase in fishing pressure that occurred in the 1960's. Templeman and Gulland (1965) concluded that "... there must be some direct control of the amount of fishing". One of the key difficulty in defining controls was the necessity to develop a system that would provide a measure of nominal effort that would take into consideration the differences amongst various gear types. Templeman and

Gulland (1965) concluded that "... all methods of doing this raised difficulties" but that the method that presented the least difficulties was the use of catch quotas. They suggested that there be separate quotas for each stock, preferably allocated separately to each section of the industry.

In 1969, the Standing Committee on Research and Statistics of the International Commission for the Northwest Atlantic Fisheries (ICNAF) noted that the catches in 1968 had reached a level that was more than twice that of the mid-fifties and indicated that it was doubtful whether such a level could be sustained. The high catches of the late 1960's were attributed partly to good recruitment and partly to the removal of the stocks accumulated under previous lower levels of exploitation. In the Proceedings from the 1969 Annual Meeting, the Standing Committee noted that in view of incoming recruitment, the immediate prospects in the major North Atlantic cod and haddock stocks were "not good". The Committee indicated that there was a "... need to at least stabilize the fishing effort on the major demersal stocks over the whole Convention Area".

In 1970, ICNAF started to implement the concept of Total Allowable Catch (TAC) for certain stocks. In 1971, the Convention was amended to permit the assignment of portions of a TAC, or quotas, to individual member countries and, by 1974, fisheries on most Atlantic groundfish stocks were controlled by TAC's.

The early TAC's were set on the concept of Maximum Equilibrium Yield, i.e., the maximum yield that could be taken over time from a cohort recruiting to the stock. However, many stocks had already been fished too hard and the TAC's based on the maximum equilibrium yield did not prevent further stock declines, in part due to incomplete catch statistics, to poor adherence to the TAC's, and to the inability to enforce the established regulatory measures. The severe declines in traditional fish stocks, falling prices and rapidly escalating costs of catching and processing combined in 1974 to threaten seriously the survival of the Canadian Atlantic groundfish industry.

The recognition that the TAC's were not preventing overexploitation developed very quickly and, by 1975, it became clear that TAC's should be based on a more conservative criterion than that used until then. The discussion and negotiations that followed led ICNAF to adopt, for 1977, the concept of $F_{0.1}$. The threat of extension of jurisdiction and the likely implications of such extension for foreign fisheries were not without significance in the negotiations that resulted in the adoption of this more conservative approach to management.

Following the decision by the Government of Canada to extend fisheries jurisdiction to 200 miles on January 1, 1977, the principal objective of fisheries management was to rebuild the fish stocks so as to improve the catches and catch rates for the benefit of the Canadian fishing industry. Canada formally adopted a constant fishing mortality strategy, with $F_{0.1}$ as the target, for the stocks under its jurisdiction.

A number of reviews have been conducted since Canada extended its fisheries jurisdiction in 1977. May et al. (1980, 1981) reviewed the transition to Canadian management with a particular emphasis on the cod stocks in the first three years of Canadian management. Halliday and Pinhorn (1985) reviewed the management strategies in Canadian Atlantic marine fisheries, their rationale and the historical context in which their usage developed. A summary of management practices, catch trends and resource trends from 1960 to 1986 is also given in Pinhorn and Halliday (1990). Detailed reviews of the northern cod stock (i.e., *Gadus morhua* in NAFO Divisions 2J-3KL), which is by size the most important groundfish stock in the Northwest Atlantic, were carried out by Alverson et al. (1987) and by Harris (1990). The emphasis of the review by Alverson et al. (1987) was on the factors that led to the 1982–1987 decline in the inshore fisheries. Harris (1990) examined the scientific advice on the state of the northern cod stock and made recommendations on management measures, data collection and research.

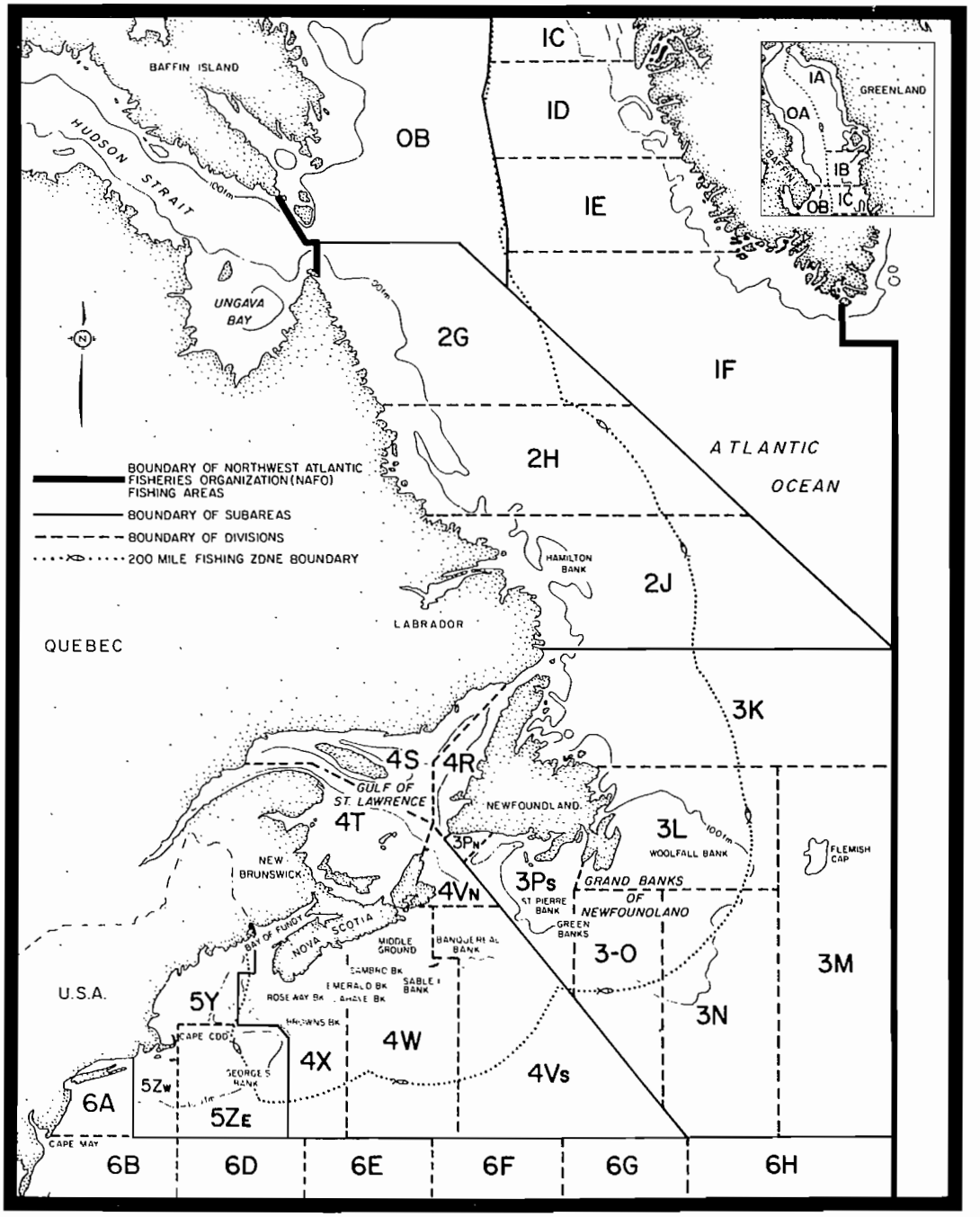
While the above-mentioned reviews had differing purposes, none aimed explicitly at reviewing the Canadian implementation of the constant fishing mortality management strategy with $F_{0.1}$ as a target. While this paper covers certain elements of previous reviews, its primary focus is to provide a comprehensive history of $F_{0.1}$ as implemented by Canada on groundfish stocks of the Northwest Atlantic (Fig. 1), including a review of the impact on catch trends and the performance of the fisheries.

Material and Methods

As no single indicator would provide a complete measure of performance for any fishing strategy, a number of indicators were used for this purpose. Recent catch trends were compared to those experienced prior to the implementation of $F_{0.1}$. The impact on commercial catch rates and on the mean age and size of fish caught was also examined. Changes in fish stocks were examined in terms of changes in recruitment, stock abundance, growth and mortality due to fishing.

Catch statistics were obtained from the annual statistics of the Northwest Atlantic Fisheries Organization (NAFO). Data for 1989 and 1990 are still preliminary. For certain straddling stocks, additional information on unreported catches in the NAFO Regulatory Area was obtained from the Report of the Scientific Council of NAFO (Anon., 1991). Typically, estimates for unreported catches originate from fisheries surveillance operations.

Because of the limitations in the data, detailed analytical assessments are available only for a limited number of groundfish stocks. In 1991, analytical assessments were available for eight stocks, namely cod off southern Labrador and on northern Grand Bank (2J-3KL), cod on St. Pierre Bank (3Ps), the two cod stocks in the Gulf of St. Lawrence (3Pn-4RS and 4T-Vn:January-April), cod on Browns Bank (4X), cod and haddock *(Melanogrammus aeglefinus)* on Georges Bank (5Zj,m), as well as pollock *(Pollachius virens)* on the Scotian



Subareas and Divisions of the NAFO Convention Area and limits of the Canadian fishing zone (East coast).

Fig. 1. Map of the east coast of Canada showing the stock-areas used by the Northwest Atlantic Fisheries Organization (NAFO).

Shelf and Georges Bank (4VWX+5Zc). The catch from these stocks summed up to about 60% of the groundfish landings from traditional fishing grounds in 1990. The stocks included in this review are those for which the analytical assessments, i.e., assessments based on an age-structured population dynamics model, were considered sufficiently precise to serve as the basis for the calculation of reference catch levels. Analytical assessments have been performed on a number of other stocks but, due to uncertainties in the data or to the poor precision of the estimates, their results are considered insufficient for the formulation of advice on TAC's. For the eight stocks listed above, the historical trends in biomass, recruitment, fishing mortality, and growth are presented, together with trends in catch characteristics (mean age of fish in catch, mean weight in catch and catch rates). Analyses for individual stocks cover the time period for which data is considered to be reliable. In order to provide a basis for the evaluation of the performance of the Canadian implementation of the

$F_{0.1}$ management strategy since extension of jurisdiction, a composite (mean or total) graph of recent trends has also been developed for the 1974–1990 period using all stocks for which an analytical assessment is available.

Detailed information on individual stocks can be found in annual assessments for each stock. In particular, the reader is referred to Baird et al. (1991) for cod in 2J-3KL, to Bishop et al. (1991) for cod in 3Ps, to Fréchet et al. (1991) for cod in 3Pn-4RS, to Hanson et al. (1991) for cod in 4T-Vn(Jan.–Apr.), to Campana and Hamel (1991) for cod in 4X, to Hunt et al. (1991) for cod in 5Zjm, to Gavaris and Van Eeckhaute (1991) for haddock in 5Zjm and to Annand and Beanlands (1991) for pollock in 4VWX+5Zc.

$F_{0.1}$: A Reference Point for Fisheries Management

In general terms, $F_{0.1}$ corresponds to a fishing mortality beyond which increases in yield-per-recruit relative to increases in fishing effort are marginal. In technical terms, $F_{0.1}$ is defined as the rate of fishing mortality for which the increase in yield resulting from a small increase in fishing mortality is one-tenth the increase that would have resulted if the same small increase in fishing mortality has been applied to the unexploited stock (Gulland, 1968; Gulland and Boerema, 1973). In mathematical terms, $F_{0.1}$ becomes simply the value of the fishing mortality, say F, at which the slope (dY/dF) of the yield-per-recruit curve (Ricker, 1975) is 10% of the slope at origin. One of the advantage of $F_{0.1}$, from the mathematical point of view, is that it can be calculated even in the cases, as in some flatfish stocks, where F_{max} is undefined or poorly defined (ICNAF, 1976). $F_{0.1}$ is also less sensitive than F_{max} to small changes in growth or partial selection patterns, particularly when the yield-per-recruit curve has a maximum that is poorly defined.

Expected Characteristics of Fish Stocks Managed at $F_{0.1}$

May (1981) suggests that the adoption of $F_{0.1}$ or similar conservation regime has a number of advantages over strategies that are based upon higher fishing mortality rates. The implications of fishing at $F_{0.1}$ versus any higher fishing mortality can be summarized as follows:

- larger total biomass allowing higher catch rates;
- larger fish in the catches, allowing higher value products, and lower processing costs;
- more stable catches and stocks from year to year, as more year classes contribute to the exploited stocks and catches;
- more year classes in spawning biomass to allow for chances of better recruitment;
- less effort (about one third to one half) than at the effort maximizing yield;
- reducing the risk of substantial over-exploitation which has more serious implications than equivalent under-exploitation;
- lower total yield (about 10% lower in the long term);
- higher "safety margin", in the sense that the higher stock levels associated with low fishing mortalities provide longer lead times to identify the effects of uncertainties in stock assessments and to take corrective measures.

For example, the effect of reducing fishing mortality from F_{max} (i.e., the level of fishing mortality giving maximum equilibrium yield per recruit) to $F_{0.1}$ on yield, catch rates, mean age of fish in the catch and mean size (or weight) of fish in the catch is illustrated in Fig. 2. While $F_{0.1}$ presents some desirable features in terms of fisheries performance or stock characteristics and offered in the late 1970's a practical solution for moving towards a more conservative management regime, it must only be seen as a step in the right direction rather than the "optimal" solution in any particular fishery. Gulland (1987) notes that $F_{0.1}$, though better on economic grounds than F_{max}, "...could be a long way from the economically optimum value of F". Gulland (1987) suggests also that better ways of dealing with uncertainty are needed. Lane (1989) argues that long term TAC strategies should make explicit the economic underpinnings of the levels of fishing mortality implied and that the stock-recruitment relationships should be taken into consideration in order to ensure the long term viability of the fisheries. Recent applications of risk analysis techniques (Hoenig et al., 1990; Mohn, 1991) aim, in essence, at addressing some of these shortcomings by taking into consideration the variability of recruitment and the uncertainties in assessment results. Similarly, some progress has been made in the evaluation of the economic implications of management strategies. For instance, the implications of $F_{0.1}$ and alternative fishing mortalities on the financial performance of a fishing operation have been described for one redfish fishery of the Northwest Atlantic by Huson et al. (1984).

Implementation of the $F_{0.1}$ Approach on Groundfish Stocks of the Northwest Atlantic

When evaluating a management strategy and associated targets, it is important to have an understanding of the general context under which it is implemented. An overview of the regulatory regime for groundfish stocks of the Northwest Atlantic is thus provided below, together with a brief description of the infrastructure supporting the implementation.

Learning from the years of overexploitation which resulted in a severe depletion of the Atlantic groundfish stocks, Canada implemented after the extension of fisheries jurisdiction to 200 miles in 1977 a constant fishing mortality strategy based on $F_{0.1}$, as initiated under ICNAF. In order to better control fishing activities, the Canadian fisheries surveillance programs and the catch monitoring programs were increased

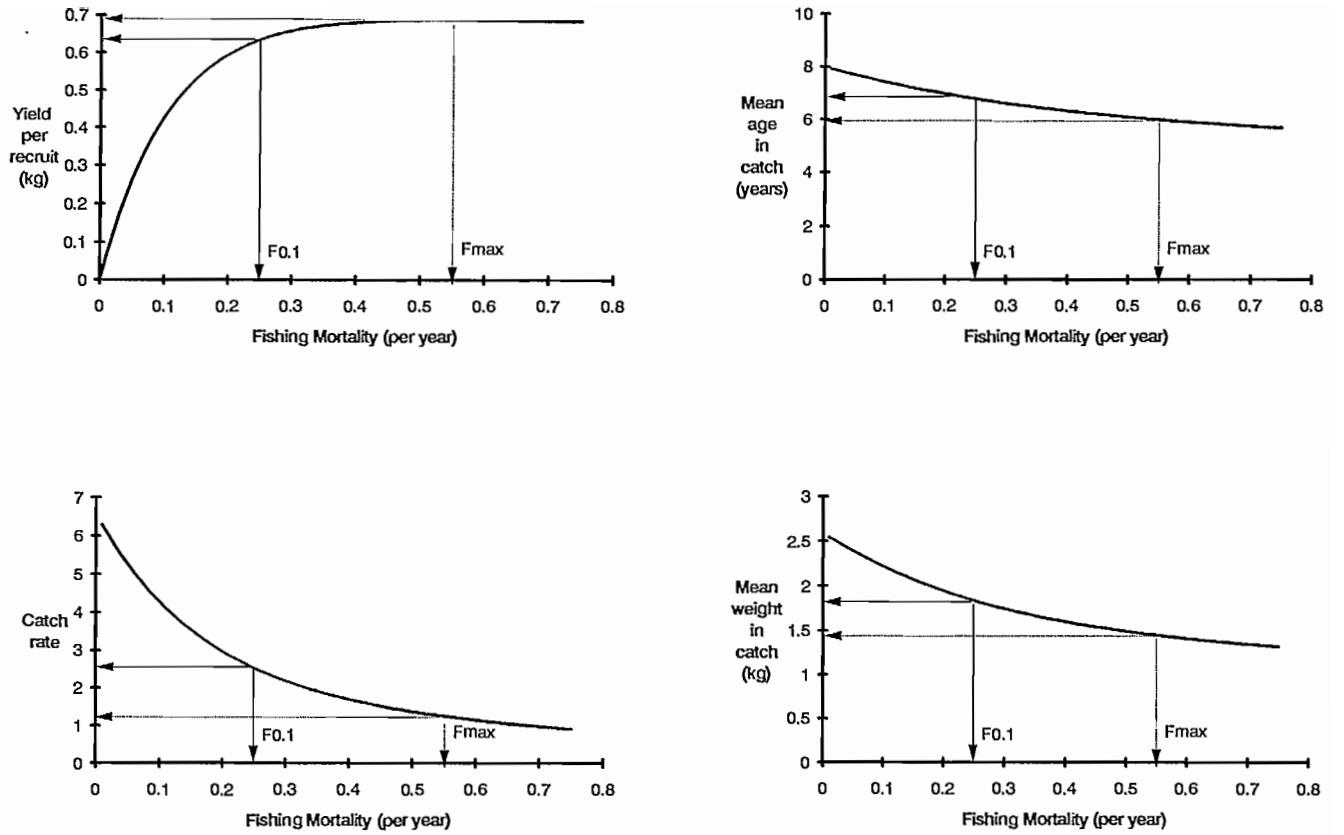


Fig. 2. Impact of fishing at $F_{0.1}$ and F_{max} on yield-per-recruit, on catch rates, as well as on mean age and mean weight of fish in the catch for a theoretical cod stock.

to reflect the extended fishing zone. Stiffer penalties were imposed for vessels found in violation of the Fisheries Act. An observer program was established to monitor the operations of foreign vessels fishing within the Canadian zone. The observer program was expanded thereafter to cover a significant proportion of the domestic fleet of larger offshore trawlers. Similarly, research and catch sampling programs were augmented so as to base management decisions on sound scientific information.

Canadian Regulatory Measures

The Groundfish Management Plan aims at conserving and restoring the fisheries resources by regulating access to all groundfish stocks by seasons, quotas, vessel size and gear types. Allocation is on the basis of equity taking into account the proximity of coastal communities and fleet sectors and their dependence on the resource as well as the economic efficiency and mobility of the different fleets. The target fishing mortality was set at $F_{0.1}$ to allow higher catch rates and larger fish size, to reduce fluctuations in catches and to provide a safety margin against overexploitation. Because the target fishing mortality was set lower than the one experienced in the mid-1970's, the strategy was also promoting stock rebuilding.

It is important to note that the limitation of the catches by annual quotas is not the sole management measure in operation on groundfish stocks. Other management measures such as mesh size restrictions to avoid catching small fish, seasons to prolong fishing and protect spawning areas, as well as boat and trip limits or quotas are part of the management system. These measures have not been consistently followed by fishermen and the usage of chokers and liners, for example, have often negated the objective of mesh size limitations. This has resulted in large amounts of small fish being caught and often discarded.

Canada has continued to use a number of controls after extension of fisheries jurisdiction in 1977:

- Area, season and gear restrictions. Many of these restrictions were in existence prior to extended jurisdiction and continued to be applied after extension. These restrictions are too numerous and complex to describe here. They may aim simply at providing a framework for an orderly prosecution of fishing, at providing a minimum protection for certain stock components (e.g., by reducing the catch of small fish or by protecting the spawning components of fish stocks), or at achieving an aggregate maximal yield-per-recruit across a number of species and stocks. It is worth noting that the trawl mesh size was modified in 1982 by removing differentials for different material and specifying 130 mm

irrespective of material. This resulted in a small (about 10%) overall increase in average mesh size. Also, a significant change in the trawl mesh regulation took place in 1991 for groundfish fisheries of the Scotian Shelf and Georges Bank; this change was driven by the industry's desire to protect small fish (Haché, 1989).

- Restrictions on catches. Total Allowable Catches (TAC's) were introduced on all stocks specified in the annual Groundfish Management Plan. Initially, many fixed gear vessels were under "allowances" rather than quotas. An allowance is a portion of the TAC reserved for a given fishing sector based on its historical levels of activity (catches are allowed to exceed the allowances but, in general, allowances have been set at a sufficiently high level to account for the activity of that fleet sector). However, in the early 1980's, most gear sectors were put under quota control, with the exception of the inshore fisheries exploiting 2J-3KL cod in Newfoundland which are still operating under global allowance. Typically, TAC's and fleet quotas are established for the inshore (vessels less than 65 feet), mid-shore (vessels from 65 feet to 100 feet), and offshore (vessels greater than 100 feet) fleets for more than 42 groundfish stocks. An additional restriction in recent years has been the introduction of a minimum fish size regulation. The regulation specifies a minimum fish size of 17 inches for cod, haddock and pollock. Changes in other regulations, such as bycatch and closed area/season have been minor (with the exception of some stocks on the Scotian Shelf, as detailed below),

- Direct controls to limit fishing effort, such as limitations in the number of fishing vessels or restrictions on vessel capacity were continued. The limited-entry licensing which was initiated in 1973 for offshore groundfish vessels was continued and, by the spring of 1980, all fishermen and inshore groundfish vessels were under limited entry. In essence, virtually all commercial fisheries on Canada's Atlantic Coast are now subject to limited entry.

- Enterprise allocations for the large offshore groundfish vessels began in 1982 with four major companies. In 1984, the program was expanded to include the participation of a group of 16 smaller trawler companies. Enterprise allocations have allowed the offshore companies to rationalize the operation of their fleets (by reducing the race for catching fish between the companies) and by allowing companies to better match market demands. In addition, a number of initiatives were undertaken throughout the zone in recent years to introduce boat quotas.

It is against that regulatory regime and under that elaborate infrastructure (surveillance and monitoring, research and sampling, etc.) that $F_{0.1}$ was implemented.

The extension of fisheries jurisdiction to 200 miles in 1977 removed a considerable amount of foreign fishing effort. However, domestic fishing capacity increased rapidly and, by 1991, overcapacity has become a major problem in many fisheries resulting in misreporting and overfishing (e.g., Haché, 1989). These problems are particularly serious in southwest Nova Scotia and in the northeastern Gulf of St. Lawrence. Attempts to control effort in southwest Nova Scotia included the imposition of catch and effort limitations aimed at restricting the activities of the fleets, e.g., quarterly limits, weekly quotas and, more recently, trip quotas. This led to a rather complex management plan. Fleet overcapacity has put considerable strain on the regulatory system and seem to have led to large-scale misreporting by species and area, and discarding. In turn, this reduced the ability to do stock assessments on certain stocks, in particular for haddock. While licensing and vessel replacement policies have been put forward early after extension of jurisdiction, they have not controlled fleet capacity.

The Canadian Implementation of the $F_{0.1}$ Strategy

More than 35 groundfish stocks are assessed each year by the Canadian Atlantic Fisheries Scientific Advisory Committee (CAFSAC). In addition, CAFSAC provide advice for the management of a number of shellfish species (e.g., shrimp, snow crab, lobster), pelagic fish (herring, mackerel, capelin), anadromous and catadromous species, marine mammals, etc. The $F_{0.1}$ approach is applicable only on species or stocks for which scientific information is sufficiently detailed to allow the application of age-structured analytical models and for which forward catch projections are possible. The philosophy of $F_{0.1}$ is pursued by the implementation of low exploitation rates in other stocks or species for which there is insufficient biological information or for which the application of a detailed age-structured analysis is not possible. For instance, the catch at 2/3 F_{msy} is often used as a proxy to $F_{0.1}$ for redfish *(Sebastes sp.)* stocks. In essence, $F_{0.1}$ has been found to be relevant for gadoid stocks, e.g., cod, haddock, pollock and silver hake *(Merluccius bilinearis)* and for certain flatfish stocks, e.g., American plaice *(Hippoglossoides platessoides)*, yellowtail flounder *(Limanda ferruginea)*. $F_{0.1}$ has also been applied to certain pelagic stocks, e.g., herring *(Clupea harengus)*, mackerel *(Scomber scombrus)*, and to certain scallop stocks.

A strategy aiming at a constant exploitation rate such as $F_{0.1}$ minimizes the variations in annual fishing effort. However, because the strategy aims for catching a constant fraction of a variable number, catches are likely to fluctuate as stock abundance can vary from year to year due to natural causes. In order to avoid changes in TAC's that could simply result from variability in the data, practical considerations dictated that TAC's be changed only when there was a difference of more than either 10% or 10,000 t between the reference catch level estimated for the upcoming year and the last TAC.

In the mid 1980's, it became obvious that a strict application of the $F_{0.1}$ reference catches could lead to drastic changes

in TAC levels between successive years. In order to minimize the impact on fishermen of drastic reductions in TAC's between consecutive years, a rule was established by which the move to $F_{0.1}$ would be phased in over time. The recognition that the industry needs some lead time to adjust to the new reality when stocks are declining and the fact that annual adjustments in stock assessments could in part be related to intrinsic variability in annual data were key considerations in the adoption of the new rule. This rule, often referred to as "the 50% rule", stipulates that the fishing mortality in the coming year of the plan would be set at a value half way between the current fishing mortality and $F_{0.1}$ according to:

$$F_{\text{next}} = F_{\text{current}} - \left(\frac{F_{\text{current}} - F_{0.1}}{2} \right)$$

where the F_{next} is the fishing mortality for the coming year of the plan and F_{current} is the fishing mortality anticipated for the current year. In addition, if F_{next} is higher than twice $F_{0.1}$, than F_{next} is set to twice $F_{0.1}$ to prevent growth overfishing. The "twice $F_{0.1}$ level" was proposed as an approximation to $F_{\max}$ due to the poor reliability of estimates of the latter.

The 50% rule thus corresponds to a compromise between the status quo, in terms of fishing mortality, and the $F_{0.1}$ reference. The 50% rule is invoked only when the calculated catches at $F_{0.1}$ are declining; it is not invoked to dampen the increase when the $F_{0.1}$ catch estimates would give an increasing TAC. The rule may be triggered because the stock is declining, or simply because TAC's based on $F_{0.1}$ have been exceeded due to misreporting, discarding, or because of difficulties in achieving reliable catch monitoring and adequate control in the fisheries. The rule may also be triggered simply because of inherent uncertainties in calculated fishing mortality levels. As these elements are frequently in operation in many stocks (e.g., discarding, misreporting, imperfect monitoring and control, poor F estimates, etc.), the 50% rule has been frequently invoked and $F_{0.1}$ became, gradually, a "long term strategic goal" rather than an immediate (or annual) tactical objective. The end result is that the reference fishing mortalities used for the calculation of TAC's have, by design, been higher than $F_{0.1}$.

The Groundfish Management Plan also stipulates that "If the stock assessment provides evidence of levels of spawning stock biomass likely to endanger recruitment, fishing effort (and thus fishing mortality) in the coming year will be reduced to allow immediate growth in spawning stock biomass." This measure has been invoked, for instance, for one haddock stock. There are also rules governing the transfer of fish among gear sectors: this, and in particular the transfer of fish from longliners to otter trawlers, has become a controversial issue in some years.

Multi-year Management Plans

In the late 1980's, there was a renewed appreciation that the uncertainties in stock abundance estimates were such that annual adjustments to TAC's could be more a reflection of variations in the data rather than a real change in stock status. In the face of the ever present overcapacity in certain areas, there was also a growing need to contrast capacity and production by providing an indication of stock productivity over a longer time horizon. These observations pointed at the need 1) for alternatives to the $F_{0.1}$ reference in years of transition, and 2) for a longer planning horizon.

The need for alternative reference points in years of transition was recognized explicitly by CAFSAC in 1990 when reference points other than $F_{0.1}$ and the 50% rule were introduced as part of the advisory process. Despite long discussions, no single set of principles were designed for making recommendations on alternatives. Instead, CAFSAC adopted a flexible approach in which options were put forward in light of stock trends and expected productivity. Typically, the principal reference points currently used are:

- $F_{0.1}$ or the 50% rule;
- surplus production (which is equivalent to status quo in terms of stock biomass);
- status quo in terms of catch levels;

In essence, the approach has been to offer options for the management of these stocks based on the principle that the fishing mortality could be allowed to "float" with the aim of minimizing the impact of stock declines on the the TAC's. This approach was developed as an interim solution only, as the introduction of multiple reference points raises a number of questions with respect to the long term (and short term) management objectives and complicates decision making in the absence of an absolute yardstick. In that context, it has also been recognized that there is a need to account for uncertainties in assessments and in stock dynamics (e.g., through risk analyses), as well as to discuss strategic and tactical decisions in light of management objectives and priorities (which may need to be restated or made more explicit).

The recognition of the need for a longer planning horizon led to the implementation of a multi-year management plan for the 1991–1993 period. An annual plan remains in effect for straddling stocks and for stocks which are too dynamic to allow long term projections, such as certain stocks in the south of the Canadian zone. While the status of fish stocks and the performance of the fisheries continue to be monitored on a yearly basis, changes are brought to the multi-year plan only when significant departures from expected results can be demonstrated (such changes are generally published in annual updates of the multi-year plan).

Transboundary Stocks

Canada shares jurisdiction over certain fisheries resources with U.S.A. on Georges Bank, with Greenland (Denmark) in the Davis Strait, with France around the Islands of St. Pierre et Miquelon and with NAFO on the Nose and Tail of the Grand Banks. The boundary between the U.S.A. and Canada in the

Gulf of Maine was determined in 1984 by decision of an International Court of Justice. There is no fisheries agreement between Canada and U.S.A. on Georges Bank fisheries resources at present; both countries have resorted to address management issues on straddling stocks through unilateral actions. The boundary around the Islands of St. Pierre et Miquelon was referred to an Arbitration tribunal in March 1989 and a binding arbitration decision was issued on June 10, 1992. The Northwest Atlantic Fisheries Organization (NAFO) manages the fish stocks beyond the 200-mile boundary on the Tail and Nose of the Grand Banks and on Flemish Cap and sets Total Allowable Catches (TAC's) that apply inside and outside the 200-mile boundary for these stocks. In recent years, fishing activities on the Grand Banks in the NAFO Regulatory Area have been such that catches exceeded significantly the recommended TAC levels.

While many straddling stocks have been fished within Canadian waters in accordance with the $F_{0.1}$ principle, the actual fishing pressure exerted on them outside the 200-mile zone has often been significantly higher, either because of inconsistent approaches to conservation or because of a lack of control on fishing activities.

Catch Trends: 1960–1990

When the $F_{0.1}$ reference was introduced, groundfish catches in the Northwest Atlantic were rapidly declining and the expectations were that a strategy based on a lower fishing mortality would lead to a stabilization and, eventually, to a certain recovery of catches. We outline below the catch trends that were realized after the introduction of $F_{0.1}$ on each of the principal groundfish groups.

Groundfish: Overall (Fig. 3)

Total landings of groundfish (including stocks outside the Canadian 200-mile boundary on Flemish Cap and straddling stocks on Georges Bank) rose from 819,000 t in 1977 to 1,180,000 t in 1986, and then declined progressively to 952,000 t in 1990 (Fig. 3, upper panel). From less than 400,000 t in the mid-1970's, Canadian landings rose quickly and reached 700,000 t in the early 1980's. Canadian catches fluctuated between 700,000–780,000 t until 1989, when they declined to 656,000 t. Preliminary information from 1990 suggests a further decline to 603,000 t.

In 1989, the product value of the Canadian Atlantic Coast groundfish fisheries was $992 million, corresponding to 48% of the value of the Atlantic fishery production for that year. However, as indicated earlier, catches are not necessarily an indication of the economic health of the industry. The Task Force on Atlantic Fisheries (Kirby, 1982) found that, in 1981, fish processing enterprises lost a total of $57 million, based upon sales of $540 million. An income survey also indicated that the net income from fishing in 1981 for full time fishermen was relatively low (i.e., of the order of $6,500 for 50% of the fishermen in comparison to an average for all occupations

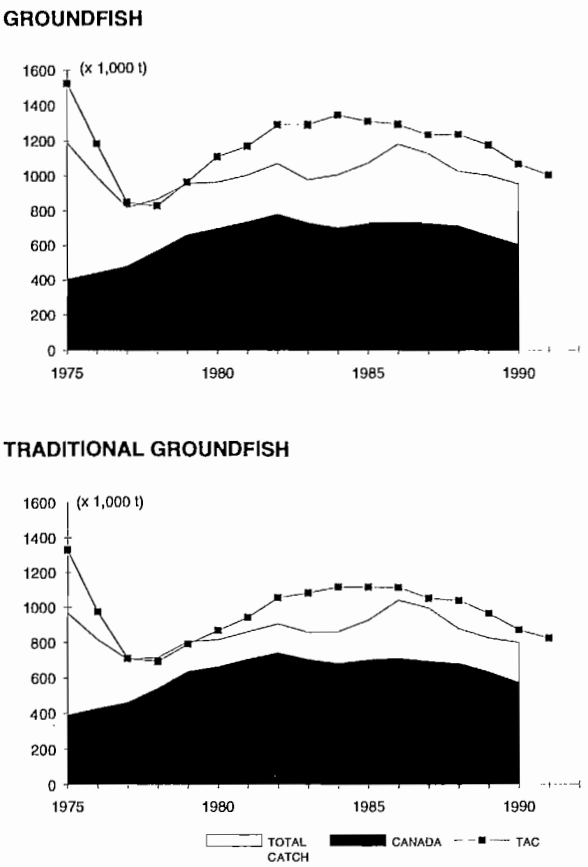


Fig. 3. Total catches, Total Allowable Catches (TAC's) and catches by Canada for groundfish stocks in the Northwest Atlantic (thousand tonnes): all stocks (top panel), traditional groundfish (lower panel).

of about $12,000 for the Atlantic regions). The restructuring of the industry that took place in the 1980's was aimed at improving its economic viability and some improvements appear to have resulted. For instance, Haché (1989) reports for Scotia-Fundy that, by 1985, the "average income for all taxfilers was not quite double its 1976 level, but the average for fishermen taxfilers had almost tripled".

Groundfish: Traditional Species and Fishing Areas (Fig. 3)

Catches for the traditional groundfish (cod, redfish, haddock, pollock, American plaice, witch, Greenland halibut, yellowtail flounder and winter flounder) in Subareas 2, 3 and 4 increased from 705,000 t in 1977, when the fisheries jurisdiction was extended, to 1,038,000 t in 1986 and then declined progressively to 800,000 t in 1990. The Canadian share of the catch increased from 30–40% in the early 1970's to 70–80% in the 1980's. The Canadian catch varied between 575,000 t and 740,000 t since 1980, compared to about 400,000 t in the mid-1970's.

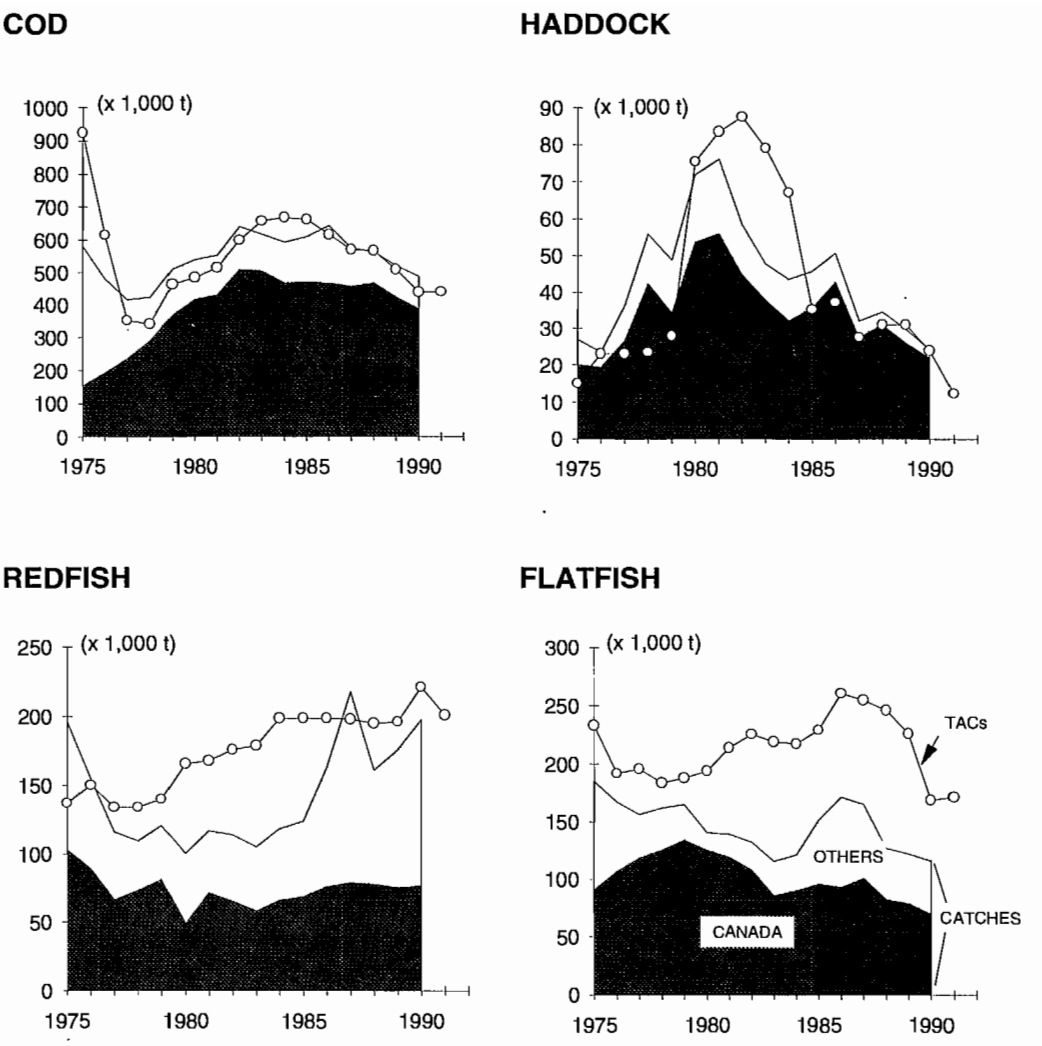


Fig. 4. Total catches, Total Allowable Catches (TAC's) and catches by Canada of groundfish by species category, in the Northwest Atlantic: (thousand tonnes).

Cod (Fig. 4)

The fishery for cod is the largest groundfish fishery in the Northwest Atlantic. There are twelve stocks within Canadian waters, spread along the coast from northern Labrador to Georges Bank. Total catches on cod stocks increased quickly after extension of fisheries jurisdiction, from 417,000 t in 1977 to 644,000 t in 1986, and declined progressively to reach 488,000 t in 1990 (preliminary). The Canadian cod fisheries benefited greatly from the extension of fisheries jurisdiction: the Canadian catch on these stocks increased from about 155,000 t in the mid-1970's to 420,000–510,000 t throughout the 1980's. Canadian catches of cod were 388,000 t in 1990, i.e., more than twice those of the mid-1970's. The Canadian share on traditional fishing grounds increased from 20–25% in the early 1970's to more than 80–85% in the 1980's.

Haddock (Fig. 4)

While the total landings of haddock have been of the order of 25,000–30,000 t in recent years, i.e., much less than the landings of redfish and flatfish, the landed value of haddock landings is comparable to that of redfish or flatfish stocks because of the relatively high unit price. Although landings rose quickly after the extension of fisheries jurisdiction, they declined thereafter to a level comparable to that of the mid-1970's. Because of the high price they fetch on the market and because of the overcapacity of the fishing fleet, the management of haddock stocks on the Scotian Shelf has proven to be difficult. While recent scientific recommendations have been for no directed fishery in that area and for a bycatch at the lowest possible level, a strict application of scientific recommendations has not been possible. Large scale misreporting

has been recognized (Haché, 1989) and the quality of catch data is considered to be so poor that reliable stock assessments are not possible. In sum, the catch trends depicted in Fig. 4 for haddock stocks are unlikely to reflect actual catches because of the quality of catch reporting. Also, in the context of this review, it is important to note that the fishing mortalities on haddock stocks on the Scotian Shelf have remained much above $F_{0.1}$ (and probably above F_{max}) throughout the 1980's.

Redfish (Fig. 4)

There are seven stocks of redfish distributed from the southern coast of Baffin Island to the Scotian Shelf. The commercial fishery for redfish usually occurs along the continental slope, in 100–700 m, and in deep gullies. While the TAC's have increased progressively since the extension of fisheries jurisdiction, the Canadian catches have remained somewhat stable since the mid-1970's. The catches by other nations fishing in the northwest Atlantic have increased in recent years, mostly as a result of increase fishing activities on the Nose and Tail of the Grand Banks and on Flemish Cap. For instance, catch estimates in NAFO Div. 3LN for redfish increased fourfold, from 20,000 t in 1980–1985 to 78,000 t in 1987.

Flatfish (Fig. 4)

The most important flatfish species fished off Canada's east coast are American plaice (5 stocks), witch (4 stocks), yellowtail flounder (1 stock), Greenland halibut (3 management units). Flatfish species on the Scotian Shelf are pooled in a single management unit and treated as a single stock. Canadian catches have varied around their long term average of 110,000 t since extended fisheries jurisdiction, with a noticeable decline after 1987. This decline can be related to reductions in the TAC's for American plaice and yellowtail flounder on the Grand Banks (transboundary stocks). The shortfall of catches relative to TAC's are principally due to shortfalls in the Greenland halibut fisheries in northern areas (Davis Strait and Labrador).

Catch Trends by Region : 1960–1990 (Fig. 5)

Newfoundland and Labrador. Catches of groundfish in this area include catches of the transboundary stocks on the Nose and Tail of the Grand Banks, around the Islands of St. Pierre and Miquelon, and the stocks that are completely outside the Canadian fisheries zone on Flemish Cap. Canadian catches have increased from 140,000 t in the mid-1970's to 300,000–375,000 t throughout the 1980's. In comparison to the early 1960's, the Canadian catch corresponds to a much larger proportion of the total catch in these areas. Total catches have also increased in the early 1980's but have declined noticeably in the late 1980's. Overall, the total catches remained, throughout the 1980's, much below the levels attained in the late 1960's.

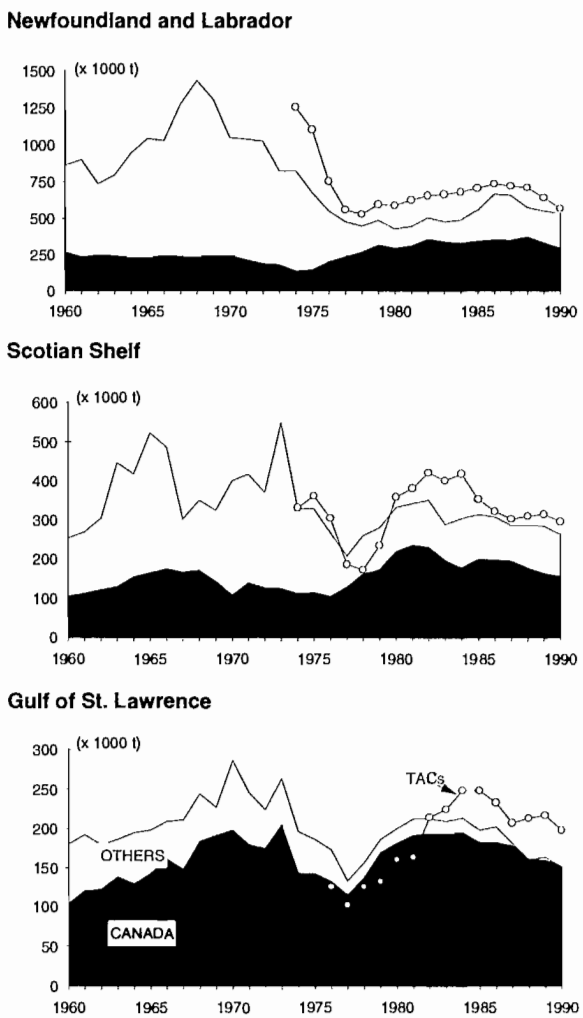


Fig. 5. Total catches, Total Allowable Catches (TAC's) and catches by Canada of groundfish by Region: Newfoundland and Labrador (top panel), Scotian Shelf (central panel) and Gulf of St. Lawrence (lower panel).

Scotian-Shelf-Bay of Fundy-Georges Bank. Catches of groundfish in this area include catches of the transboundary stocks in the Georges Bank/Gulf of Maine area. There is also a large fishery on silver hake by U.S.S.R. and, in essence, the peaks in the catches (Fig. 5) of other countries in the early 1960's and early 1970's were largely due to sudden expansions of the USSR Scotian Shelf fishery for silver hake. From annual catches of 300,000 t in 1973, the catches of silver hake were reduced to 96,000–116,000 t in 1974–76, and to 37,000 t after extension of fisheries jurisdiction in 1977. The changes in the catches of silver hake explain much of the decline in the total catches in that area between 1973 and 1977. Total catches increased quickly after extension the fisheries jurisdiction, mainly as a result of increased catches by Canada on traditional species, which have increased from 110,000 t in the mid-1970's to 150,000–215,000 t throughout the 1980's.

Gulf of St. Lawrence. The principal fisheries contributing to the landings in that area are cod and redfish. There is at present little fishing by foreign countries on these stocks. The catches increased quickly after 1977, from 114,000 t in 1977 to 195,000 t in 1984, but declined progressively thereafter to 151,000 in 1990.

Profile of Fish Stocks

The historical trends in recruitment, growth, fishing mortality and biomass provide an indication of the response of the stocks to the implementation of $F_{0.1}$ in the northwest Atlantic. These trends were available only for eight of the groundfish stocks, as indicated above (Material and Methods Section). In essence, these eight stocks accounted for about 60% of the catch of traditional species in recent years. Detailed information on stock characteristics is not available for many of the transboundary stocks of the Grand Banks (e.g., cod in 3NO, American plaice and yellowtail flounder in 3LNO) and for haddock on the Scotian Shelf. In any case, the haddock stocks on the Scotian Shelf have been subjected to a number of management measures (bycatch, area closures, Cod-Haddock-Pollock combined quotas, trip limits, etc.) which are not related to the implementation of $F_{0.1}$.

It is important to explore what events, fishery-related, environmental or biological (stock related), may have contributed to the trends in stock abundance. Following the extension of fisheries jurisdiction, initial expectations of the productivity of fish stocks were based on the productivity of Atlantic groundfish stocks experienced until then. There are two components to the productivity of a fish stock: recruitment (i.e., the production of young fish) and growth (i.e., accrued weight of individual fish). Were there changes in the productivity of fish stocks in recent years, and to what degree did they influence our results? From the management standpoint, other questions are equally important. Have fishing mortalities been reduced to $F_{0.1}$? Did the stock biomass increase as expected? Did $F_{0.1}$, as implemented, contribute to restoring the groundfish resource and did it improve the economic viability of the fisheries? While someone would not expect to find a perfect agreement with the original objectives, negative answers to these questions would give rise to concern. It is in that context that the analysis of "profile" of these eight groundfish stocks has been undertaken.

Information on the recent trends in recruitment, growth, fishing mortality and biomass for the eight stocks studied are provided for each stock in the Appendix and an overall (or average) measure of the response of these stocks is given in Fig. 6.

Recruitment

While there may be a correspondence in the occurrence of strong or weak year classes for some stocks (e.g., 2J-3KL and 3Ps cod, Fig. A1), recruitment does not seem to have followed, in general, similar trajectories for all stocks. Recruitment estimates for cod in 2J-3KL suggest that recruitment was much higher in the 1960's than in the 1970's and 1980's. Additionally, the 1976 and 1977 year classes, i.e., immediately prior to the extension of fisheries jurisdiction, and the 1983 and 1984 year classes were amongst the lowest observed on that stock. For cod in the southwestern Gulf of St. Lawrence (4TVn), contrary to northern cod, recruitment in the 1960's and early 1970's was much weaker than that of the late 1970's and 1980's. For both stocks, periods of high recruitment seem to correspond to periods of high stock abundance.

The tendency for many stocks (cod in 2J-3KL, 3Ps, 3Pn-4RS, 4TVn, 4X) to have better than average recruitment for the 1986 and 1987 year classes should also be noted. The overall recruitment trend across all stocks (Fig. 6) suggests that these year classes (i.e., 1986 and 1987) are amongst the highest seen in the last 20 years. These are the year classes that will support the fisheries in the years ahead.

Fish Growth

The average weight of a 7-year-old fish in the catch has decreased on the average since extension of fisheries jurisdiction (Fig. 6). This was particularly felt in the cod stock in southwestern Gulf of St. Lawrence, where a 7-year-old fish, which weighted approximately 2.3 kg in the mid-1970's, weighted only one kilogram in the late 1980's. While age seven has been chosen here because it is fully recruited to the fisheries for all stocks (see Fig. A2; age 5 was chosen for Georges Bank cod because of the fast growth rate in that area). However, similar trends in average weight at age can be found for other fully recruited ages. These observations led scientists to investigate the causes for the changes in mean weight at age observed in the catch. Such changes could be due to seasonal changes in the fisheries, to changes in the proportion of catch by the various gears, or to changes in somatic growth. These investigations led to the conclusion that the observed changes in mean weight at age in the fishery were real and were not an artifact of variation in the timing of the fishery or of changes in the proportion of catch by the various gears from year to year. The possible causes for changes in growth were also investigated, particularly in relation to the changes in stock abundance since extension of the fisheries jurisdiction. In particular, the evidence for density-dependent growth on the cod stocks of the Northwest Atlantic was investigated using not only fishery-related data, but also research survey data. Consistent with the hypothesis of density dependence, fish size was found to be negatively correlated with population abundance in five of the eight cod stocks investigated, and the effect was particularly strong for the southern Gulf of St. Lawrence cod stock, for northern cod (2J-3KL), and for cod on the northern section of the Scotian-Shelf. The effect was moderate for cod in the northeastern Gulf of St. Lawrence (short time series) and present, but weak, in the St. Pierre Bank stock (3Ps). It was noted that the stocks which did not present significant correlations with abundance were in the southern end of the range: these are stocks where growth was

the fastest and density was low, indicating that food may not be a limiting factor.

An attempt was also made to identify other variables that could have contributed to the declining trends in growth (e.g., temperature, availability of prey species or food). It was concluded that where the data were available, environmental conditions explained little, if any, of the variation in cod growth. Stock biomass and population abundance remained the best explanatory variable for these stocks where the effect has been demonstrated. However, data gathered in 1991–1993, i.e., after these investigations, suggest that changes in growth cannot be solely attributed to density dependent mechanisms because fish weight did not increase when stocks declined further. Other factors, such as the availability of food or environmental conditions, must also have had an influence. The mechanisms controlling growth are complex and deserve further studies. Attempts to provide a mechanistic approach to density dependence also proved unsuccessful. Two hypotheses have been put forward to relate the effect of density on growth: 1) the continuous competition hypothesis where growth is affected by competition throughout the lifetime of a cohort and 2) a bottleneck effect where the growth history is determined early on by the initial density of the cohort. It was recognized that the inclusion of density dependence of the first type will have serious impact on the current management of the fisheries and the reference fishing mortality (Anon., 1990B).

Fishing Mortality

On average across the eight stocks under consideration (Fig. 6), the fishing mortality on fully recruited ages has decreased slightly from 0.6–0.7 in the mid-1970's to 0.4–0.5 throughout the 1980's. Fishing mortality is estimated to be slightly lower than 0.4 in 1990. Typically, for these stocks, the fishing mortality on fully recruited ages at $F_{0.1}$ would be of the order of 0.2 to 0.3. On a stock by stock basis (Fig. A3), the measures implemented after the extension of fisheries jurisdiction on these eight stocks seem to have resulted in a stabilization of fishing mortalities by preventing the occurrence of extreme fishing mortalities, i.e., higher than the 0.6–0.7 range. In addition, on average, annual fishing mortalities were reduced after 1977 in comparison to the pre-1977 values.

Biomass Trends

The total biomass of the eight stocks studies increased from approximately 1 million tonnes in 1975 to a peak of 2.4 million tonnes in 1984, decreased to 1.8 million tonnes in 1987 and increased slightly thereafter (Fig. 6). In 1990, the total biomass on these stocks is estimated at 2.2 million tonnes. In other words, it is estimated that the total biomass has doubled since the extension of fisheries jurisdiction. Of the eight groundfish stocks studied (Fig. A4), all showed a general recovery after extension of the fisheries jurisdiction. However, for one stock (namely, cod in the northeastern Gulf of St. Lawrence), the fast recovery after extension of the fisheries jurisdiction was followed in the mid-1980's by a fast decline. While the decline on that stock has now stabilized, its abundance remains in 1990 below that of the mid-1970's. In hindsight, a number of factors may have contributed to the decline of that stock: e.g., TAC's were set above recommended levels in some years; catches exceeded the TAC's in many instances; and wide scale misreporting and discarding have been recognized as key issues.

Characteristic of the Catches: Factors affecting economic performance

Total Catch (Fig. 6 and Fig. A5)

From a total of approximately 400,000 t in the mid-1970's, the total catch on the eight stocks studied increased rapidly after 1977 and fluctuated between 525,000 and 570,000 t between 1982 and 1988. Catches then declined slightly, to a level of 474,000 t in 1990. While nominal catches have increased after extension of jurisdiction, catches attained in the 1980's are much lower than those attained in the 1960's and early 1970's, in particular for 2J-3KL cod and 3Ps cod. As indicated above by the trends in the biomass and fishing mortality, the catches of the late 1960's and early 1970's were the result of fishing mortalities that were in excess of 0.8 in some years and resulted in a rapid decline of biomass. These catch levels could not be sustained and exceeded the productivity of the stocks.

Mean Age in Catch (Fig. 6 and Fig. A6)

The mean age in the catch is expected to respond not only to changes in fishing mortality but also to changes in recruitment. Variations in recruitment could induce rapid changes in the mean age in the catch, particularly for fast growing stocks which tend to be recruited in the fisheries at a an earlier age. For that reason, one would expect that the impact of fishing mortality could be detected only on the long term and that there could be some variation around the long term trend due to changes in recruitment (particularly on those stocks which are recruited at a young age). In order to depict the long term trends in mean age of fish in the catch, a 5-year running average has been superimposed on the actual data to provide an indication of the long term trends (Fig. A6). From this, it appears that the mean age has increased by more than one year on many stocks since 1977, following a progressive decline throughout the 1960's and early 1970's. This is also reflected in the overall mean for these stocks (Fig. 6).

Mean Weight in Catch (Fig. 6 and Fig. A7)

While the mean age of fish in the catch has increased since 1977, it appears that the mean weight of fish being caught has not (Fig. 6). This is not surprising since the growth of fish has declined over the period, as indicated by the changes in mean weight at age (also Fig. 6). In other words, it appears that the

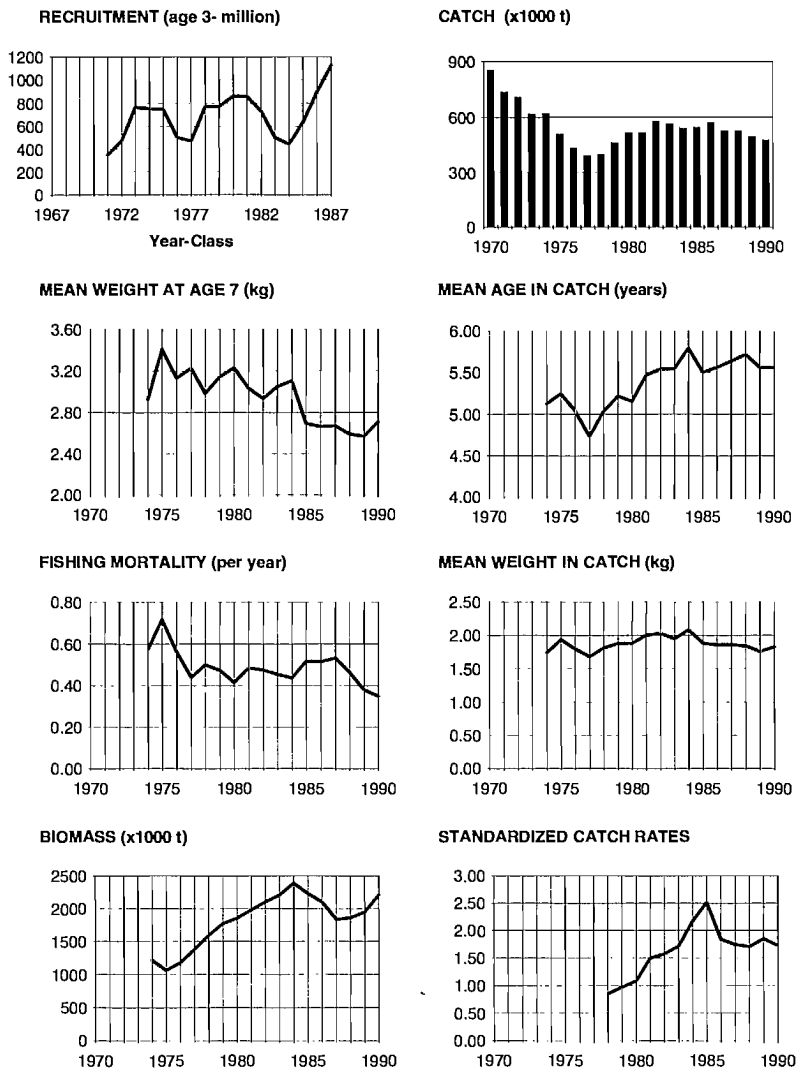


Fig. 6. Indices of stock abundance and of the performance of the fisheries for the eight groundfish stocks with a detailed analytical assessment for 1991. The nominal catches, recruitment and biomass estimates given here correspond to the total for the eight stocks. The other indices (mean weight at age 7, fishing mortality, mean age in catch, mean weight in catch and standardized catch rates) correspond to the mean calculated for these stocks. Detailed information on each stock is given in the Appendix.

expected benefits from the catch of older, larger fish did not occur simply because of the changes in growth. While there were some gains in the early 1980's, with a mean weight in the catch of around 2.0 kg, in comparison to 1.75 kg in the mid 1970's, mean weight has returned to the pre-1977 level since 1985. On a stock by stock basis (Fig. A7), there were some gains in the mean weights of fish caught for 2J-3KL cod, for 3Ps cod and for Georges Bank haddock, some loss for the two cod stocks of the Gulf of St. Lawrence (3Pn-4RS and 4TVn) and no noticeable change in the other stocks studied.

Commercial Catch Rates (Fig. 7 and Fig. A8)

Canadian offshore vessels have increased their catch rates to more than 15 to 19 tonnes of fish per day between 1977 and the mid-1980's, from eight tonnes in the mid-1970's (Fig. 7). The increase in catch rates came at an opportune time, when the price of fuel was experiencing significant increases world-wide. Higher catch rates translate into reduced harvesting costs. It is recognized that the increasing trend in the early 1980's is not only the result of increased stock abundance. Technological innovations, seasonal changes in the fisheries, and changes in fishing practices due to the introduction of enterprise allocations may all have had an influence on the trend depicted.

The standardized catch rates are also given (Fig. A8) for cod in NAFO Div. 2J-3KL, 3Ps, 3Pn-4RS and 4T-Vn (January to April). Because these catch rates are standardized to account for changes in the composition of the fleets and for changes in the seasonal distribution of fishing over time,

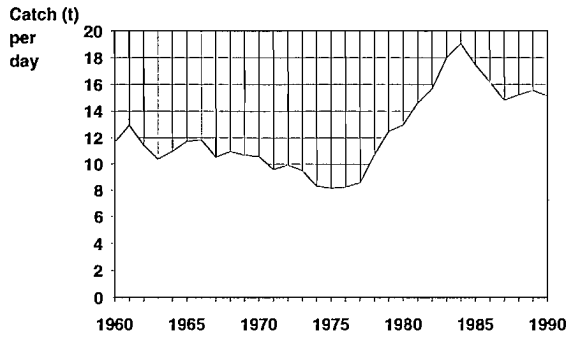


Fig. 7. Catch rates for the Canadian offshore trawlers fishing for groundfish in NAFO Areas 2 to 4: 1960–1990. Data were compiled from the NAFO catch and effort data files for the 1960–1988 period and from the DFO Zonal Interchange Files for 1989 and 1990 (J. McMillan, Marine Fish Division, Fisheries and Oceans, Dartmouth, Nova Scotia, pers. comm.).

they have been used on the four stocks, together with the research surveys, as providing an indication of stock trends (not only of fisheries performance). Standardized catch rates have not been used as an indication of abundance for the other four stock studied because of the introduction of trip limits and multi-species quotas, which changes in the management measures cannot be accounted for in the standardization.

What Has Been Achieved?

The initial objective after extension of fisheries jurisdiction was to rebuild the fish stocks and to provide the industry with a higher degree of stability and profitability. The implementation of $F_{0.1}$ was accompanied by a series of management measures aiming at improving various aspects of the industry. The collection of scientific data and scientific programs was improved to provide a sound basis for management decisions. Fisheries surveillance and enforcement programs were improved to better control the fisheries. A number of measures were implemented to improve the economic viability of the industry, measures which have led to a restructuring of the groundfish industry, to the introduction of Enterprise Allocations, limited-entry for fishing licenses, etc. Without these measures, it is unlikely that the adoption of a new strategy would have led, by itself, to the conditions depicted above.

Canada chose a management strategy of fishing at a low exploitation rate ($F_{0.1}$) in 1977 for Atlantic coast marine fish stocks because it promised a satisfactory balance among a variety of social, economic, regulatory and biological objectives. At $F_{0.1}$, spawning stocks are likely to be maintained within a "comfort zone". As the ecological system is only moderately stressed by fishing, changes that could occur in the balance of species are less probable; on the U.S.A. Northeast continental shelf, for example, the domination of the demersal finfish biomass by cartilaginous fish may be an expression of such ecological changes. A low exploitation rate strategy also provides a buffer against intrinsic deficiencies in the system, such as stock assessment uncertainties, statistical misreporting and enforcement difficulties. From an industry viewpoint, the low exploitation rate results in higher stock abundance and hence higher catch rates, i.e., good fishing success. While employment in harvesting is lower than at high exploitation rates, incomes are higher. In short, the aim was to put the industry in a more desirable situation than that prevailing in the early 1970's, as a result of high exploitation rates.

Where are we now (i.e., at the end of 1991) in relation to where we were fifteen years ago and in relation to the targets established in the mid-1970's? Have the stocks rebuild and have the danger zones been avoided? The analysis of the fisheries data and of scientific assessments suggests the following:

- *Catches:* As a whole over all groundfish stocks combined, both the total catches and the Canadian catches have improved in relation to the mid-1970's. The increase is particularly strong for cod, despite the decline noted in the late 1980's. Notable exceptions to this are the haddock stocks on the Scotian-Shelf, which are now reduced to bycatch fisheries, and the flatfish stocks on the Nose and Tail of the Grand Banks, which are transboundary stocks.

- *Fishing Mortality:* The fishing mortalities have not been reduced to $F_{0.1}$ for any significant amount of time on any of the stocks reviewed. However, the high fishing mortality exercised in the late 1960's and early 1970's seem to have been avoided and, on the average, the level of fishing mortality is somewhat lower than that exerted prior to 1977.

- *Stock Abundance:* Of the stocks reviewed, most stocks have recovered and reached a level that is about twice that of the mid-1970's. The one exception is cod in 3Pn-4RS, where a fast recovery after 1977 was followed by a quick decline in the late 1980's.

- *Factors affecting fishing operations:* Commercial catch rates of large Canadian offshore trawlers have almost doubled since 1977.

- *Factors affecting processing:* There was a small gain (about 15%) in the average weight of fish caught in the early 1980's but that gain was lost thereafter and the average weight of fish caught is now about equal to that of the mid-1970's. With that respect, there seem to be no net gain or loss.

In sum, all indicators suggest a significant improvement on most grounds in relation to the mid-1970's, despite the setback that took place in the late 1980's. However, while the fishing pressure on most stocks seem to have been reduced, it is by no means close to $F_{0.1}$. Nonetheless, in the 14 years of application of this strategy, exploitation rates were reduced substantially in comparison to the levels of the early 1970's and could now be described as being moderate. The biomass

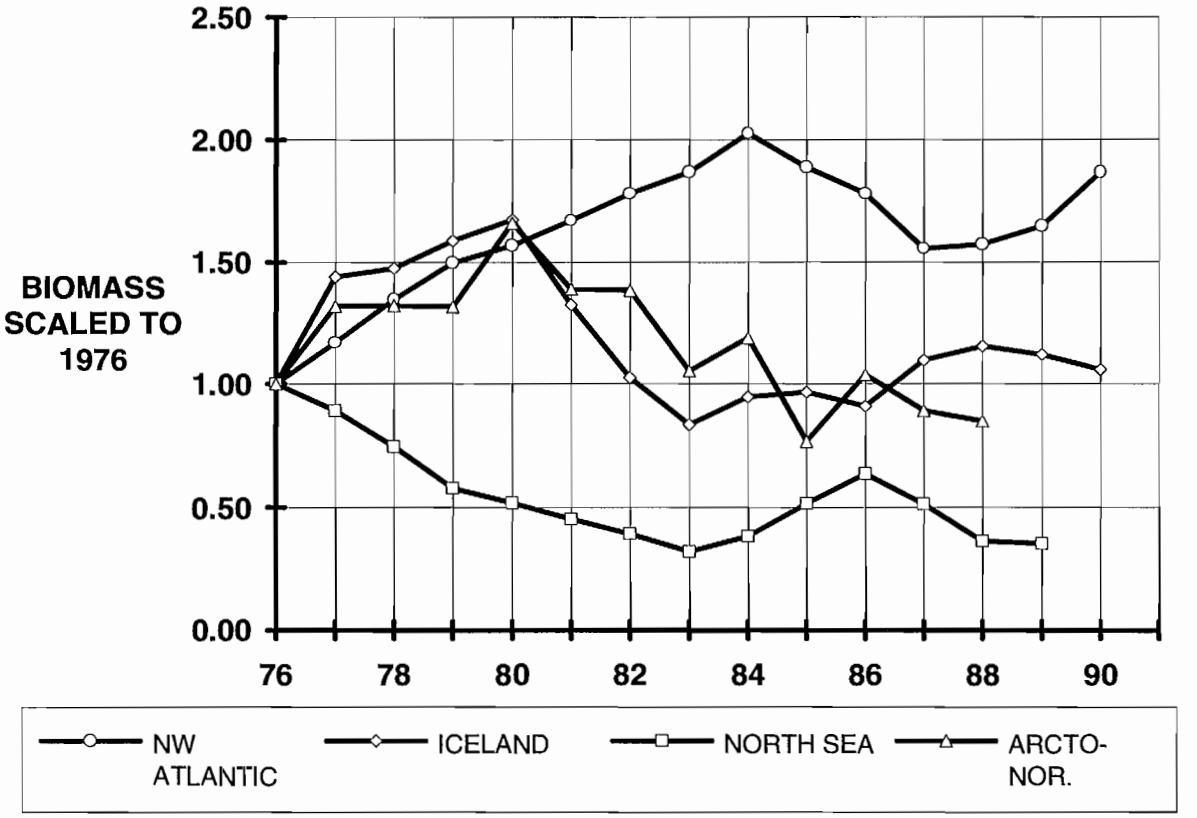


Fig. 8. Comparison of relative trends in the biomass estimated for the cod stocks in the NW Atlantic with that of the cod stocks around Iceland, in the North Sea and the Arcto-Norwegian area (Data source: International Council for the Exploration of the Sea).

of the resource is now, in aggregate, substantially higher than before extended jurisdiction and is likely close to the long-term value associated with moderate exploitation rates. There are, however, some important exceptions, in particular the haddock stocks, which are depleted and in need of protection. Thus, in general, conservation objectives have been at least partly met and, in hindsight, the buffer provided by $F_{0.1}$ has appeared a regulatory necessity. Canadian groundfish landings, meanwhile, have increased substantially in both quantity and value while employment has increased in both the harvesting and processing sectors. Trends in economic viability of fishing enterprises and in fishermen's income, where documented, have generally shown increases. While a low exploitation rate strategy may establish the conditions within which industry viability and adequate incomes are possible, it is the method of implementing the strategy which establishes whether satisfactory benefits actually result.

baselineskip12pt minus 1pt The rebuilding of cod stocks stands in sharp contrast to the trends seen in cod stocks managed under management strategies in other jurisdictions (Fig. 8). The major Northeast Atlantic cod stocks have shown either fairly consistent declines or, at best, variations with no trend over the period. The stocks having faster individual growth rates and younger ages of maturity, such as cod in the North Sea, would be expected to increase at a faster rate than cod in Canadian waters and to respond faster to reductions in fishing pressure due to their intrinsic growth rate. The fact that these stocks experienced fishing mortalities much higher than those realized on cod stocks in Canadian waters may explain, at least in part, the differences in stock trends.

Despite the stock recovery that took place following the extension of fisheries jurisdiction in 1977, the Canadian groundfish stocks are not as abundant as the industry believed they would be. The forecasts of the late 1970's early 80's were optimistic as they were based on the conditions that existed in the northwest Atlantic until then. Just like analysts forecasting the demand for petroleum products in the early 1970's could not predict the 1973 oil price increases and the ensuing decrease in demand, fisheries biologists could not predict that recruitment in the post-extension of jurisdiction period for 2J3KL cod would be so drastically smaller than the long term average and that weights at age would decrease so markedly in many cod stocks. Even in hindsight, there were no data or method available in the late 1970's that could have

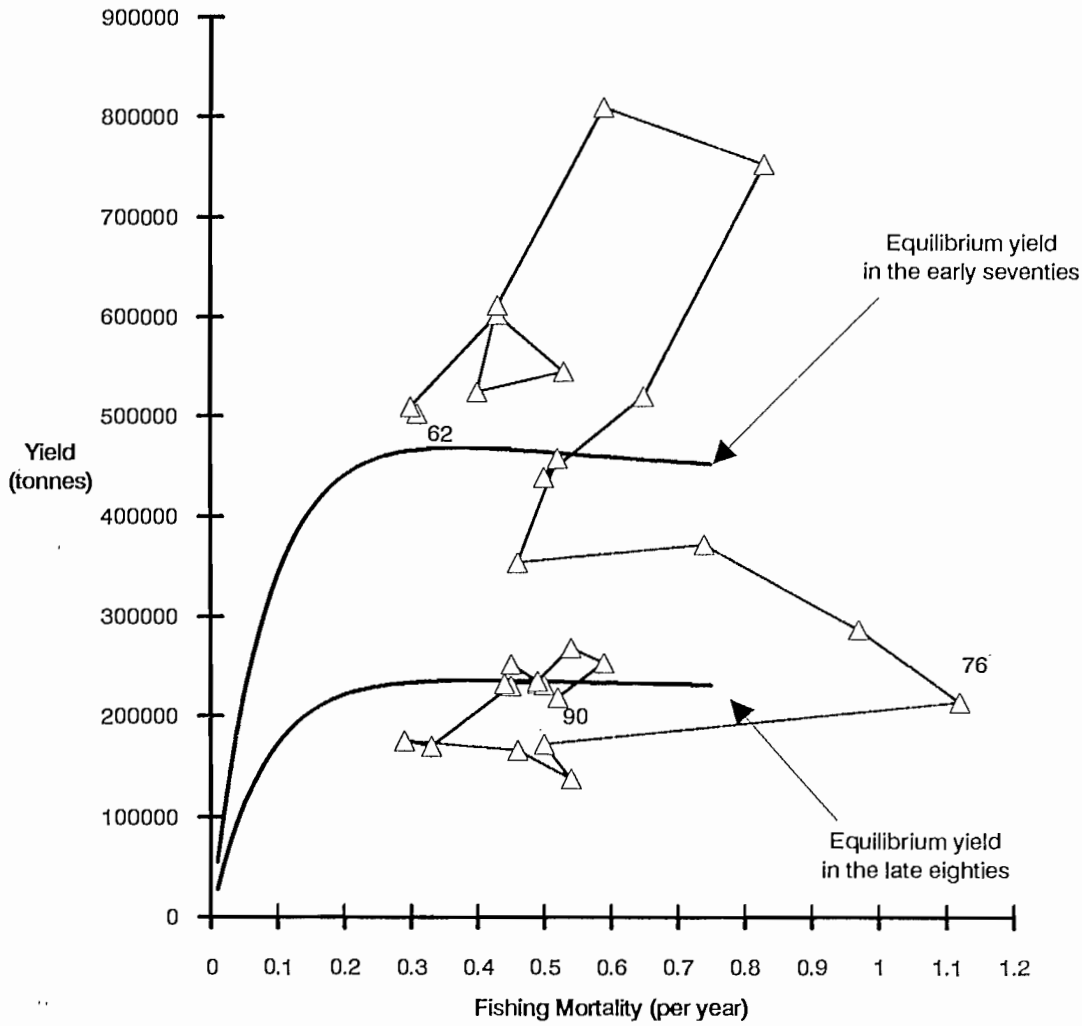


Fig. 9. Productivity of the northern cod stock (NAFO Div. 2J, 3K and 3L). The productivity of the early 1970's is compared with that of the late 1980's. The productivity curves correspond to the yield per recruit (using the growth characteristics for these periods and assuming knife edge selection at age 5) multiplied by the average recruitment for the year classes contributing to these periods.

allowed predictions of the changes in recruitment and growth that were observed.

While the fisheries management measures implemented since 1977 have played a key role in achieving a reduction of fishing mortality, there are a number of reasons that could explain the excess fishing mortality above the reference $F_{0.1}$ level: in particular,

- As per the provisions of the Groundfish Management Plan, the TAC's have often been set from the implementation of the "50% rule" rather than $F_{0.1}$.

- Misreporting and enforcement difficulties have led to catches in excess of the TAC's on some stocks. Misreporting and discarding practices have been recognized and may have been particularly strong in southwest Nova-Scotia and in the northeastern Gulf of St. Lawrence. A gradual erosion of the scientific data required for stock assessments has been observed, particularly for straddling stocks and in areas where overcapacity of the fishing fleets remains a key issue.

- There is little control of fishing activities beyond 200-miles on certain transboundary stocks and the catches realized on straddling stocks have often been in excess of the recommended levels in recent years.

- As documented recently for many stocks assessed with age-structured models (Anon., 1991B), there is a certain inconsistency between recent and historical population and fishing mortality estimates obtained from such models (Sinclair et al., 1991; Rivard and Foy, 1987). While the reasons for this are still unclear, the factors leading to such systematic effects are probably different for each stock. Sinclair et al. (1991) suggest that certain misreporting patterns may result in inconsistencies of the type identified in the Northwest Atlantic.

In one other case (Anon., 1991B), migratory patterns unaccounted for in the models used are believed to be a key factor. The increase of catchability in the offshore fleets due to learning and fleet improvements may also have affected the commercial catch rates used in the assessment.

- The relatively low precision of population estimates may be another contributing factor. A poor precision would not lead, by itself, to fishing mortalities higher than the reference on average. However, in the context of the Groundfish Management Plan, a low precision means that the 50% rule will be invoked even when there is no real change in fishing mortality.

In view of the above, one has to ask the question — Was $F_{0.1}$ really given a chance? In this evaluation of $F_{0.1}$, it quickly became obvious that it is not possible to separate the problems in the implementation of $F_{0.1}$ from the $F_{0.1}$ reference itself. In the implementation of any strategy, it may not be possible to achieve the theoretical gains attributed to the strategy simply because of the constraints or limitations inherent in its implementation. The empirical evidence developed here provides solely an evaluation of the implementation of $F_{0.1}$ in the Canadian context.

The setback of the late 1980's, which resulted in a decline of the catches in relation to the levels reached in the early 1980's, has been attributed to two factors:

- For northern cod, recruitment has been much lower after 1977 than in the 1960's and early 1970's. The differences in the productivity of northern cod for the early 1970's in comparison to that of the late 1980's are explained solely by the differences in recruitment (Fig. 9). As a consequence of this, the increased effort of the 1960's, rather than leading to a stabilization of the catches around 450,000 t as expected from the productivity of the stock at that time, lead to rapidly falling catches. The reductions in effort that followed led to a stabilization of the catches and a recovery to the new equilibrium, which reflects a production that is much lower than in the early 1970's. The trajectory of the catches and fishing mortality presented in Fig. 9 from 1962 to 1990 illustrates the transient path in relation to the equilibrium production curves.

- Changes in growth: In most cod stocks, the weight of an individual fish is now considerably smaller than it was in the late 1970's. Consequently, even if the total number of fish has increased, the total weight has not increased as much because each fish weighs less than a fish of the same age did in the late 70's. This was illustrated quite eloquently in the southern Gulf of St. Lawrence cod stock where an age 7 cod weighed about 2.4 kg in 1977 but, after a steady decline in the years following extension of fisheries jurisdiction, was only 1 kg in 1987.

The environment is often considered to be either stable or fluctuating without trend. The fact that it is not so is becoming more widely known as discussions on climatic changes are more common. It became difficult to ignore in 1991, when environmental conditions appear to have been the driving factors in an abrupt decline of the northern cod stock. While CAFSAC could not identify a single factor, water temperatures off Labrador and northeastern Newfoundland were below normal; the ice coverage was greater than normal and persisted for a record length of time; low ocean temperature persisted throughout summer; and early autumn and the cold intermediate layer of waters less than 0°C of the Labrador current was at or near its greatest size. While the environment affects fish populations in ways that are still unclear, the impact of changes in environmental conditions on geographic distributions, on mortality of fish eggs and larvae, and on growth rates can be substantial and, in reality, fluctuations in stock sizes and catches are natural phenomena that cannot be eliminated. The fact is that changes or shifts in the environment may be sufficient to annihilate the gains made under stock rebuilding strategies; alternatively, when such changes have a positive effect on populations, they may contribute to stock rebuilding even under poor management.

Discussion

In economic terms, the extension of fisheries jurisdiction in 1977 stimulated a decade of economic growth in Atlantic Canada's commercial fisheries. By 1986, the quantity of groundfish landed had increased by 67%. Between 1977 and 1988, the number of fish plants increased from 519 to 953. This, in return, has resulted in increased pressure for larger quotas.

In summary, this review leads to the following conclusions:

- Attempts within ICNAF to regulate for maximum yield ended up with drastic overfishing and stock declines.

- Pursuit of an $F_{0.1}$ target exploitation rate has been accompanied with a general resource recovery, with the exception of haddock stocks on the Scotian Shelf and some transboundary stocks on the Nose and Tail of the Grand Banks.

- Fishing at $F_{0.1}$ improved the economic viability of the Canadian fishing industry by reducing operating costs and improving efficiency. It has established conditions for a stable and economically viable industry.

- Licensing and vessel replacement policies have not proven to be an adequate vehicle for control of fleet capacity and over-capitalization is stressing the management system. Although progress has recently been made, more attention needs to be paid to this element by initiation of ongoing, active, capacity management supported by appropriate capacity monitoring and research on factors affecting fleet capacity and utilization.

While the initial Canadian objective was to move the industry in a direction where stability and profitability could occur, the Task Force on Atlantic Fisheries (1982) established firmer objectives. The primary objectives identified by the Task Force were to bring economic viability and maximum employment. Despite a general recovery, the problems that developed in the Canadian groundfish industry after extension of jurisdiction were sufficient to convince certain sectors of the industry that the benefits that they expected were not happening. These problems still remain and are clearest in the Scotia-Fundy Region where the active inshore fleet has expanded beyond the capability of the resource to support the aspirations of this sector under a strategy based on a moderate exploitation level. This is an incipient problem in other Regions and stems from the inability of existing controls to counteract the tendency for over-capitalization inherent in common-property systems. While this may have resulted, at least temporarily, in higher employment than in a more rigorously controlled system, it has dissipated a significant proportion of the economic benefits which would have been accrued. Furthermore, the demands of over-capitalized fleets for more fish are stressing the management system and may be the primary reason why its adequacy is being questioned. Coincidentally, over-capitalization of onshore processing capacity is also generating demands for increased fish supplies. Although the need to match fleet capacity with resource availability has been recognized, little progress has been made and little attention has been paid to monitoring capacity trends or to developing an understanding of the factors affecting fleet capacity and vessel fishing power as a result of vessel design, fishing gear innovation and technological developments.

The Canadian public debate on groundfish management is focussed primarily on the suitability of various control measures: for example, the focus is often on the implementation of measures such as catch quotas, closed areas or gear restrictions so as to better control exploitation rates or to provide a better distribution of exploitation over the age groups in fish populations. Although arguments are often couched in conservation terms, the underlying issues are in reality of a social and economic nature and usually concern the distribution of benefits. The present problems in the industry are not rooted in biology, i.e., they did not originate from conservation issues, except in the case of haddock and some transboundary resources. Interest parties debate about management strategies and annual management plans in light of their immediate operational requirements. The debate need to be broaden so as to provide a forum for a discussion of management objectives and strategic requirements, which should provide the context for a meaningful discussion on tactics. Only when operational definitions can be given to management objectives can the utility of alternative measures for their achievement be evaluated. In support of such an initiative, greater emphasis on research and monitoring of socio-economic factors would be required.

In the context of this workshop on the evaluation of reference points and risk analyses, there are a number of elements that should be retained from the Canadian experience in implementing the $F_{0.1}$ strategy:

- Possible density-dependent responses in fish populations should be taken into consideration in the evaluation of reference points for the 1990's. It does remain, however, to define how this should be done in absence of quantitative estimates of the density-dependent effects and in absence of a complete understanding of the underlying biological mechanisms.

- Risk analyses should account for all types of uncertainties in assessments. In particular, the tendency to overestimate stock abundance in hindsight (as identified through retrospective analyses) should be taken into consideration in the quantification of risks.

- Recruitment variability has played a key role in driving variations in TAC's. Should we design reference points that protect against undue variations in the TAC's in the face of varying recruitment. Alternatively, should we take higher risks in years of poor recruitment, in so far the risks remain within prescribed limits (to be defined beforehand)?

- There is a need to reiterate both long and short term management objectives in the context of economic, social and biological implications, and ensure that reference points are consistent with them.

- With the flexibility provided by multiple reference points comes the risk of not achieving any specific objective, particularly when these objectives are in conflict. There is a need for guidelines in selecting reference points appropriate for a given case and consistent with management objectives (both short and long term).

- A choice of the most suitable management strategy and the best measures for an effective control of fisheries can be reached only in the context of overall objectives. General objectives of economic viability and maximization of employment have not been given operational definitions to provide yardsticks against which system performance can be measured. More attention needs to be paid to monitoring of industry trends and research on socio-economic aspects of regional fisheries.

Acknowledgements

We would like to thank Jim McMillan, Marine Fish Division, Fisheries and Oceans, Dartmouth, Nova Scotia, who provided us with a summary of historical catch rate trends for offshore trawlers. His contribution permitted us to include the most recent information on catch rates, as compiled from the Zonal Interchange Files. We would also like to thank Dr. R. G. Halliday, Department of Fisheries and Oceans, Dartmouth, Nova Scotia, and an anonymous referee for reviewing earlier versions of the paper and making many helpful suggestions.

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Appendix

Trends in stock abundance and in fisheries characteristics for stocks with analytical assessments.

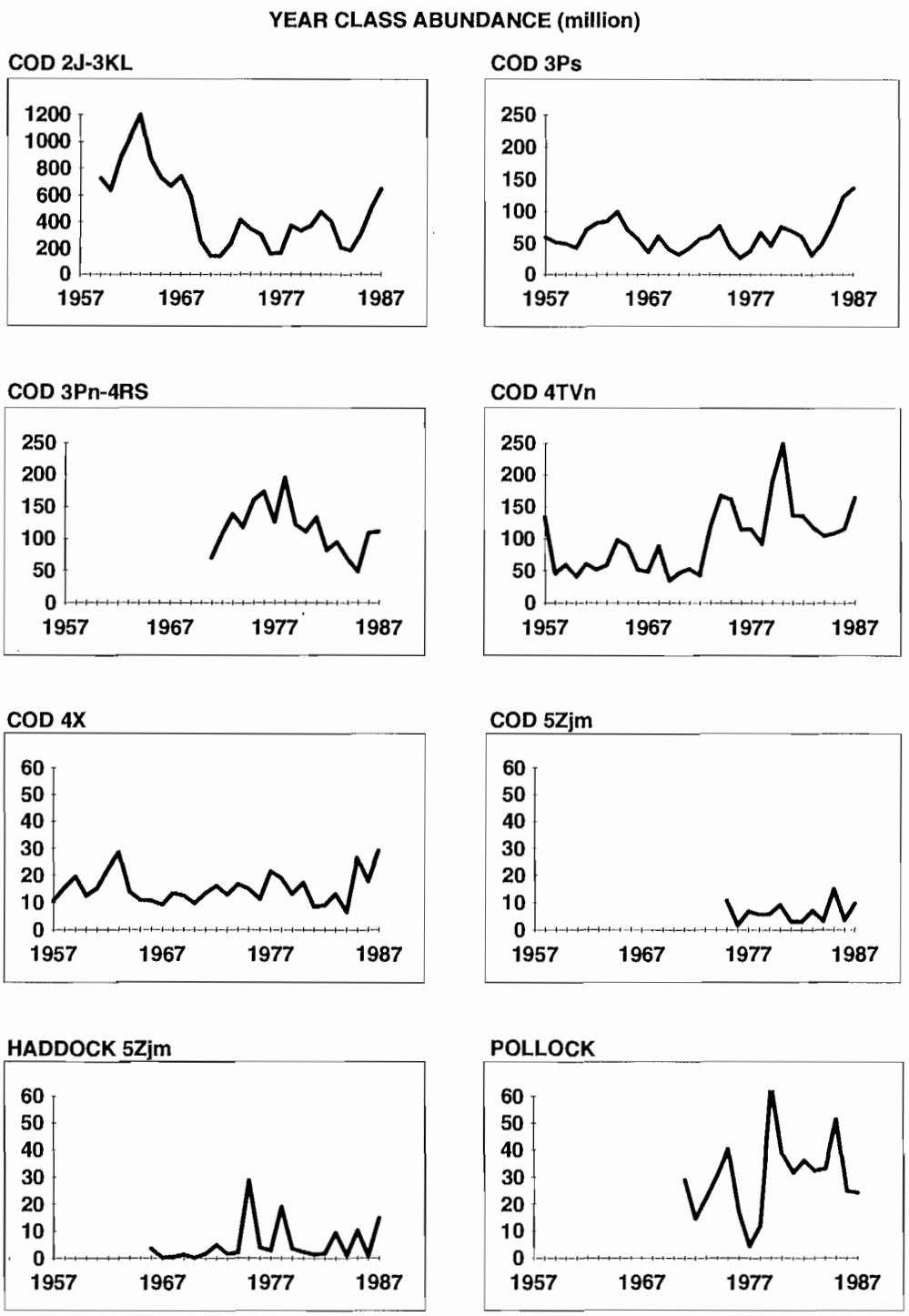


Fig. A1. Year-class abundance (million fish).

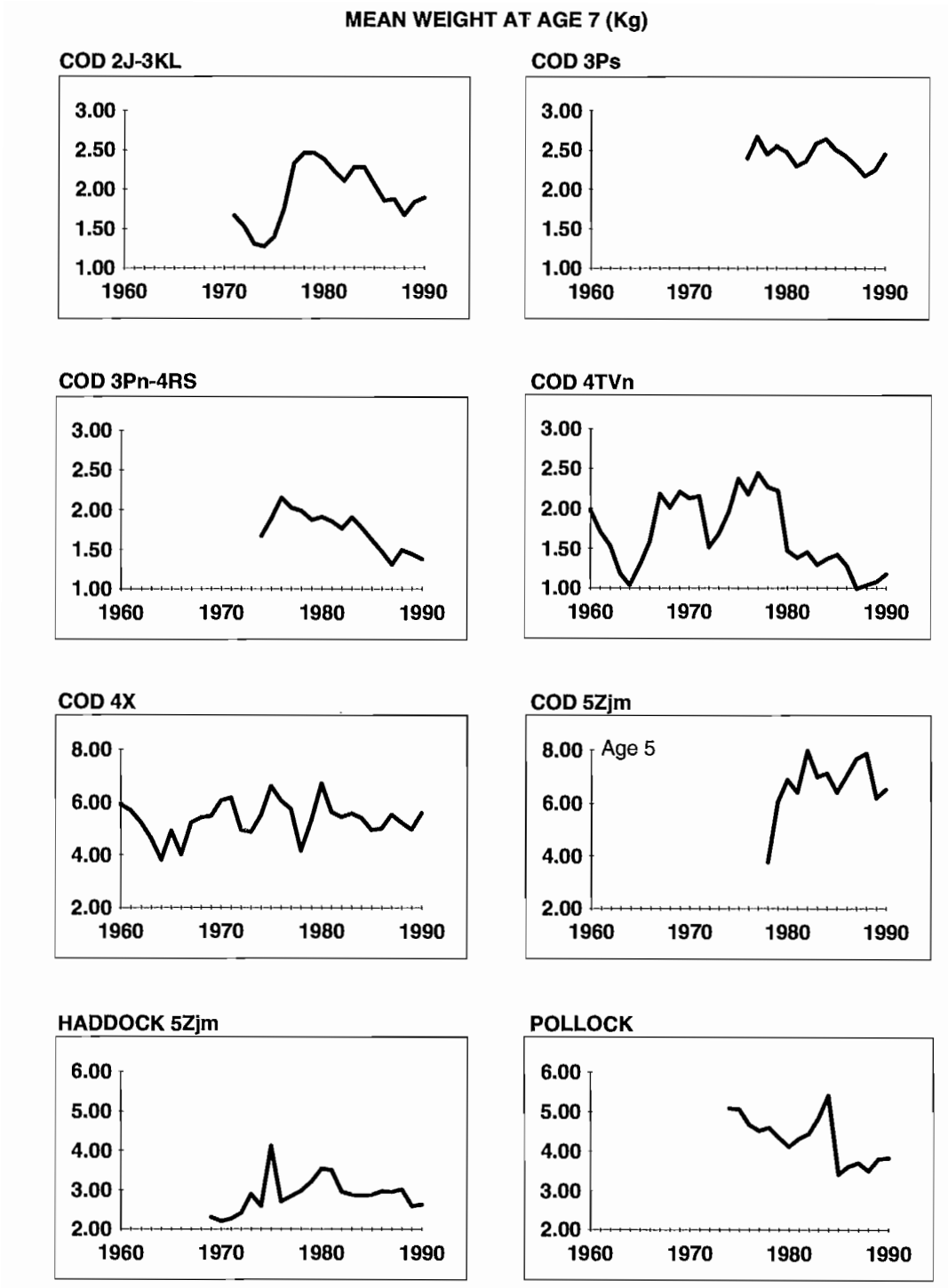


Fig. A2. Mean weight at age 7 (except for cod in 5Zjm where mean weight at age 5 is given because of faster growth rates).

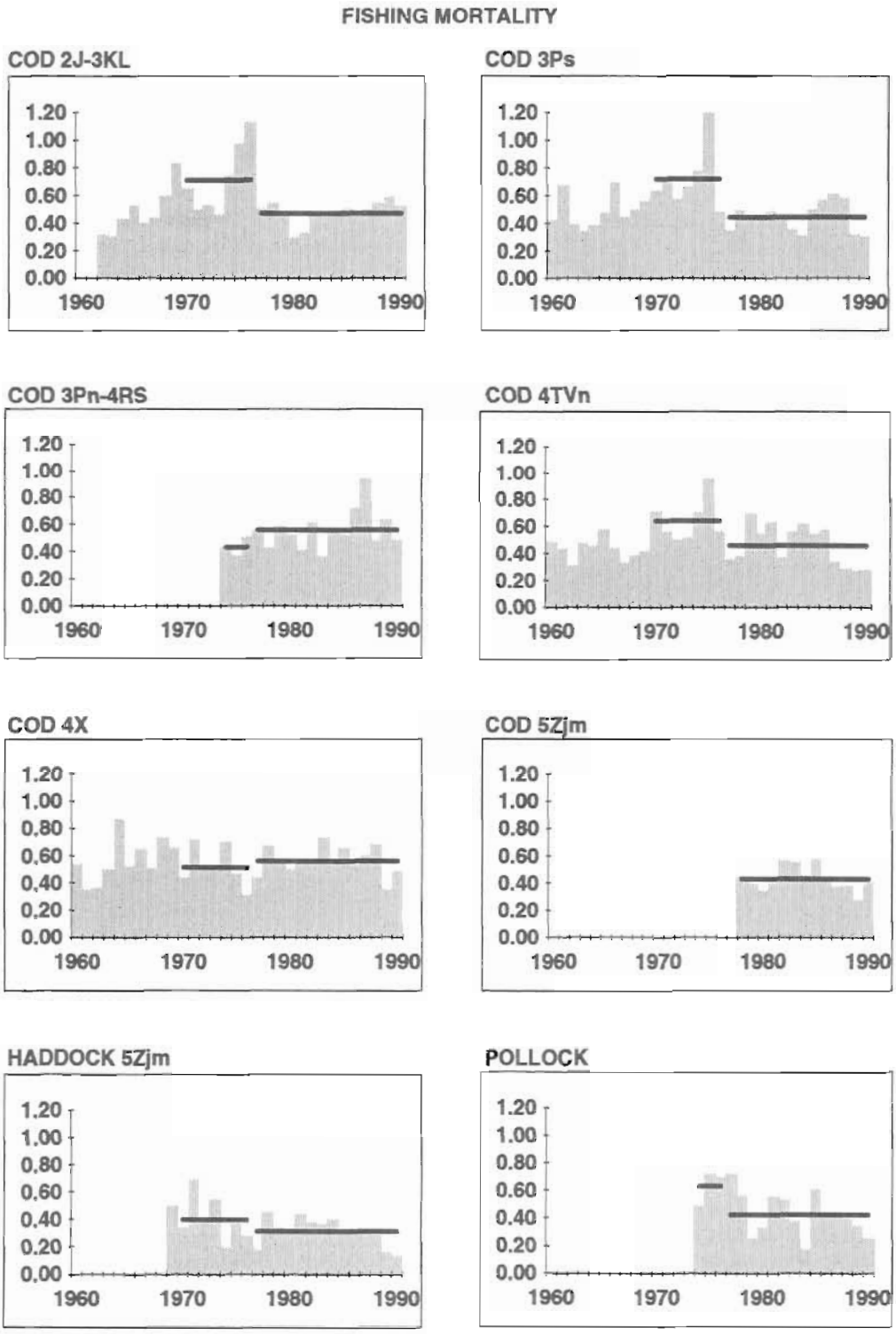


Fig. A3. Fishing mortalities (instantaneous rates per year) for fully recruited ages. The solid line represents the average of annual fishing mortalities for the 1970–1976 period (where data available) and for the 1977–1990 period. Typically, for these stocks, the fishing mortality on fully recruited ages at $F_{0.1}$ would be of the order of 0.2 to 0.3.

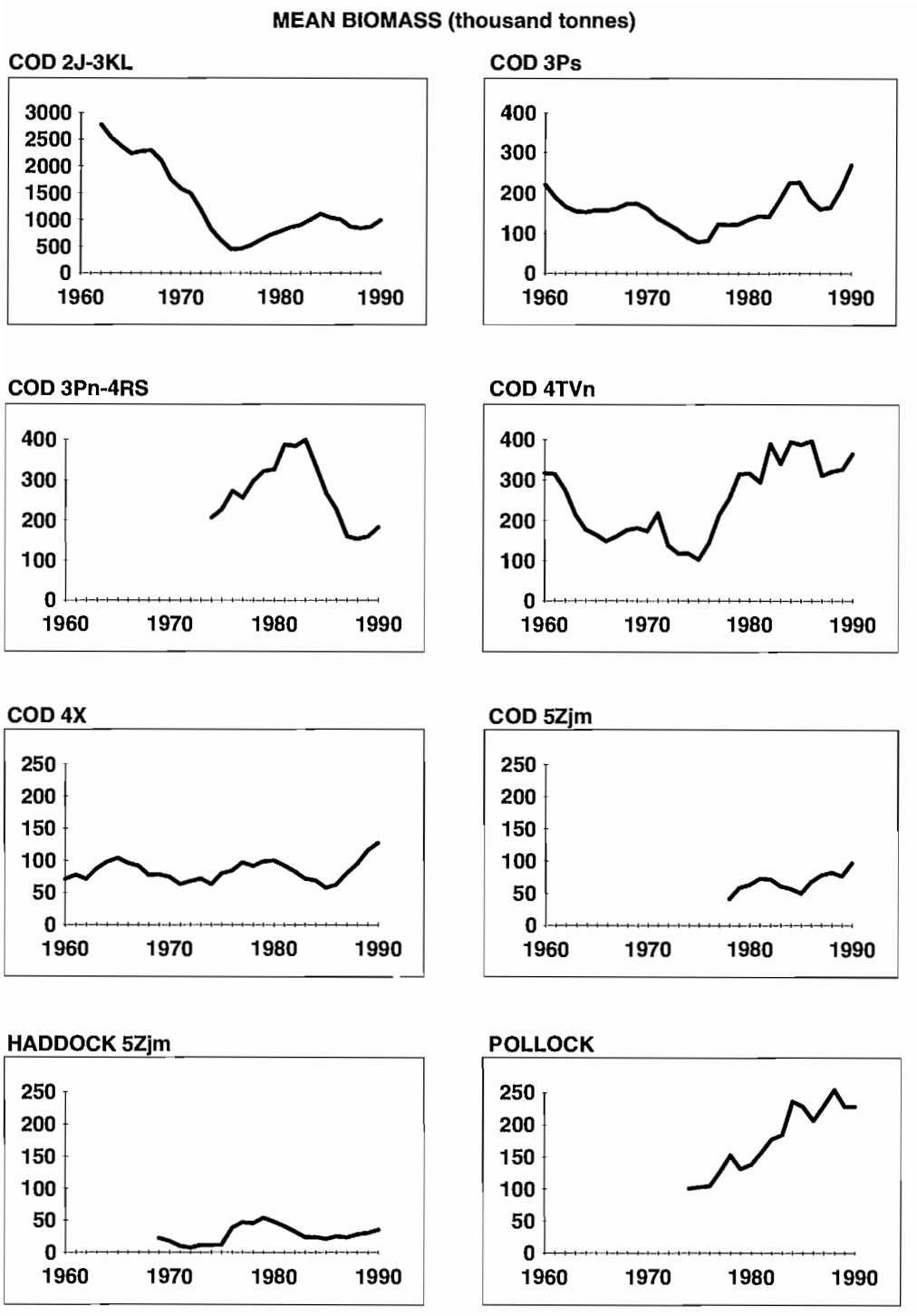


Fig. A4. Mean biomass (thousand tonnes) estimated from the assessments carried out in 1991.

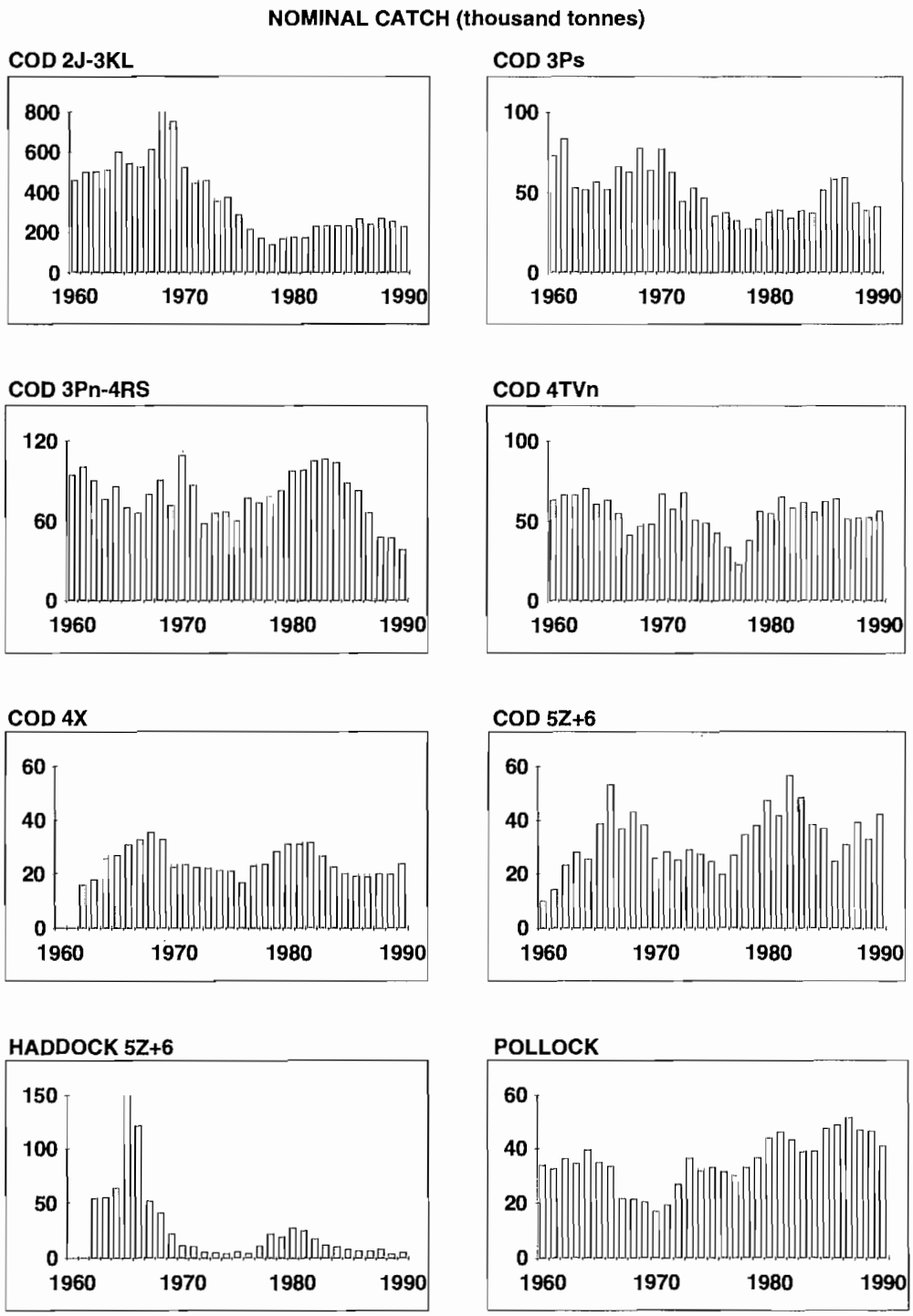


Fig. A5. Nominal catch (thousand tonnes): 1960–1990.

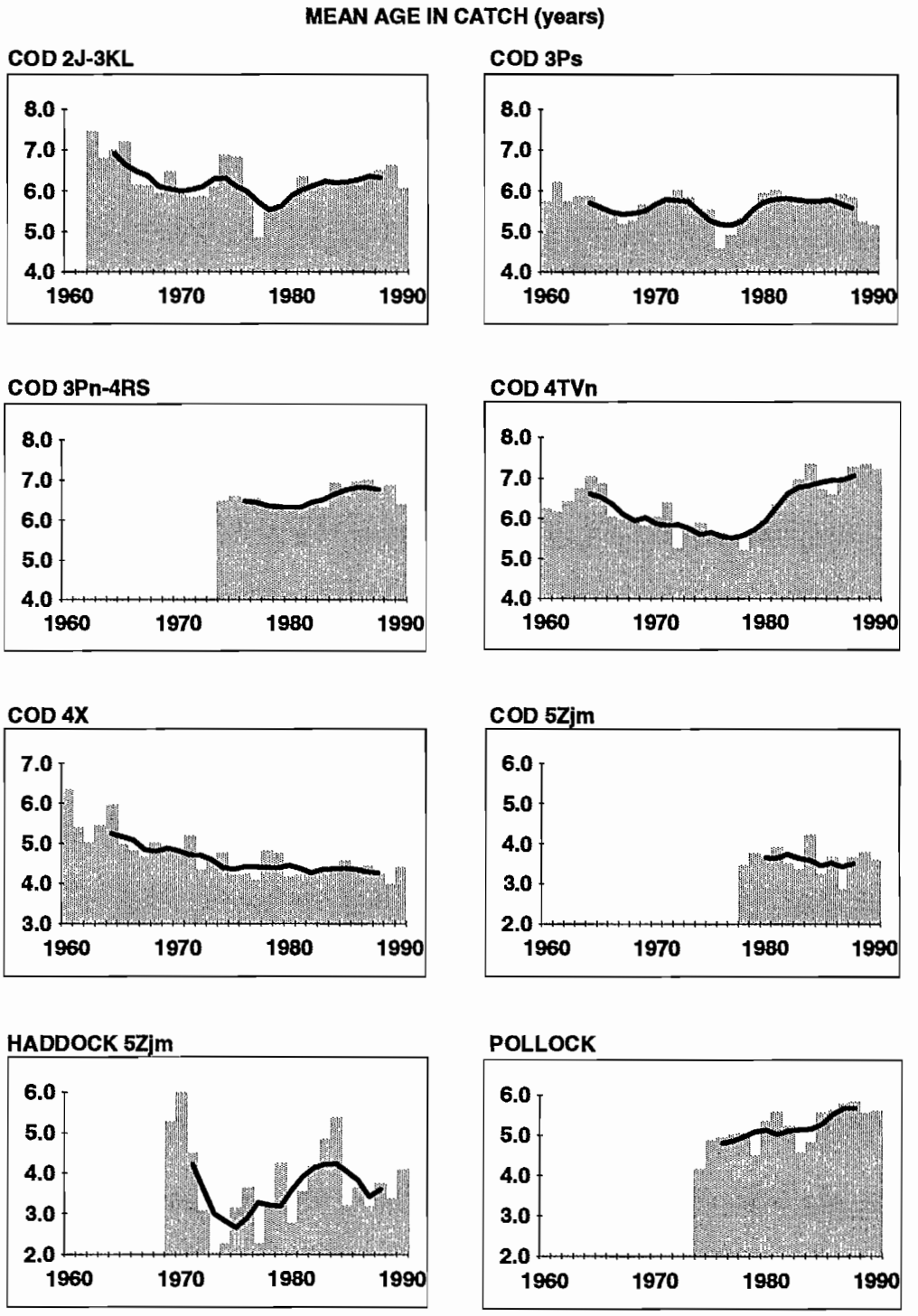


Fig. A6. Mean age in catch (years) derived from commercial catch sampling. The shaded area represents the annual estimates while the solid line represents the 5-year running average.

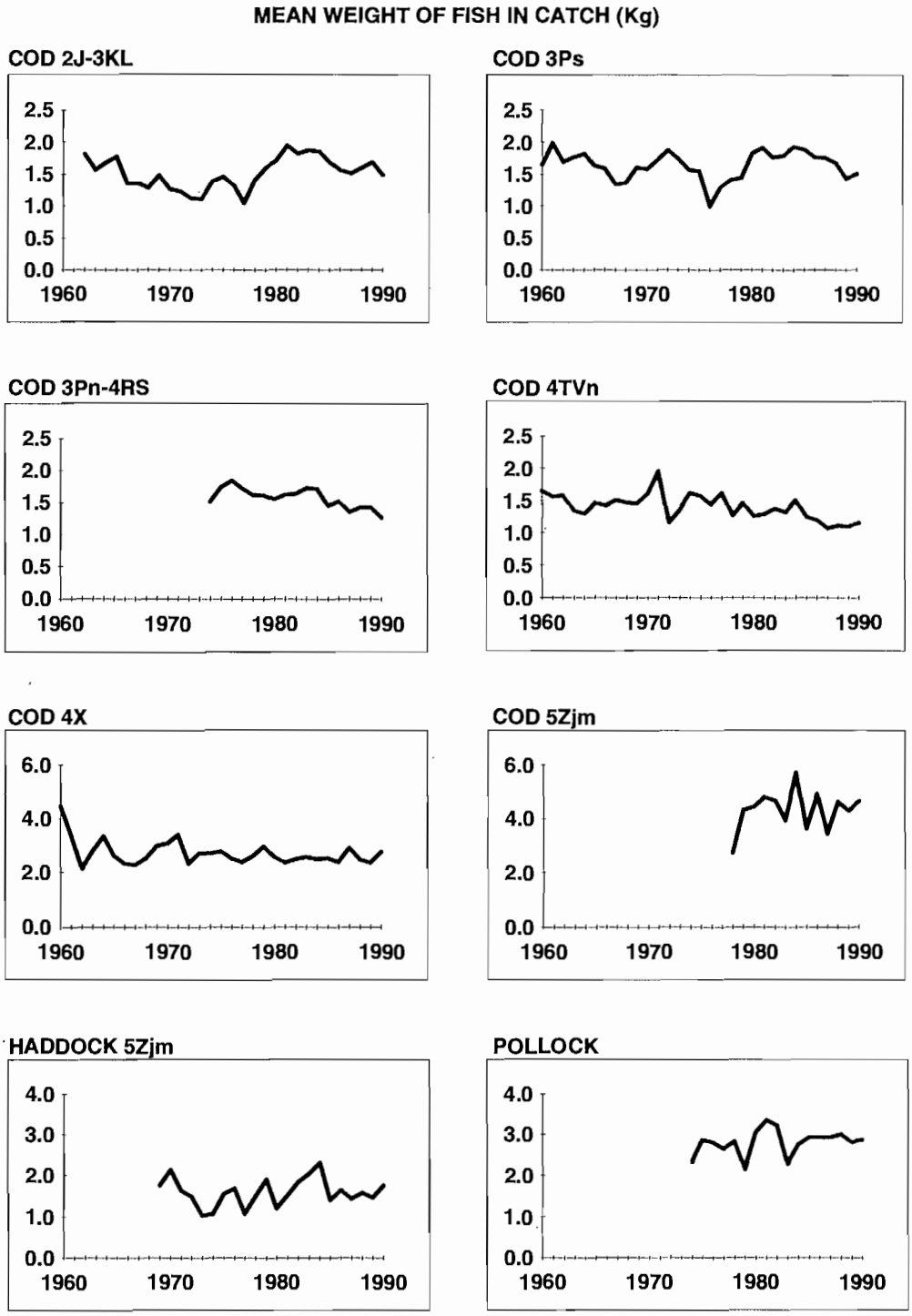


Fig. A7. Mean weight of fish in catch (kilograms).

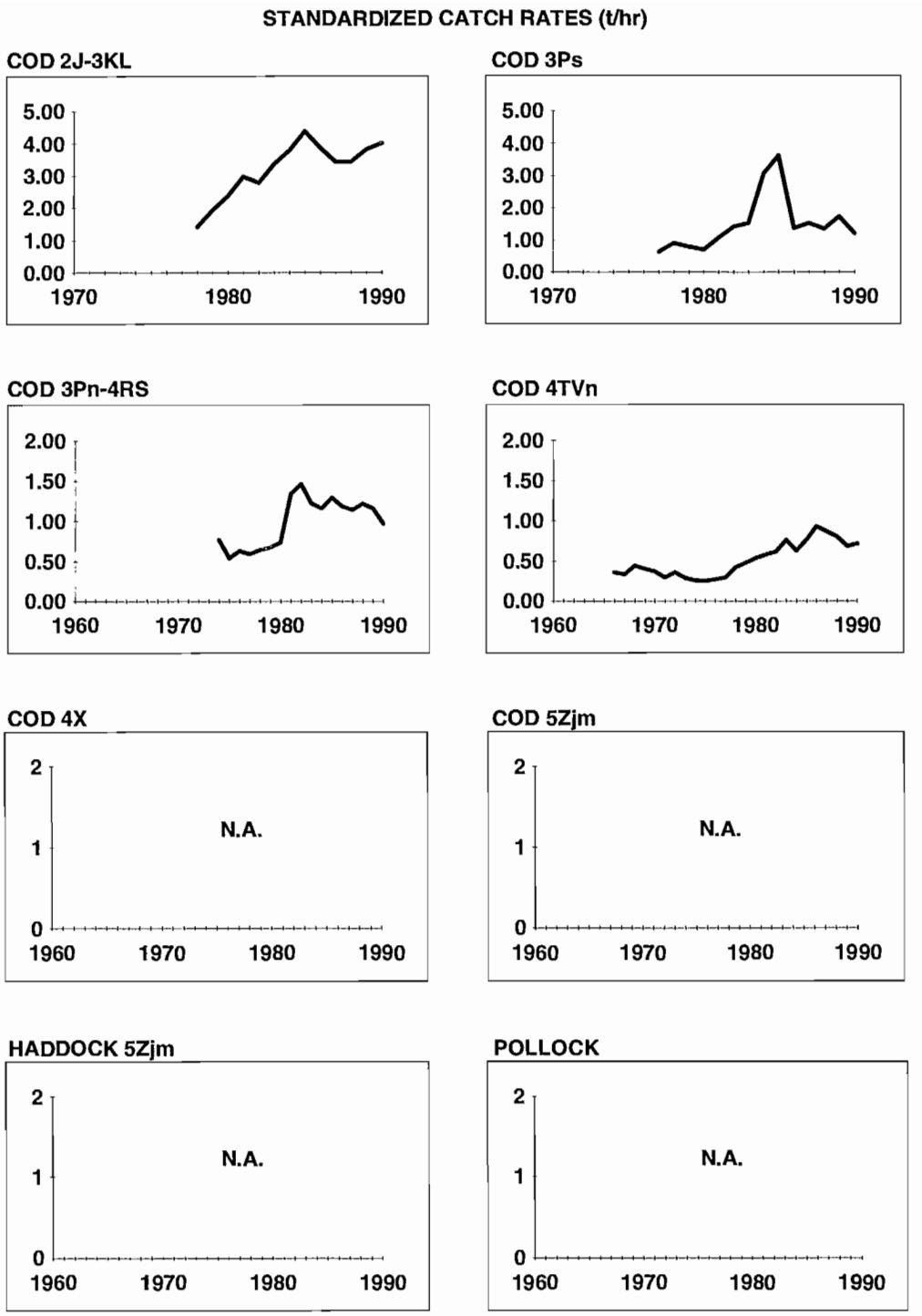


Fig. A8. Standardized catch rates (t/hr), as determined in the 1991 assessments. Information on catch rates is not available for stocks in the southern areas.

Reference Points for Fisheries Management: the ICES Experience

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Hildén, M. 1993. Reference points for fisheries management: the ICES experience. p. 59–65. *In* S. J. Smith, J. J. Hunt and D. Rivard [ed.] Risk evaluation and biological reference points for fisheries management. Can. Spec. Publ. Fish. Aquat. Sci. 120.

The ICES fisheries management advice has made extensive use of the reference points F_{max} and $F_{0.1}$. The fishing mortality rates of many of the stocks assessed by the ICES are well above these reference points. ICES has recommended total allowable catches (TAC) that have aimed at reducing the mortality rates, but the recommendations did not lead to a general improvement in 1978–1990. Depleted pelagic stocks were successfully rebuilt, but for many stocks fishing mortality rates rose, contrary to the aims of the recommendations. This suggests management needs to take other objectives into account besides those implied by the reference points used by the ICES. Uncertainties in the available data and the lack of adequate control variables may also have contributed. In these circumstances, F_{max} and $F_{0.1}$ are not very useful as operational reference points. Greater emphasis should be given to analyses of likely events and short and intermediate term strategies and tactics.

L'avis de gestion des pêches du CIEM a fait appel énormément aux points de référence F_{max} et $F_{0.1}$. Le taux de mortalité par pêche de bon nombre de stocks évalués par le CIEM est bien supérieur à ces points de référence. Le total des prises admissibles (TPA) recommandé par le CIEM visait à réduire les taux de mortalité, mais les recommandations n'ont pas entraîné d'amélioration générale entre 1978 et 1990. Les stocks pélagiques affaiblis ont été rétablis, mais pour de nombreux stocks, le taux de mortalité par pêche a augmenté, ce qui est contraire aux buts des recommandations. Ces résultats portent à croire que la gestion doit prendre en compte d'autres objectifs en plus de ceux qui sont sous-entendus par les points de référence utilisés par le CIEM. Les incertitudes des données existantes et l'absence de variables de contrôle appropriées peuvent aussi être en cause. Dans ces conditions, les valeurs F_{max} et $F_{0.1}$ ne sont pas très utiles comme points de référence opérationnels. On doit insister davantage sur les analyses de phénomènes probables et sur les stratégies et tactiques à court et à moyen terme.

Since the quantitative analysis of fish population dynamics published by Beverton and Holt (1957), the concept of maximum yield per recruit has been an important element in ICES fisheries advice. The first report of the Liaison Committee dealt largely with mesh experiments conducted to find ways of improving yields (ICES 1963). The second report explicitly gave yield (per recruit) curves, and dealt with the relationship between recruitment and spawning stock (ICES 1964).

In the 1970's, the objectives of fisheries management and fisheries management advice were specified on several occasions. In 1973, the Gdansk agreement on the Baltic Sea submitted a request to ICES "to analyse the state of exploitation of the stocks of herring, sprat, cod and flatfish in the Baltic Sea and to advise as to regulation which might be used for approaching optimum yield of the stocks of the said species" (ICES 1974). The report of the Liaison Committee to the 14th annual meeting of the North East Atlantic Fisheries Commission in 1976 also discussed the objectives of fishery management (ICES 1977). The Liaison Committee defined an integrated management objective that incorporated the concepts of F_{max} and $F_{0.1}$, optimization of the exploitation pattern, the maintenance of spawning stock for desirable recruitment, buffer stocks to mitigate the effects of recruitment variability and a high catch per unit of effort.

In 1977, the system for producing scientific advice was changed within the ICES when the Liaison Committee became the Advisory Committee on Fishery Management (ACFM), a body with national representation (ICES 1979). The general objectives of management advice did not change, but in 1981 the ACFM stressed the role of management bodies:

> Ideally managerial authorities would define their objectives for the different stocks or fisheries and ACFM would thereafter evaluate the consequences of these management strategies and define the biological constraints for the attainment of these objectives. Without clear objectives at hand from managerial bodies, ICES has had to develop certain management objectives which are mainly based on purely biological considerations (ICES 1982).

During the 1980's, fisheries advice was thus largely based on two reference points, F_{max} and $F_{0.1}$, suggesting that managerial bodies generally accepted the biological objectives, at least on paper. As noted above, the International Baltic Sea Fishery Commission had already agreed on predominantly biological objectives, except for the reference to optimum yields

rather than maximum yields (ICES 1974). In 1987, two additional reference points, F_{med} and F_{high}, were introduced and the indicative nature of the reference points was made clear (ICES 1988).

The main instruments for achieving the objectives of fisheries management in the ICES area have been technical regulations and, since the mid-1970's, total allowable catches (TAC). Since 1974, ICES has issued TAC recommendations for most major stocks (ICES 1974 and later). In 1990, the ACFM adopted a different policy for some groundfish stocks by recommending a direct reduction in effort rather than a TAC to achieve a specific F.

The role of the reference points in fisheries management advice can be clarified by looking at the successes and failures of fisheries management. Several reviews of the management of species or stocks have been made (e.g., Daan 1978; Jakobsson 1985; Thurow 1993). Studies of pelagic stocks, for example, have shown that fishing has undoubtedly caused a decline in several stocks (Beverton 1990). In this study I compare the advice given by the ACFM in the late 1970's to recent advice (ICES 1991) in order to establish how our perception of the state of the fish stock has changed and how stocks have developed during a period of basically consistent management advice based on biological reference points. Against this background I discuss constraints on the management of fishery resources and the usefulness of biological reference points in management advice. In particular, I look at the feasibility of actively reducing fishing mortality rates.

Material and Methods

This study is based on the reports of the ICES Liaison Committee (ICES 1978) and its successor, the ACFM (ICES 1979–1991), which are a source of information on all stocks for which the ICES has given recommendations. The recommendations were divided into six classes according to the need for a reduction in mortality rates as follows:

1. Unknown status. Stocks for which there is essentially no information other than (unreliable) catch statistics.

2. Status-quo recommendations. Stocks for which incomplete information is available, but for which exploitation rates seem to be sustainable according to catch data and other information.

3. Recommendations to keep the present fishing mortality rates. Stock about which there are sufficient data to enable fishing mortality rates and of stock sizes to be estimated. Present fishing mortality rates appear to be sustainable.

4. Recommendations to reduce fishing mortality rates. Stocks about which there are sufficient data to enable fishing mortality rates and stock sizes to be estimated. Present fishing mortality rates are usually higher than the reference points (except F_{high}), and the stock could benefit from a reduction in fishing mortality rates by, say, 10 to 30 %.

5. Recommendations to substantially reduce fishing mortality rates. Stocks about which there are sufficient data enable fishing mortality rates and stock sizes to be estimated. Present fishing mortality rates are considerably higher than the reference points (except possibly F_{high}). Exploitation is considered highly unsatisfactory, and the stock would gain considerably from a lower fishing mortality rate.

6. Recommendations to stop fishing. Stocks in a poor state with low recruitment. Biologically prudent management would prohibit all fishing, but recommendations are usually formulated to discourage targeted fishing.

These categories are similar to those for stocks used by the ACFM until 1990 (ICES 1991), but here I focus on the class of the recommendation as formulated by the ACFM and not on the state of the stock, which may be more controversial. The reference values for the fishing mortality rates are generally F_{max} and $F_{0.1}$.

The relationships between the recommended TAC, the agreed TAC, estimated catches and the assumed fishing mortality rates are compared for North Sea cod and haddock. A similar analysis was made by de Wilde (in press), except that he converted the recommendations into effort indices and used status-quo estimates. The following takes account of the strong recommendations to reduce fishing mortalities, and uses $0.8 \times$(current F) unless some other level has been clearly specified. For 1991 the level was $0.7 \times$(recent average of F) in line with the recommendation (ICES 1991). The stocks discussed were selected because the conditions of the fishery were such that the ACFM stopped issuing normal TAC recommendations in 1990 (ICES 1991).

I have also investigated the fate of the mesh size recommendations for *Nephrops* (ICES 1978, ICES 1991), as these too were largely based on yield per recruit and F_{max} concepts.

Results

Changes to Recommendations

An overview of the recommendations for 1979 and 1991 suggests that fisheries management in the 1980's was unsuccessful in terms of the biological reference points adopted by the ACFM. For 18 of the stocks assessed in 1978 the recommendations to reduce fishing were tightened in 1990 (Fig. 1).

An increase in the fishing mortality rates was not the only reason for tightening the recommendations, although it was the most common reason (Table 1). Altogether 14 stocks have improved in line with the recommendations. None of the stocks for which the recommendations in 1978 implied a considerable reduction in the F level have reached a state of accepted sustainable exploitation.

For nine stocks the same basically pessimistic recommendations were tabled in 1978 and in 1990. North Sea haddock and cod are typical examples of this group.

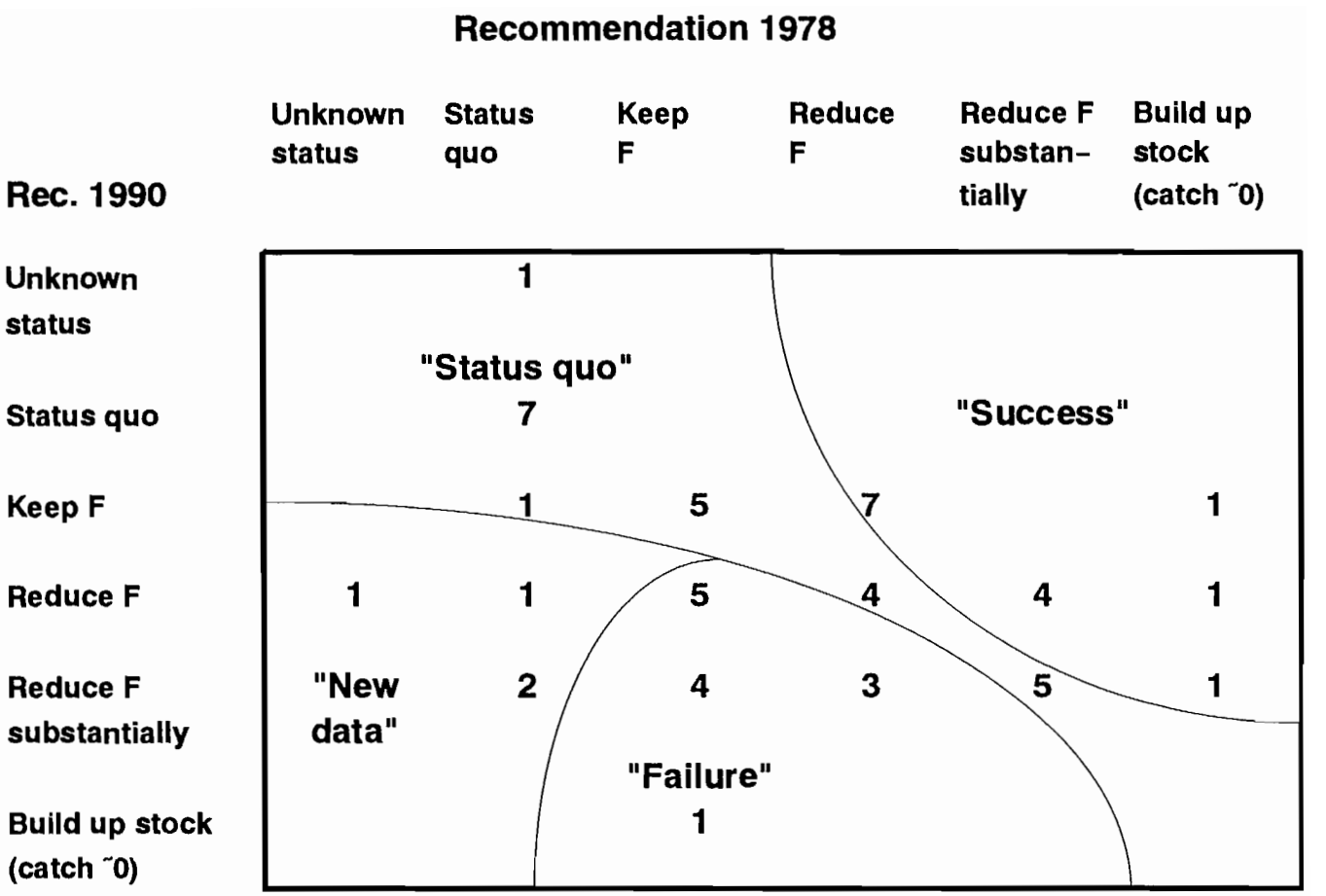


Fig. 1. The distribution of recommendations for stocks dealt with by the ACFM in 1978 and 1990. Management advice was issued for 54 stocks in all.

Altogether half of the stocks assessed in 1978 and 1990 were considered to require active management. Of the stocks assessed in 1990 but not in 1978 approximately half belong to the category for which little information is available. Substantial reductions in fishing mortality rates have been recommended in less than 10% of the cases.

Relationship Between Recommendations, Catches and F Levels for North Sea Cod and Haddock

Since the mid 1970's, the ICES has recommended a reduction in the rates of fishing mortality for North Sea cod and haddock. Fishing mortality rates have increased since the 1970's, and have remained on a relatively constant high level since the early 1980's (Fig. 2). In recent years, estimated catches, recommended catches and agreed TAC's have been fairly close. However, despite this agreement between the control and the monitoring variables, differences between recommended F and observed F levels have remained substantial (Fig. 2).

Recommendations for Mesh Changes in *Nephrops* Fisheries

Most of the recommendations for mesh changes have been accepted, although some have had to be reassessed (Table 2). As a result, mesh sizes have now been largely standardized over large areas. Apart from this practical success and the probable reduction of bycatches, it is difficult to judge whether long-term gains have materialized or not. For example catches have increased in subarea VIIa, but we do not know whether the increase is due to improved recruitment or to the larger mesh size.

Discussion

A historical analysis of F management through TAC's raises several questions. One key question concerns the true objectives of management. As already noted by the ACFM in 1981 (ICES 1982), it is easier to formulate advice when objectives are known. Knowledge of the objectives is also useful in judging the relationship between recommended and accepted TAC's. Other issues are the availability of control variables, the general ability to control the fishing mortality rates and the ability to detect changes in the central reference variables.

There is no doubt that the reference point F_{max} has played an important role in the analysis of the status of fisheries and

Table 1. The main reasons for increasing the stringency of fish stock recommendations between 1978 and 1990.

From status quo recommendation to recommendation to substantially reduce F.	
Greenland halibut V and XIV	Conservative estimates of fishing in 1978, F has increased, knowledge of stock status has improved.
Irish sea whiting	F has increased, high discard rates, TAC and catches have been kept high despite decline in recruitment, knowledge of stock status has improved.
From recommendation to keep F at present level to recommendation to reduce or substantially reduce F.	
Icelandic saithe	Catches and F have increased despite advice to the contrary.
Cod IIIa	Assessment uncertain in 1978, information on stock status has improved.
Sole VIIe	F has increased, recommended TAC < catches.
Plaice VIId,e	Bycatch species, reduction necessary because of sole VIIe
Herring SD 22–24	Perception of stock status approximately unchanged, recommendation stronger.
Sebastes mentella I and II	Increase in F, targeted fishery has increased.
Greenland halibut I and II	Increase in F, poor exploitation pattern, in 1978 uncertain signs of increasing F.
Faroe saithe	F has increased, catches have increased or remained constant despite advice to the contrary.
Western mackerel	F has increased, catches have exceeded recommended TAC's.
From recommendation to reduce F to recommendation to reduce F substantially or to avoid targeted fishery.	
Faroe cod	F has increased despite continuous advice to the contrary.
Cod SD 22–24	Continued high exploitation rates $\gg$ recommended, recommendation made stronger.
Cod SD 25–32	Recruitment has declined, F has remained high.
North East Arctic haddock	Assessment uncertain but recruitment known to be very poor.

in formulating the advice given to management bodies. Reference point $F_{0.1}$ has often been referred to and found useful, although its arbitrary nature has been stressed by the ACFM on several occasions (e.g., ICES 1986, p. 137).

Although management bodies have obviously been keen to have information about the reference points and the recommendations made because of them, the track records of managed stocks suggest that other objectives have been more important in many cases (Fig. 1). One of these is simply to reach an agreement (Lassen 1990). The importance of reaching an agreement lies in the need for cooperation and negotiation over distributional issues. Conditions in the Baltic Sea are a case in point. Despite the objective of attaining "optimum yield" (ICES 1974), the International Baltic Sea Fishery Commission has agreed on TAC's which have exceeded not only the recommended TAC's but also the actual catches. Cod stocks are currently in a poor state (ICES 1991). The TAC's agreed for 1992 imply an increase in the fishing mortality rates. Similar discrepancies have emerged in relation to other biologically based objectives such as safeguarding certain stock components. Despite the explicit objective to safeguard small natural salmon stocks, the hidden objective of maximizing catches may overrule management advice based on the stated objective (Hildén 1990).

Other important objectives which tend to overrule biological considerations are those related to short-term economic considerations. In many of the cases in Fig. 1 and Table 1, fishing mortality has increased because the fishery has attempted to keep catch levels unchanged even though recruitment has declined. It is also worth noting that one of the few real (although temporary) decreases in the fishing mortality rates of North Sea haddock occurred in 1981–1982, when marketing problems led to the introduction of voluntary boat quotas in the Scottish fishery (ICES 1983, p. 410).

Because short-term economic issues are important, it is questionable whether F_{max} or $F_{0.1}$ can be advocated by referring to long-term benefits. Especially for stocks for which the results of detailed monitoring are lacking, it is difficult to demonstrate benefits even retrospectively, as shown by the case of *Nephrops*.

In the case of collapsed stocks, it is generally simpler to follow biologically based advice, because in such cases everybody loses equally. The objectives of rebuilding stocks, pelagic stocks in particular, have therefore been achieved in several cases (Fig. 1; Corten 1990). The difficulties begin when a substantial recovery has occurred: should rebuilding be continued until some historical level is reached or is some lower level acceptable? In some herring stocks, recovery has been slowed down substantially by the development of a new fishery.

The problems involved in the management of rebuilding stocks is closely related to those involved in managing developing fisheries in general. When a fishery becomes viable in economic terms, the rate of investment and the increase of effort are usually high. If management advice and practical measures are mainly based on biological information describing the state of the stock, management is almost doomed to fail. As fisheries expand, our knowledge of the state of the "newly discovered" stocks accumulates too slowly to allow intervention based on precise biological information. Consequently, fishing mortality rates can reach high levels before proper advice on the adjustment of fishing mortalities relative to reference points can be produced. This is shown by several

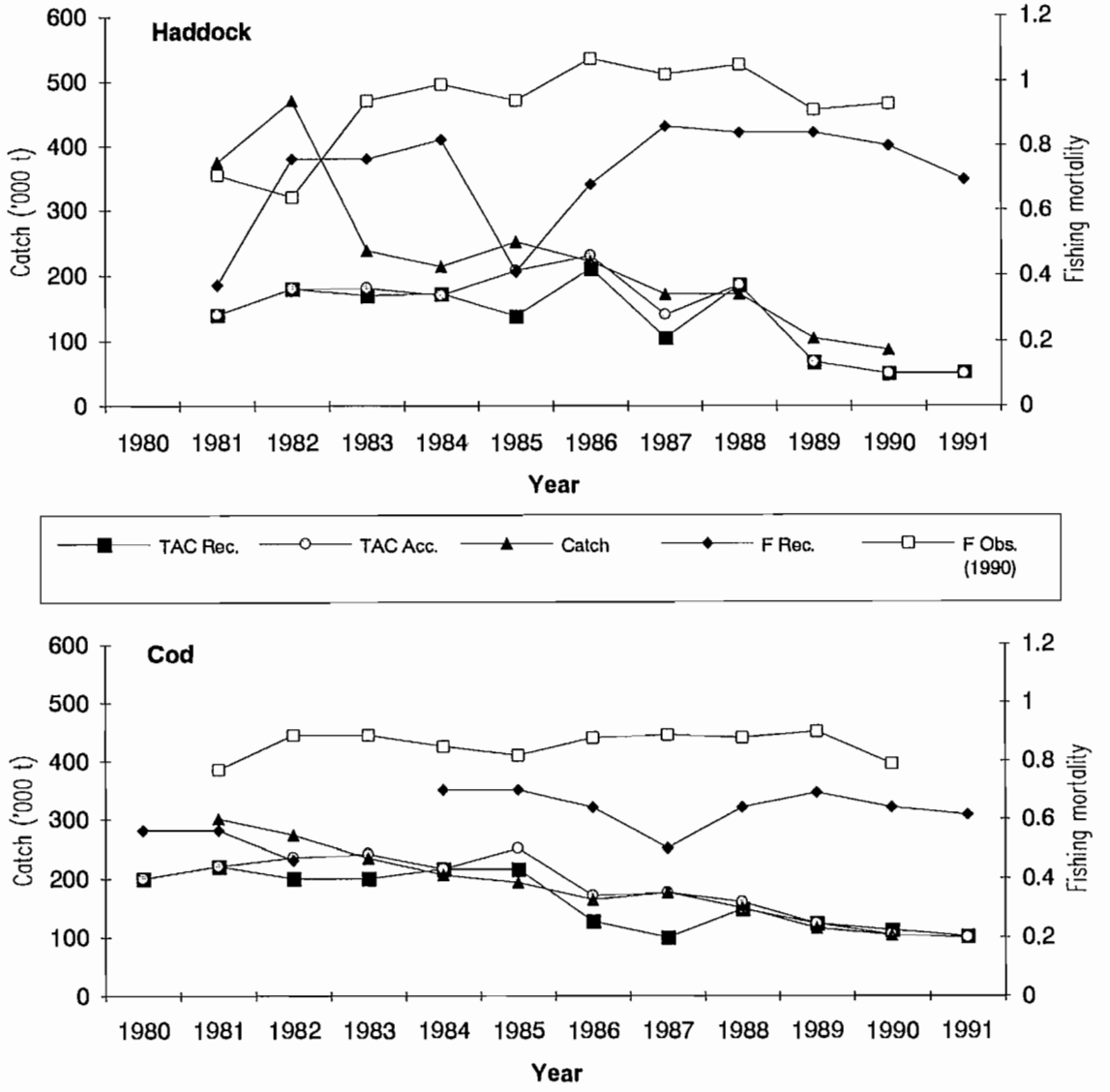


Fig. 2. Time series of management advice and management measures, and their effects on fishing mortality rates of North Sea haddock and cod in the 1980's.

of the stocks for which recommendations to reduce fishing mortalities were issued in 1991 (Table 1). In 1978, many of these stocks were relatively less exploited, but fisheries have since expanded without decisive management intervention.

The lack of information is not easily remedied for developing fisheries. Several studies have demonstrated the unreliability of crucial monitoring results, even for extensively fished and investigated stocks (e.g., Brander 1987; Weber 1989; Sjöstrand 1989; Anon. 1991; Thurow 1993). Reference points such as F_{max} or $F_{0.1}$ do not therefore appear to be very useful operational objectives for developing fisheries. Observations of the rapid expansion of developing fisheries suggest that it is more important to monitor and control directly investments and decisions to enter the fishery. If management measures are delayed until biological data unambiguously justify them, action will seldom be taken before problems have arisen. This is particularly true when the data available from sources other than the fisheries themselves are sparse. Data on catch per unit of effort, for example, may give a false impression of changes in the stocks (e.g., Crecco and Overholtz 1990). If biological factors are used in the initial stages of a developing fishery, they should be in the form of simple rules of thumb for cautious adaptive exploitation. These rules could be related to the rate of increased effort relative to the generation time of the fish species to be exploited.

The problem of the availability of information and control variables is also evident in heavily exploited stocks (Fig. 2). Fishing mortality rates are undeniably very high, and the message that there is a need to reduce mortality rates if stocks are to be stabilized is clear. It has even been accepted by management bodies as shown by the consistency between agreed TAC's and recommended TAC's. Catch data indicate that management has been successful, but the monitoring of fishing mortality rates shows that TAC's have been inefficient as

Table 2. Mesh recommendations for *Nephrops* from 1978 to 1990. All mesh sizes in millimeters, gains in 1978 as increase in long-term yield in per cent. Ranges are based on different selective properties and differences between male and female *Nephrops*. Gain for 1990 is based on a 10-mm increase on the mesh size used in 1990. + = significant long-term gain; 0 = little effect; - = long-term loss. Variability within areas due to separate assessments for males and females.

	1978			1990		
	Mesh size			Mesh size		
Area	In force	Suggested	Gain	In force	In use	Gain
Iceland	80	90	0–9	80	80	0
Denmark IIIa	35	70	0–13	70	70	?
Sweden IIIa	60	70	−2–5	70	70	?
Ireland VIIa	50	70	5–64	70	70	+0
NE England IV	70	80	−1–12	70	70-80	+0
N Ireland VIIa	50	70	−2–70	70	65	+0
Spain VIIg,k	60	80	2–20	70	74-80	0-
France VIII	39	60	2–43	55	50	+0
Spain IX	40	60	2–16	55	40-50	?

control variables (Fig. 2). This may be due to the discarding or misreporting of catches or to other problems in the assessment. In 1990, the ACFM recommended that effort should be reduced for example by cutting the number of days spent at sea (ICES 1991). However, it is not clear how this should be implemented. A uniform reduction of the days at sea for all vessels is unlikely to be economically viable because of the heterogeneity of the fleets. Implementation of a true reduction of effort may also be hindered by the present management structure, which requires joint action by a number of countries and an even larger number of fishing fleets with different arrays of political affiliation. The general ICES experience has been that — under these conditions — only a stock collapse gives a strong enough signal to the involved parties that they must do something to bring about a substantial reduction in fishing mortality. Other cases of actively reduced fishing mortalities have involved a reshaping of the management conditions and infrastructure following the acceptance of the 200-mile exclusive economic zones (e.g., Pinhorn and Halliday 1990; Stocker and Leaman 1990; Skarphedinsson 1991). Such changes do not, however, guarantee a reduction in fishing mortalities as shown by the New England groundfish fishery in the USA (Anthony 1990).

The preceding discussion supports the conclusion of the NAFO special session on management under uncertainties (Chairman's report, NAFO Scientific Council, unpublished) that management, and hence management advice, based on F_{max} or $F_{0.1}$ is difficult to implement. This suggests that management advice should not focus on identifying the difference between the present state of the stock and the fishery and some, possibly unattainable, objectives defined by reference points. It is more important to analyse in detail the short and intermediate term consequences of events or actions. The ACFM reports have frequently contained statements such as "advantage should be taken of strong year classes" or "TAC's should be lower if recruitment is poorer than estimated", but few, if any, quantitative analyses of these events have been presented in the reports, although the calculations are simple and straightforward. One reason for the lack of such analyses is evidently the perceived overwhelming importance of the TAC recommendation and immediate catch forecasts used in quota negotiations. The potential of TAC advice to ensure sustainable use of a resource may, however, be rather limited (Fig. 1).

One way of increasing the emphasis on sustainability and tactical and strategic decisions is to use various forms of risk analysis (e.g., Francis 1990; Kuikka and Varis 1991). Reference points such as those developed by Serebryakov (1990) also emphasize the probability of a particular sequence of events affecting the development of a stock. These approaches may help fisheries managers by highlighting stock-related problems more clearly. They do not, however, necessarily result in managements decisions which are better than the present ones from a biological point of view. Like the reference points F_{max} and $F_{0.1}$, risk analyses focusing on the fate of fish stocks are most useful for rational sole owners of resources. Fisheries management and the exploitation of fishery resources are a social activity. Analyses of social risk management (e.g., Merkhofer 1987) demonstrate that subjective preferences, the problems of collective decision-making and value perceptions dominate these issues. The risky business of managing fisheries is unlikely to be any different. Therefore, methods for analysing different options, connections and trade-offs between fleets and fisheries are likely to become increasingly important. The analysis of risks has to expand accordingly.

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Spawning Stock Biomass per Recruit in Fisheries Management: Foundation and Current Use

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Goodyear, C. P. 1993. Spawning stock biomass per recruit in fisheries management: foundation and current use. p. 67–81. *In* S. J. Smith, J. J. Hunt and D. Rivard [ed.] Risk evaluation and biological reference points for fisheries management. Can. Spec. Publ. Fish. Aquat. Sci. 120.

Spawning stock biomass per recruit (SSBR) estimates the expected lifetime reproductive potential of an average recruit (*P*), which is an important correlate of population growth potential. The ratio of the fished to unfished magnitude of *P* is the spawning potential ratio (SPR) and is a measure of the impact of fishing on the potential productivity of a stock. Current use of SPR merges concepts developed to quantify the compensation required for population persistence given anthropogenic increases in mortality with observations of stock productivity and SSBR for fisheries in the western North Atlantic. It has a firm theoretical basis and is evaluated against yield per recruit and contrasted with other traditional biological reference points. SPR is widely used in U.S. fisheries managed under the Magnuson Fishery Conservation and Management Act, usually in the context of a percentage of the unfished SSBR. Its implementation is intended to be risk-aversive through selection of minimum acceptable levels above which stocks maintain acceptable productivity. The behavior of the underlying principles suggests SPR values below about 0.2 should be avoided unless there is evidence for exceptionally strong density-dependence in the stock. Critical levels have typically been set in the range of 0.2 to 0.3 primarily based upon the experience in the northwest Atlantic.

La biomasse du stock reproducteur par recrue (BSR/R) évalue le potentiel reproducteur prévu pour la durée de vie d'une recrue moyenne (*P*) qui est un important corrélat du potentiel de croissance d'une population. Le rapport entre la grandeur de *P* pour un groupe exploité et un groupe non exploité est le rapport du potentiel reproducteur et une mesure de l'effet de la pêche sur la productivité potentielle d'un stock. L'utilisation actuelle du rapport du potentiel reproducteur combine des concepts élaborés pour mesurer la compensation nécessaire à la persistance d'une population étant donné les augmentations anthropogéniques de la mortalité aux observations de la productivité du stock et de la biomasse du stock reproducteur par recrue pour les pêches de l'Atlantique nord-ouest. Ce concept a une base théorique solide et il est évalué par rapport au rendement par recrue et est mis en contraste avec d'autres points de référence biologiques classiques. Le rapport du potentiel reproducteur est largement utilisé aux États-Unis pour les pêches qui sont gérées en vertu de la Magnuson Fishery Conservation and Management Act, en général dans le contexte d'un pourcentage de la biomasse du stock reproducteur par recrue qui n'est pas exploitée. Son application est destinée à limiter les risques en choisissant des taux minimum acceptables au-dessus desquels les stocks conservent une productivité acceptable. Le comportement des principes sous-jacents porte à croire qu'il faut éviter que le rapport du potentiel reproducteur soit inférieur à 0,2 environ à moins que l'on démontre que le stock est exceptionnellement très dépendant de la densité. Des taux critiques ont en général été établis dans la plage de 0,2 à 0,3 fondés surtout sur l'expérience dans l'Atlantique nord-ouest.

The fundamental question in formulating advice to managers on harvest levels that can produce desired catches over a long-term period concerns the level of the reproductive capacity of the stock sufficient to maintain levels of high productivity. As evidenced by the classic works of Ricker (1954) and Beverton and Holt (1957), such stock-recruitment considerations have long been a matter of concern for fishery scientists. However, in practice, stock-recruitment data for most stocks display a wide scatter of points that belie underlying functional dependencies. This scatter often defeats statistical demonstration of the relation of recruitment to stock size and has not generally been convincing to managers. For these reasons, spawning stock size has not generally been an explicit consideration in developing advice for total allowable catches.

The problem of evaluating the dependence of recruitment on stock size has not been the sole province of those trying to grapple with safe harvest levels. Some industrial practices, notably the temporary withdrawal of condenser cooling water by poorly located steam-electric power plants, can result in the deaths of enormous numbers of larval and juvenile fishes and have initiated considerable scientific debate (e.g., Barnthouse et al. 1988). Measures to minimize these losses could in some instances impose direct economic costs on the order of billions of dollars.

With the implementation of the U.S. National Environmental Policy Act of 1969 (U.S. Congress 1970), scientists were required to assess the impacts of such losses on the affected populations and the fisheries and ecosystems they supported. It became immediately clear that the key questions involved the nature and force of the density-dependent processes that may be available to offset the increases in mortality; i.e., the relation between stock size and subsequent recruitment. The inadequacy of existing knowledge prompted

research by both government and industry to develop methods to quantify the population level consequences of such losses. This research led to an initial conference in Gatlinburg, Tennessee (Van Winkle 1977) where several authors dealt directly with the issue (Campbell et al. 1977; Christensen et al. 1977; Goodyear 1977; Lawler et al. 1977; McFadden 1977). Subsequent studies included reviews of compensatory phenomena (Saila et al. 1987; Jude et al. 1987) and of mathematical models that may be applied to the problem (Chen 1987). The research continues to receive significant support from the COMPMECH program of the Electric Power Research Institute, Palo Alto, California.

One of the studies at the initial conference presented a method to quantify the degree to which anthropogenic mortality required compensatory changes in natural mortality or fecundity for population persistence (Goodyear 1977). The method utilized the observation that population stability requires that mortality be exactly offset by reproduction (on average). This fact was exploited to develop an estimator for compensation (the Compensation Ratio, CR) based on changes in the reproductive potential of the average recruit (P). This knowledge permitted exploration of several important stock-recruitment considerations without the need to know the form of the curve or its parameters (Goodyear 1977, 1980). Although proposed as a potentially useful fisheries management tool in the original paper, implementation required empirical estimates for critical levels against which to judge the significance of any particular level of CR. Such estimates were not then available.

Meanwhile, expansion of the fisheries off the U.S. coast of the Northwest Atlantic in the late 1960's and early 1970's precipitated declines in landings of several important species (Brown 1990). This spurred increased emphasis on evaluation and advice on the condition of the spawning stocks (e.g., Shepherd 1982; Anon. 1983, 1987, 1988abc, 1989abc; Sissenwine and Shepherd 1987; Brown 1990). Several of these studies noted the relation between productivity and SSBR (Shepherd 1982; Overholtz et al. 1986; Mathews 1991), and others observed declines in productivity when SSBR fell below a general range of 20–40% of the unfished level (Gabriel 1985; Gabriel et al. 1984, 1989; Anon. 1987).

Appropriately estimated, SSBR is equivalent to Goodyear's (1977) definition of potential fecundity per recruit (P); consequently, the observations of declining productivity with declining SSBR provide estimates of the critical levels. These developments and an urgent need to establish quantifiable conservation objectives to rebuild depleted Gulf of Mexico red drum (*Sciaenops ocellatus*) and red snapper (*Lutjanus campechanus*) stocks led to additional evaluations of the underlying theory, much of which was incorporated into stock assessments or other management advice (e.g., Goodyear 1987, 1990; Goodyear and Phares 1990). This paper summarizes the results of this work.

The Spawning Potential Ratio

Basis

Goodyear's (1977) compensation ratio (CR) is an index of the degree of compensation required of a fished population to persist. It is a measure of the overall change in the survival and fecundity parameters that must occur for a population undergoing exploitation to stabilize at a new equilibrium. The development of the CR details the assumptions involved in such analyses and is condensed here for convenience. The CR is defined as:

$$\mathrm{CR} = \frac{P_{\text{unfished}}}{P_{\text{fished}}}$$

P_{unfished} = potential recruit fecundity in the absence of fishing mortality;

P_{fished} = potential recruit fecundity in the exploited stock.

Potential recruit fecundity (P) is defined as the number of eggs that could be produced by an average recruit in the absence of density dependence. It carries the designation "potential" to highlight the notion that it includes both the actual average lifetime production of eggs per recruit at equilibrium population densities plus those eggs that would have been produced by an average recruit in the absence of any density-dependent suppression of maturities or fecundities at age or of survival in post-recruit ages. It is determined as:

$$P = \sum_{i=1}^{n} E_i \sum_{j=0}^{i-1} S_{ij}$$

where:

n = number of ages in the unfished population;

E_i = mean fecundity of females of age i in the absence of density-dependent inhibitions of growth;

S_{ij} = $\exp\left(-\left(F_{ij} + M_{ij}\right)\right)$, density-independent annual survival probabilities of females of age i while they were age j;

F_{ij} = the fishing mortality rate of females of age i when they were age j; and

M_{ij} = the natural mortality rate of females of age i when they were age j.

The convention of employing F_{ij} in the computation of P allows the estimate of P to reflect the historical pattern of fishing mortality on the year classes comprising the population. For equilibrium analyses all F_{ij} would equal F_{ij} for all j, and the equations simplify to referencing only the age dimension. The compensatory density dependence that brings about population stability can be manifested as changes in early life stage (pre-recruit) survival, or as changes in survival or fecundity in the later life stages. It is probably impossible to forecast the

partitioning of each possible compensatory response between increased survival and fecundity by life stage for a virgin stock that is to be the target of a new fishery. However, in a heavily exploited stock the reduced density of older fish often relaxes any density-dependent suppression of growth, survival, and fecundity that may have existed in these older age classes in the virgin population. As a consequence, the parameter values needed to solve for P can often be derived from studies of an existing stock if exploitation has been sufficient to fully utilize any potential density-dependent growth and survival that might occur within the adults. Specifically, no further reduction in abundance would cause the recruits to be subject to less natural mortality, grow faster, be more fecund, or mature earlier. It is assumed that half the eggs will be female. For most applications this assumption has little importance since the initial fraction of females will cancel when ratios are taken. However, this assumption does pose a problem for those species that transform from one sex to the other during their lifetime.

Recall that for population persistence each egg in the parental generation must on average produce a single egg in the filial generation. Thus, if a population can persist in both the fished and unfished state, the ratio of the average potential lifetime fecundities of pre-reproductive recruits of the same age is a direct measure of the compensation that occurred between the two states to allow persistence in the fished state; hence the term compensation ratio. The response of the CR to fishing mortality is illustrated in Fig. 1. Its actual shape will vary by species. A CR value of 100 means that in order for the population to persist at the corresponding level of fishing mortality the mean survival probability of an average recruit must increase a hundredfold over that existing in the unfished condition.

The calculation of spawning stock biomass per recruit for a fished stock is analogous to the computation of P, except that the units may be in terms of the biomass of the entire adult stock, the biomass of mature females in the stock or the biomass of the eggs they produce. SSBR calculations are typically made to characterize the reproductive potential of a stock. When it is estimated in terms of the biomass of the eggs, it differs from P only by the mean weight of the eggs. Note that this term would cancel when the compensation ratio is taken. Given this observation, it is clear that for practical purposes SSBR and P are effectively identical. These two terms are henceforth treated synonymously with the understanding that the estimates of SSBR that do not conform to the precise definition of P are intended as surrogates for the estimate of potential recruit fecundity. The spawning potential ratio (SPR) is then estimated as the reciprocal of CR; i.e.,

$$\mathrm{SPR} = \frac{P_{\text{fished}}}{P_{\text{unfished}}} = \frac{\mathrm{SSBR}_{\text{fished}}}{\mathrm{SSBR}_{\text{unfished}}}.$$

SPR has a maximum value of unity and declines toward zero as fishing mortality increases. As the SPR declines with

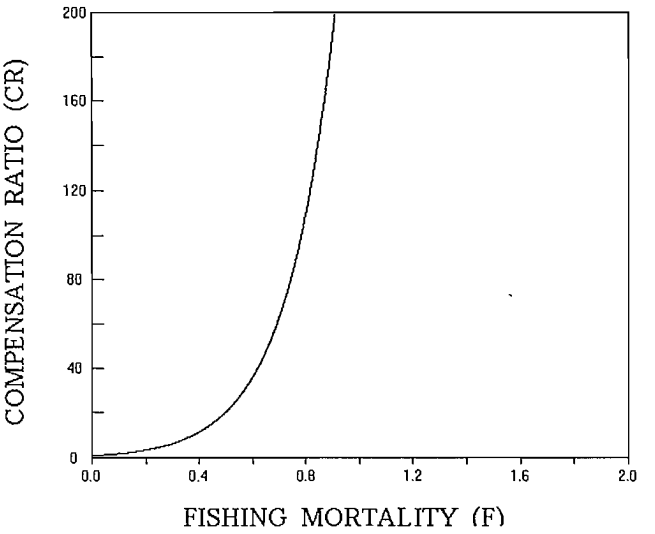


Fig. 1. Effect of fishing on the level of compensation required for population persistence in striped bass (*Morone saxatilis*) as a function of fishing mortality (modified from Goodyear 1977).

increased fishing mortality, the required compensatory decline in density-dependent mortality or increased fecundity per spawner increases.

The spawning potential ratio is an alternative measure of the reproductive fitness of a stock that is conceptually based on the degree of departure from the virgin condition. It is a measure of the extent to which fishing has reduced the potential reproductive output of an average recruit from that which would have existed in the absence of fishing. This measure does not require knowledge of the size of the virgin stock. It is based on the premise that the size of a stock is the emergent property of the force exerted by the growth potential of its excess reproductive capacity against the limitations imposed by the environment, including fishing. This approach attacks the stock-recruit question from a different perspective than the traditional statistical analysis of stock-recruit data pairs. The often observed statistical independence of stock and recruitment and the required functional dependence of recruitment on stock are not conflicting concepts, but rather are each valid attributes of the recruitment problem. The observations of stock and recruitment focus on the realized production of recruitment and are confounded by environmental variation and measurement error.

SPR on the other hand focuses on the influence of fishing mortality on the slope of the spawner-recruit curve at its origin; i.e., in the absence of compensatory density dependence. In this situation the initial slope of the stock-recruit relation is proportional to the average lifetime fecundity of the recruits (Goodyear 1980). For example consider the hypothetical population of Fig. 2 which is near the origin of a stock recruitment curve. In this example, fishing mortality is scaled so that it reduces the expected lifetime fecundity of the offspring to 1/3 of that in the unfished stock. In the fished state the cluster of parental stock sizes and environmental influences that produced the recruits depicted by the solid circles would produce the cluster of recruits represented by the "×" symbols. A line

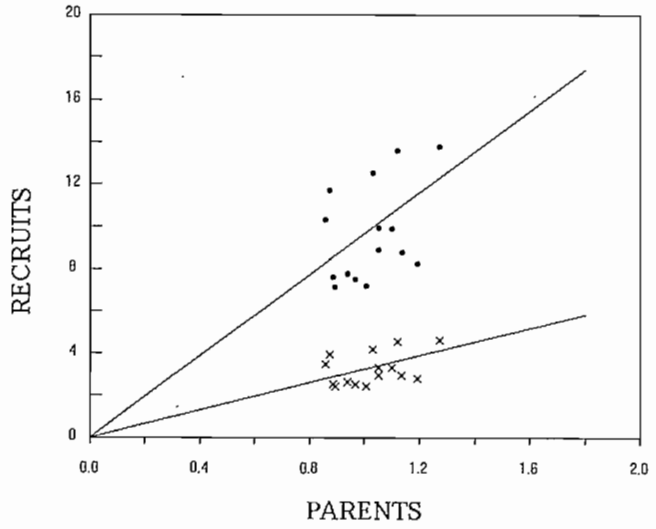


Fig. 2. Hypothetical stock-recruit data in the absence of density dependence with (×) and without fishing (·) plotted with corresponding regressions through the origin.

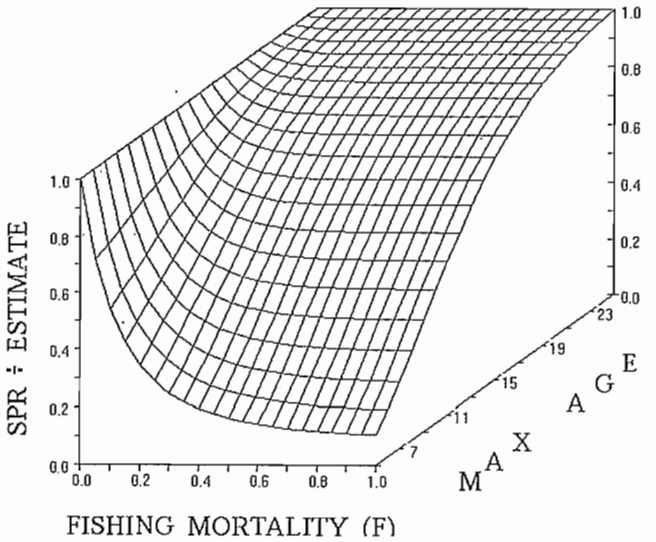


Fig. 3. Bias in SPR estimates from employing maximum ages too young to represent the unfished age distribution.

drawn through the mean of the unfished cluster has a slope of 10, and a line drawn through the mean of the fished cluster has been reduced by fishing to a slope of 3.3. Note that in this example recruitment is in excess of that needed for replacement for both situations, thus neither situation would necessarily lead to extinction. However, the growth rate for the unfished stock is greater than for the fished stock since it produces three times more mature offspring per spawner than does the fished stock. Further, the force of this differential in growth potential on the environmental resources might cause the two situations to reach even much different average equilibrium stock sizes with different temporal dynamics. The impact of fishing on the potential growth rate of a population is therefore different from its effect on the average population size, but it is also inexorably tied to it by ecological processes.

The basic concept involved in the application of SPR stems from the observation that significant changes in vital rates must occur on a fished stock, if it is to persist. The average parental egg in a population must produce an egg in the filial generation for population persistence. Fishing reduces the probability that individuals will survive to each age subsequent to entry into the fishery. If growth and natural mortality remain constant at the levels of the unfished stock, the number of eggs produced by the filial generation would always be fewer than their parents. As a consequence, to prevent population collapse the egg to pre-recruit survival probability and/or fecundity at age must rise in response to the lowered expected lifespan of the recruits. In the interest of brevity, the following discussion will assume that compensatory changes are manifested only as changes in early life stage survival probabilities. This simplifies the discussion but does not diminish the applicability of the basic concept.

This compensatory rise in the egg-to-pre-recruit survival probability is itself an emergent property of the dynamics of whatever actually happens during the growth and development from egg to recruit. However, it must occur for a population to sustain itself. Recruitment may or may not decline with increasing fishing, but this component of population survival must increase. The concept is really quite simple. If an average pre-reproductive recruit produced one-tenth as many eggs in the fished versus unfished population, then the survival probability of the eggs must increase tenfold to permit the population to persist in the fished state.

Stability is a relative concept here. It does not imply that recruitment in the population is the same from year to year, either in the fished or unfished state. The concept is that if the fishery can persist indefinitely at a given level of fishing mortality, then there is a corresponding long-term average egg-to-recruit survival that meets the replacement requirement. If no compensatory change in the mean survival rate were to occur for the preceding example, recruitment would fall 90% each generation from what it would otherwise have been. After two generations recruitment would be only 1% of its unfished state with the same environmental history.

The examples in the subsequent discussion that require data on the life history characteristics of a species use data from the red snapper fishery of the U.S. waters of the Gulf of Mexico (Goodyear and Phares 1990). Computations adopt a convention of 30 age classes in the stock because the sum of the contribution of all older ages to egg production is less than 2.5% for the unfished population. Natural mortality (M) is 0.20 for all ages, and a length-fecundity relation is used to estimate the age-specific reproductive potentials of the average recruit at age from mean length at age. Forecasts of future trends, though arbitrary for the purposes of this document, assume initial conditions based upon VPA analysis of fishing mortality and numbers at age for this stock.

The last age class included in the analysis (n) should be old enough so that the sum of the reproductive contribution of all older age classes represents a vanishingly small part of the reproductive potential of the unfished age structure. Selection of a younger maximum age, perhaps based upon the

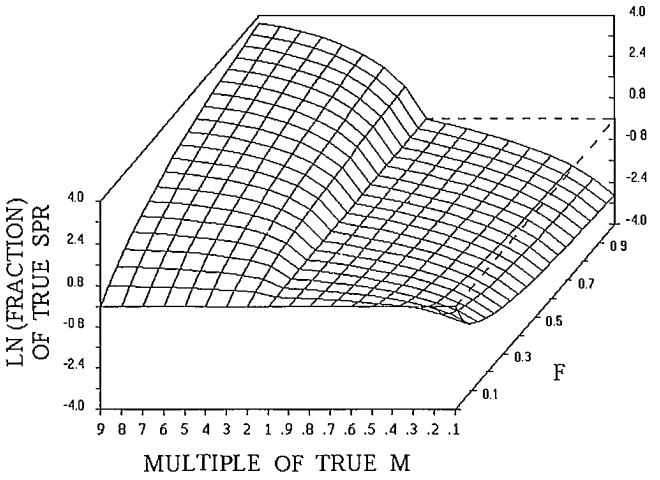


Fig. 4. Bias in SPR estimates from employing erroneous estimates of natural mortality (M).

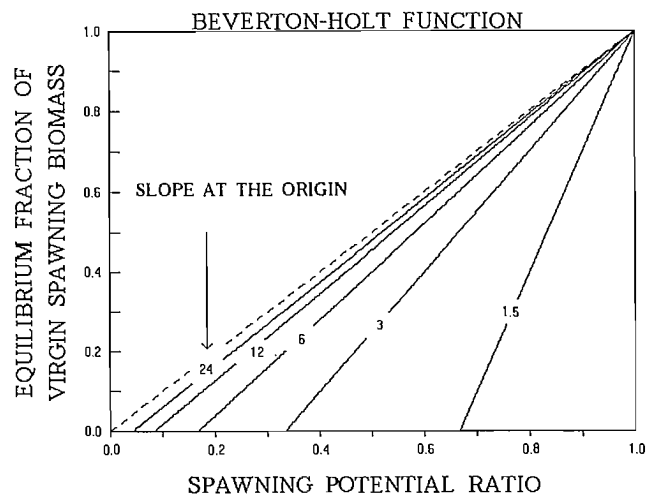


Fig. 5. Equilibrium fraction of virgin spawning biomass as a function of SPR for several slopes at the origin for the Beverton-Holt spawner-recruit model.

maximum age observed in a sample from a fished population, will introduce a positive bias into the estimate of SPR. The extent of this bias as a function of fishing mortality and the oldest age included in the estimate of SPR is illustrated for red snapper in Fig. 3. This sensitivity to the oldest age class included in the analysis is the result of the fact that both the cumulative effect of fishing mortality on cohort survival and the reproductive importance of each individual increases with age. As a consequence, omission of the older ages can cause significant bias in the estimation of SPR. The oldest age in a population is a function of the initial cohort size and survivorship. In the unfished population, survivorship is solely a function of natural mortality. As a consequence, error in the estimate of natural mortality can cause significant bias in the estimate of SPR, particularly at high levels of fishing mortality (Fig. 4).

SPR and the Equilibrium Spawning Biomass

As already noted, SPR is a measure of the depression of the potential growth rate of a population caused by exogenous mortality. It is not directly tied to population size. However, if environmental perturbations lack temporal trends then a particular average stock size would be expected for any given constant level of SPR. The particular association between SPR and average stock size will be dependent upon the nature of the density-dependent suppression of population growth. Thus it would be both stock and area specific.

One manifestation of the anticipated relation could arise because the reduced growth potential of the fished population would cause slower recovery from negative environmental perturbations. Similarly the fished population would be slower to respond to temporary shifts toward improved environmental conditions. The overall result of these responses would be to reduce the average population size in the fished condition. Since the depression of the growth rate is measured by SPR, the mean stock size would decline with declining SPR. However, the primary factor controlling the magnitude of the response to a change in SPR is the actual change in the growth potential of the population. The magnitude of this change is the joint property of the magnitude of SPR and the slope of the spawner-recruit relation in the absence of density-dependent suppression of survival and fecundity. Thus, the expected mean population size for any given level of SPR for any particular stock-recruitment function is determined by the slope of the spawner-recruit curve at its origin and some scalar effect related to the size of the environment.

These principles can be illustrated for the traditional Ricker and Beverton-Holt stock-recruit models (Ricker 1975). The Ricker model used is of the form

$$R = \alpha P e^{\beta P}$$

and the Beverton-Holt model is of the form:

$$R = 1/(\alpha + \beta P) .$$

To facilitate the examination of the relation among the variables, the scalar effect of the environment was removed by normalizing the equilibrium stock sizes to the virgin stock. Two analyses were performed for each model. The first was to plot the equilibrium fractions of the virgin spawning biomass as a function of SPR for several levels of the slope of the spawner-recruit curve at the origin (Fig. 5 and 6). The second was to plot the equilibrium fractions of the virgin spawning biomass as a function of the slope at the origin for several levels of SPR (Fig. 7 and 8).

For both models the equilibrium spawning biomass declines monotonically with declining SPR (Fig. 5 and 6) for all slopes of the spawner-recruit curve at the origin. However, the initial rate of decline observed for any given spawner-recruit slope is less for the Ricker model (Fig. 6) than for the Beverton-Holt model (Fig. 5) and the difference is more pronounced at higher slopes at the origin. This characteristic

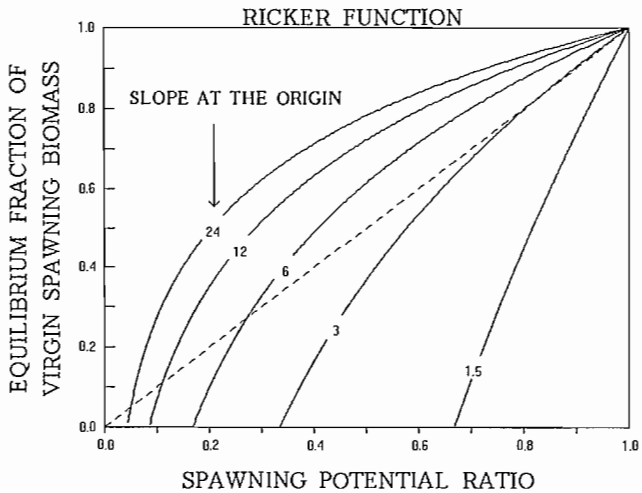


Fig. 6. Equilibrium fraction of virgin spawning biomass as a function of SPR for several slopes at the origin for the Ricker spawner-recruit model.

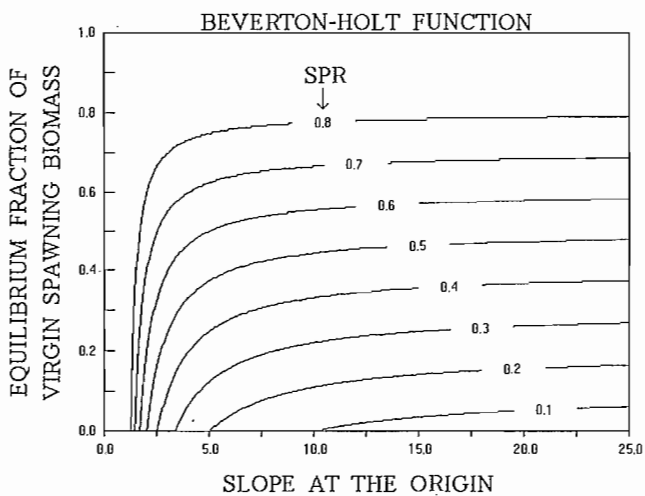


Fig. 7. Equilibrium fraction of virgin spawning biomass at several levels of SPR as a function of the slope of the Beverton-Holt spawner-recruit model at the origin.

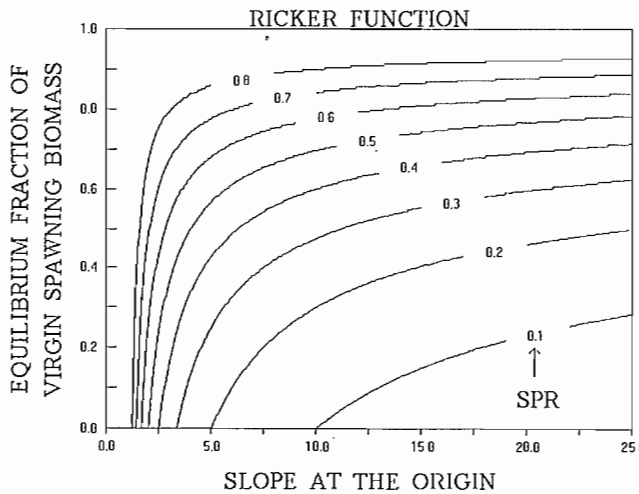


Fig. 8. Equilibrium fraction of virgin spawning biomass at several levels of SPR as a function of the slope of the Ricker spawner-recruit model at the origin.

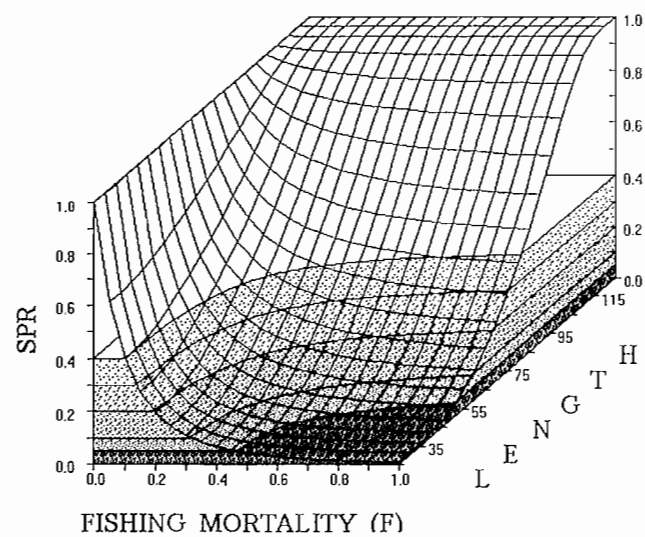


Fig. 9. SPR estimates for Gulf of Mexico red snapper as a function of fishing mortality and minimum size assuming no mortality for sublegal fish that are caught and released.

is the result of the dome-shaped form of the Ricker model. Inspection of Fig. 5–8 provides ample evidence that the equilibrium size of a stock cannot be reliably predicted from SPR where the stock-recruit relationship is not well known. The fractional reduction from the virgin condition may be more or less than the SPR depending on the form of the stock-recruitment relation and its slope at the origin. Note, however, that the Y-axis intercepts are the same for the two models in each of the figures. This is because the limiting value of SPR for population persistence for any stock-recruit relation is the reciprocal of the slope of the spawner- recruit curve at its origin.

Equilibrium Analyses

SSBR and SPR decline with increasing fishing mortality and can be evaluated using methods analogous to traditional equilibrium analyses of yield per recruit. The spawning potential ratio is plotted as a function of length at recruitment and fishing mortality in Fig. 9 assuming no incidental fishing mortality exists for fish below the minimum size.

SPR decreases monotonically with increasing fishing mortality for all size limits that permit harvest. Similarly SPR decreases with decreasing minimum size for all levels of fishing mortality above zero. If significant mortality occurs for fish caught and released below the minimum size, then some fishing mortality is exerted below the minimum size and the

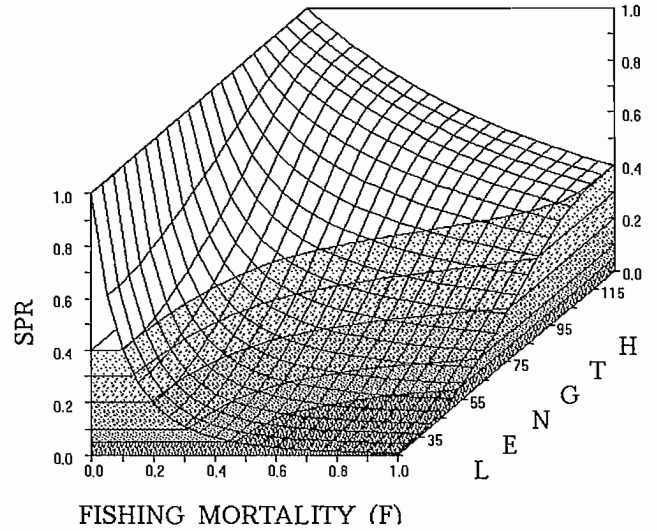


Fig. 10. SPR estimates for Gulf of Mexico red snapper as a function of fishing mortality and minimum size assuming a mortality of 10% for sublegal fish released after capture.

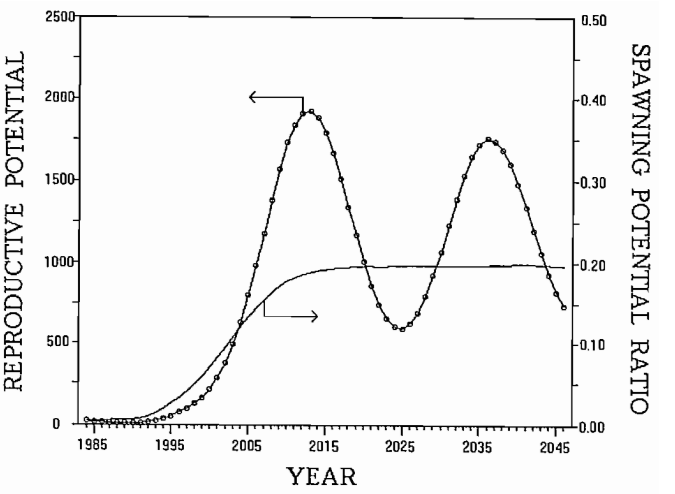


Fig. 12. Spawning stock biomass and SPR subsequent to management intervention for a depressed stock regulated by a strongly dome-shaped stock-recruit relationship.

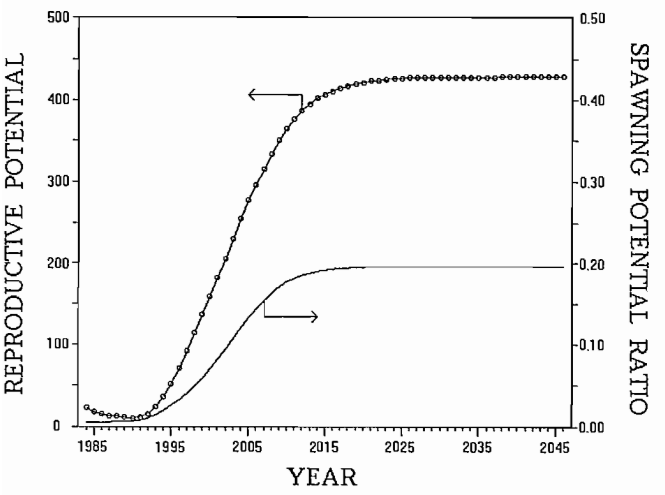


Fig. 11. Spawning stock biomass and SPR subsequent to management intervention for a depressed stock regulated by an asymptotic stock-recruit relationship.

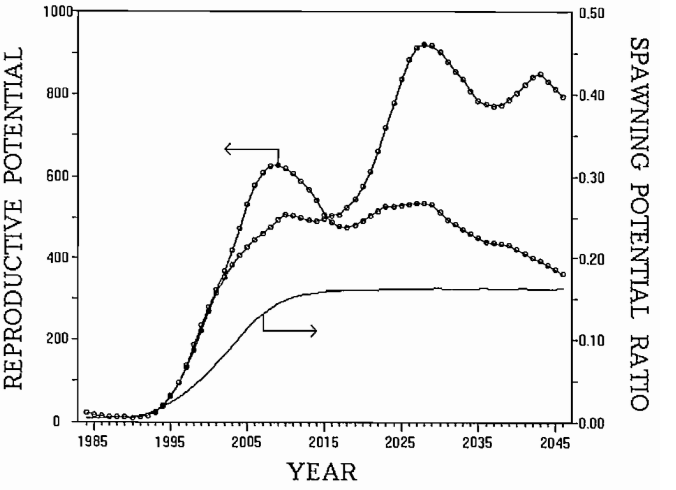


Fig. 13. Spawning stock biomass and SPR subsequent to management intervention for two cases where the stock is regulated by an asymptotic SR curve subject to significant annual environmental modification.

surface of the relationship in Fig. 9 would decline with increasing F for all length. An example is presented in Fig. 10 where sublegal fish were assumed to be caught with the same frequency as fish above the minimum size and suffered a 10% mortality as a result of the experience.

Analyses of Temporal Variation in SPR

If information on temporal trends in fishing mortality is available, then both the historical levels of SPR and its transient behavior resulting from changes in fishing mortality can be evaluated and/or forecast. Computations of past, present, and future trends in spawning stock and SPR were performed for hypothetical red snapper populations using computer simulation. Fishing mortality rates and initial cohort abundances by age were taken from values for the U.S. Gulf of Mexico stock estimated by VPA analysis for 1984–1988 (Goodyear and Phares 1990). SPR for 1984 was estimated from the age distribution of the initial year's estimates of fishing mortality assuming these values applied to all prior years. This assumption allows estimates of the initial recruitment to all year classes comprising the population. Estimates in subsequent years gradually replace these initial equilibrium values with realized recruitment for years where data are available; these are replaced by forecasts for future years.

I simulated future population trends for three arbitrary stock-recruitment options to highlight the difference between SPR and the actual spawning potential of the stock: 1) a deterministic Beverton-Holt curve (Fig. 11); 2) a deterministic

Ricker curve (Fig. 12); and 3) two stochastically driven cases with a Beverton-Holt curve where year-class strength was strongly impacted by random environmental events (Figure 13).

Fishing mortality rates for each of these scenarios were adjusted from the historic levels beginning in 1991 so that the equilibrium SPR would be about 0.2. Note that the time trends of SPR in Figures 11–13 are independent of the time trend in the spawning potential of the stock. They are also independent of the particular stock-recruitment relation postulated.

This point highlights two important characteristics of SPR. First, setting spawning stock goals using this criterion assumes that the particular critical level chosen is sufficiently high that the size of the stock will be limited in part by density-dependent growth or mortality when SPR is at or above the critical value. In such circumstances the stock will be able to rebound from environmental perturbations. Viewed from a different context, at SPR = 0.1 the slope of the stock-recruit (escapement) curve is 10% of its unfished level. Since the escapement curve must be greater than 1.0 to permit persistence, the slope of the unfished stock-recruitment curve of a stable fishery must have been greater than 10 to prevent a stock collapse and perhaps greater than 20 to accommodate the fishery.

The second consideration is a consequence of the fact that SPR does not measure the actual spawning potential of the stock. Management of the spawning potential of a stock using SPR proceeds from the conceptual notion that the fishing mortality should never be so severe that the average female recruit will be unable to replace herself (and her mate) in the next generation. To accomplish this goal, the reproductive potential of her average offspring (egg) must be limited by density-dependent reductions in survival or fecundity.

If such a population were to be depressed by some temporary environmental perturbation, the consequent reduction in the density dependence would permit the population to increase. This growth would continue until the population is eventually suppressed by density-dependent factors. The actual size of the spawning stock at this equilibrium point is primarily established by some factor related to the scale of the environment (often referred to as its carrying capacity). Thus while the estimate of SPR does not consider the actual size of the spawning stock, it anticipates an average mean spawning stock size. For constant recruitment, the percentage effect of fishing on SPR and the spawning potential of the stock are the same. However, if recruitment responds to the reduced abundance of the adults caused by fishing mortality, the two measures of the reproductive potential of the stock are potentially quite different.

The actual size of the spawning stock in a sustainable fishery retains the scaling effect of the carrying capacity of the environment. This factor provides no information related to the capacity of the stock to sustain further increases in fishing mortality beyond that measured by SPR. In order to employ the estimated size of the spawning stock in management, it is first necessary to develop a time series of stock and recruitment for that particular stock. This information is needed to judge the significance of the magnitude of any particular realization of the spawning stock size. Absent such knowledge, SPR can be very informative. However, without knowledge of stock trends there is no basis for judging the adequacy of any conservation measure. Consequently, conservation standards based upon SPR must be augmented by monitoring trends in the actual size of the spawning stock. In most situations this should not be a significant impediment, since collection of annual statistics that lead to estimates of SPR also provide information on the size of the spawning stock.

Selection of Critical Minima for SPR

Requirements

Obviously, there is a limit to the degree density-dependent processes can relax to permit increased survival. Once fishing causes the required increase in compensation to pass that limit, the population will begin to decline toward extinction. For most stocks that limit is probably variable in time and space due to variable environmental factors. Recruitment could be maintained relatively high (on average) until the limit is exceeded by increased fishing or its own random variation. The population and the fishery it supports could then suddenly collapse without warning.

Goodyear (1977) designated the upper limit on the compensatory capacity of a stock as CR_{crit} to represent the upper (critical) threshold beyond which additional mortality would drive the population to extinction. He further suggested that for management purposes CR_{crit} should be set at a value to protect against serious stock depletion. The analogous level of SPR, SPR_{crit}, is the reciprocal of CR_{crit}. The relationship between SPR and the increase in pre-recruit survival probability required for population persistence is given in Fig. 14. This relationship is independent of the particular species, but some (or all) of the required compensatory change may be the result of increases in age-specific maturities or fecundities depending on the specific characteristics of the species. Assuming a 50/50 sex ratio of eggs, the theoretical lower limit to the value of SPR_{crit} is the value $2/P_{unfished}$. The realized level is the reciprocal of the product of $P_{unfished}$ and the mean annual density-independent natural survival probability. In species that produce large numbers of eggs, the density-independent survival probability is very low. This situation is typical of pelagic spawners (see, for example, the many papers in Blaxter 1974). This low density-independent survival probability causes $P_{unfished}$ to have no predictive value for SPR_{crit}. Several potential approaches to establishing guidelines for selection of SPR_{crit} are available: 1) the relative magnitude of the compensatory change in survival per unit change in SPR; 2) observations of declines in recruitment in stocks where SPR has been estimated; and 3) information derived from spawner-recruit relations that have been fitted to other fish populations.

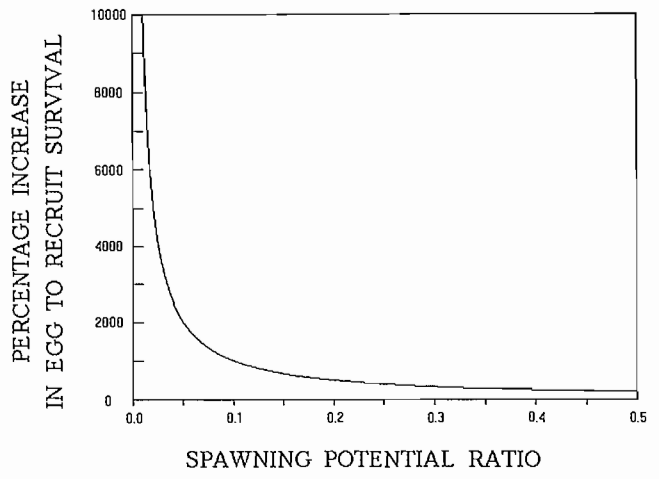


Fig. 14. Relationship between SPR and the proportional increase in egg to recruit survival required for population persistence.

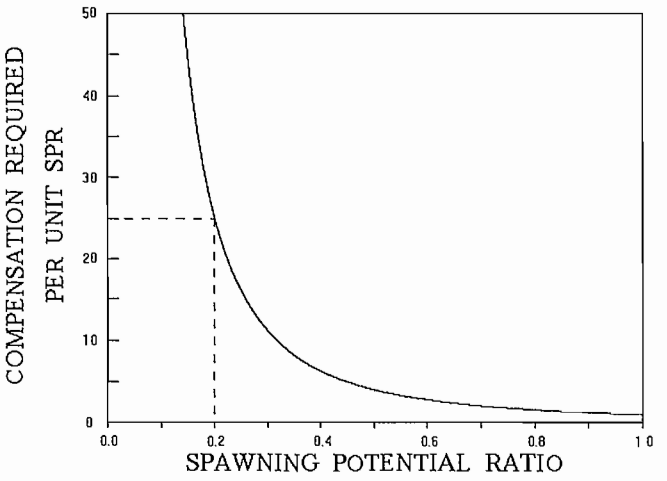


Fig. 15. Increase in magnitude of compensation required for persistence as SPR declines. Ordinate values are ratios of the compensatory change in mortality required to offset a reduction in SPR relative to that required at SPR=1.0.

Force of Exogenous Mortality

The relation between SPR and the required mean (equilibrium) increase in pre-recruit survival rates for population persistence presented in Fig. 15 provides information relative to the first of these approaches. The slope of the curve rises rapidly below about SPR = 0.20, suggesting that the compensatory reduction in mortality required for population persistence becomes disproportionately larger than the change in the SPR as SPR declines. The rate of increase in the amount of compensation required per unit reduction in SPR is presented in Fig. 15. The values on the ordinate are the ratios of the change in compensatory mortality for a unit reduction in SPR relative to the first unit reduction from the unfished stock. Thus, for population persistence, a unit reduction in SPR at SPR=0.2 requires a compensatory increase in survival which is about 25 times greater than the change required at SPR=1. Although there is no discontinuity in the curve which signals a transition from safe to unsafe levels, simple inspection of its shape suggests that the threshold level for the SPR should probably not be set much below 20% without considerable justification and that 30% might be a more reasonable first choice.

Observed Recruitment Patterns

The ICES working group on methods of fish stock assessment (Anon. 1983) outlined a method for the examination of SSBR using scattergrams of stock and recruitment to define levels of fishing mortality (F_{high}, F_{med}, and F_{low}) that correspond to high, moderate, and low risks of stock collapse. Gabriel et al. (1989) and Gabriel (1985) applied these concepts in an evaluation of SPR and recruitment for several demersal finfish species of the Northwest Atlantic. They noted declines in recruitment at levels of SPR below 20% in several species. Their results led to recommended conservation targets for SPR of no less than 20% for cod, 28–30% for haddock, 25–30% for yellowtail, and 20% for other species (Anon. 1987). More recently, Mace and Sissenwine (1993) used this approach to examine data for 91 fished stocks of 27 species and found that SPR averaged about 0.19 overall with variations according to taxonomic affinity and life history characteristics.

Analysis of Fitted Stock-Recruitment Curves

A third method of evaluating possible critical limits for SPR in stocks where knowledge is limited is through analyses of the frequency distributions of fitted stock-recruitment curves from the literature. Note that the limiting SPR for a population's persistence is the reciprocal of the slope of its stock-recruitment curve at the origin (Goodyear 1977, 1980; Anon. 1988a). Thus, one alternative would be to construct a cumulative frequency distribution (CDF) of fitted stock-recruitment curves (perhaps by groupings of similar life histories). The critical levels could then be based on some agreed-upon criterion, for example the value corresponding to the 20th percentile of the distributions. Assuming the fitted slopes had been accurately estimated, this procedure would establish a limit that would protect 80% of the stocks from complete collapse (extinction). However, criteria based on the minimum values of SPR consistent with population persistence would probably be too dangerous to employ.

A somewhat better approach might be to establish a minimum conservation standard for SPR based on a safer criterion. One approach would be to adopt a standard for SPR consistent with average recruitment rates that would produce some agreed upon fraction of recruitment required for Maximum Sustainable Yield (MSY). These levels should be somewhat higher than the minimum consistent with population persistence and would therefore be more protective of the stock. In deterministic stock-recruitment relationships there is a level

of parental stock that produces a maximum of excess recruitment. By analogy, in the real (stochastic) world, there is an average parental stock level that produces on average a maximum excess in recruitment. Fishing at an exploitation rate that results in such a parental stock and also maximizes the biomass yield for each recruit to the population produces MSY.

Note that this theoretical MSY requires: 1) a stock size that produces maximum excess recruitment, and 2) exploitation of the excess recruits in a manner which maximizes yield for each harvested recruit. Theoretically, this would require harvesting all of the excess recruits at the time of their maximum biomass. Fisheries on wild stocks cannot generally be prosecuted in this manner. Consequently, there exists a separate value of MSY that would be realized for any particular ogive of age-specific selectivities of a population to fishing mortality. Consequently, from a practical point of view, MSY and F_{MSY} are not unique parameters because the maximum yield attainable from a stock involves both the production of recruits and their exploitation. However, on the production side, there is a unique value of spawning stock in deterministic stock-recruitment models that produces maximum excess recruitment. Since equilibrium requires harvesting all of the excess recruits, there is a unique fishing mortality rate F_{MSY} that accomplishes this constraint.

I evaluated the behavior of the SPR required for recruitment rates that produce MSY and 50% of MSY for the standard Ricker and Beverton-Holt stock-recruitment model of the forms previously employed. For both models, the recruitment at MSY is given where the slope of the stock recruitment curve is unity (Ricker 1973). Recruitment at MSY and 50% of MSY was evaluated numerically over a large range of possible parameter sets for both models. The absolute level of recruitment for both of these models was a function of the parental stock size and model parameters. However, the level of SPR associated with recruitment rates that produced MSY and 50% of MSY were solely a function of the slope of the stock-recruitment curve at its origin (Figures 16 and 17). The SPR required for 50% of MSY recruitment also is remarkably similar for the two models (Fig. 17).

The value of SPR associated with MSY recruitment might be considered a management goal while the lower values associated with declines to 50% of MSY may represent a lower critical threshold. SPR levels for MSY recruitment vary over a range of 15–90% over the parameter space evaluated. Clark (1991) found that a large fraction of MSY for several groundfish stocks could be obtained by harvesting at about 35% SSBR over the likely range of density dependence for these stocks (initial SR slopes of 4–16). Given the general shape of the curves of Fig. 17 and the knowledge that near maximum yields are obtained in the regions just above the isopleths, it is evident that his results are in close agreement with the present analyses and are a consequence of his selection of reasonable slopes for the spawner-recruit curves.

To compare the SPR for 50% of MSY recruitment to estimates of the slopes of stock recruitment curves at their origin, I

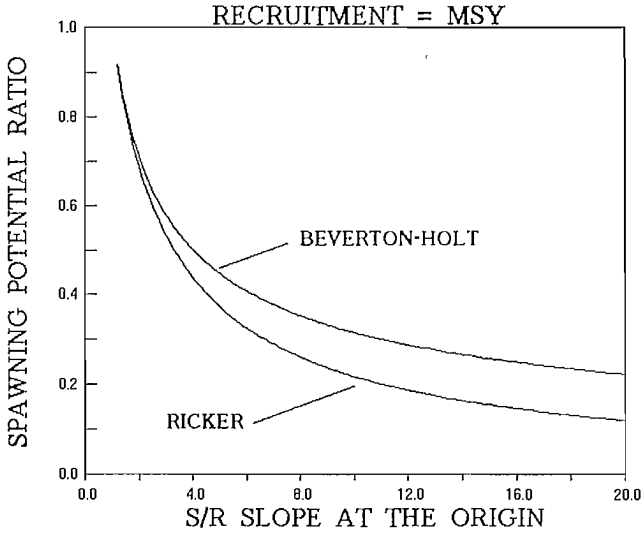


Fig. 16. SPR at MSY recruitment as a function of slope at the origin for two spawner-recruit curves.

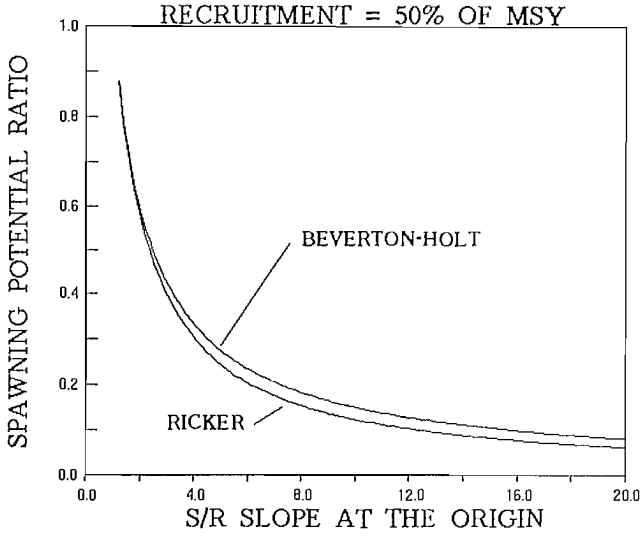


Fig. 17. SPR below which recruitment will fall below 50% of recruitment at MSY as a function of slope at the origin for two spawner-recruit curves.

overlaid the Ricker model estimates for 33 exploited stocks of marine and anadromous fishes derived by Cushing and Harris (1973) on the data of Fig. 17. The results are shown in Fig. 18. The 20th and 50th percentiles of the cumulative distribution were slopes of about 1.55 and 2.4 respectively. These levels suggest that the limiting SPR for 50% of MSY recruitment is 0.5 at the 50th percentile and 0.65 at the 20th percentile of the distribution. Both values are considerably more conservative than the 0.2 value derived from both inspection of the shape of the relationship of Fig. 15 and from the behavior of demersal stocks of the northwest Atlantic.

Three factors influence the adequacy of the results of Cushing and Harris for the present purpose. First, these authors did not account for the influence of fishing in the construction of the estimates of recruitment. Hence, their estimates of density dependence in the fished stocks should

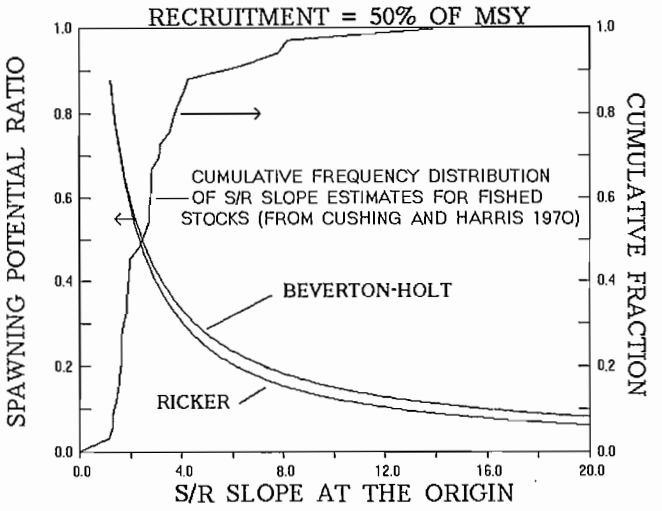


Fig. 18. Cumulative frequency distribution of estimated slopes at the origin of stock-recruit curves and limiting SPR below which recruitment will drop below 50% of recruitment at MSY for two spawner-recruit models.

already reflect the influence of some fishing mortality and, depending on the stock, are actually estimates of the slopes of the escapement curves. From this point of view the seemingly high critical values of SPR developed with this approach can be viewed as limitations on the expansion of the fisheries that existed for the various stocks over the levels existing when the stock and recruitment data were taken.

The second factor relates to the adequacy of the model representation for multiple age-class spawners. Christensen and Goodyear (1988) used simulation techniques to examine the robustness of statistical procedures to estimate the slope at the origin of an underlying Ricker stock-recruit relation in a multiple-age spawning population. Their results indicated that, if the stock-recruitment data do not correspond to parental stock fecundity and the lifetime egg production of the filial yearclass, regression techniques will commonly provide estimates of the slope at the origin that are biased high for low true values and low for high true values.

The third factor results from a statistical artifact encountered when fitting the Ricker model to random "observations" of stock and recruitment. The linearized form of the model involves regressing R/P as a function of P. This results in estimates of slopes at the origin of about 2.7 with correlation coefficients of 0.71 (see Kenney 1982). Unfortunately, nonlinear fitting techniques such as those employed by Cushing and Harris (1973) do not remove the bias in estimates of the slope from such data (Christensen and Goodyear 1988). Thus, to the extent that random noise influences the estimates from Cushing and Harris, all of the slopes will be biased towards a median value of about 2.7. Given these uncertainties, it seems clear that the CDF obtained from the fits presented by Cushing and Harris cannot be used to establish limits for SPR without additional analysis. However, it is also clear that their results do not support selection of critical values lower than the 20% derived from the first two approaches.

Taken together with the findings of Mace and Sissenwine (1993), these observations suggest a working critical minimum for SPR of about 0.2 (20%) for stocks where the spawner-recruit relation is indeterminate or cannot be evaluated with existing data. An evaluation of the adequacy of this estimate should be considered an important research problem. This could be accomplished by compilations of additional empirical evidence of the relation between SPR and recruitment or of the distributions of slopes of stock-recruit curves at their origins.

SPR Contrasted with Other Biological Reference Points

Yield Per Recruit

Note that the size and mortality dimensions of Figures 9 and 10 conform to a common presentation of yield-per-recruit analyses. Consequently it is possible to calculate yield and SPR for each level of fishing mortality and minimum size. This permits evaluations of the location of isopleths of SPR on standard yield-per-recruit analyses and on the surface of yield as a function of fishing mortality and minimum size.

Analyses of yield per recruit are given in Fig. 19 assuming no fishing-induced mortality of fish below the minimum size and in Fig. 20 for a 10% mortality of released fish. Release mortality was incorporated in these calculations following the methods of Waters and Huntsman (1986). For comparison the isopleths of 5, 10, and 20% SPR are superimposed on the surface of yield per recruit for the same parameter space.

Two biological reference points related to yield per recruit are often discussed in relation to stock management: F_{max} and $F_{0.1}$. F_{max} is a value of fishing mortality that maximizes yield per recruit, and $F_{0.1}$ is the value of fishing mortality at which the incremental gain in yield for an increase in fishing mortality is 10% of the yield per recruit produced at very low levels of F (Gulland and Boerema 1973). A typical analysis is presented in Fig. 21 for Gulf of Mexico red snapper along with SPR. In this example SPR is 30% at $F_{0.1}$ and just under 20% at F_{max}. These analyses assume age-1 snapper to be 0.02 and age-2 snapper to be 0.389 as vulnerable to fishing as all older ages in the population. Depending upon natural mortality and growth in a population, F_{max} may approach infinity because yield per recruit increases monotonically with increasing fishing mortality. It is always greater than $F_{0.1}$, and for a wide class of models F_{max} is also greater than F_{MSY} (Deriso 1982). Because F_{max} may contemplate extremely high levels of fishing mortality that would reduce the spawning potential of the stock to near zero, F_{max} is not a generally useful conservation standard. On the other hand, $F_{0.1}$ is always less than infinity, and because of the nature of growth and mortality it is often at a level that provides substantial survival of recruits into the age classes contributing to the spawning potential of the stock. Thus, while this measure does not directly incorporate notions of the spawning stock, its adoption as a conservation standard

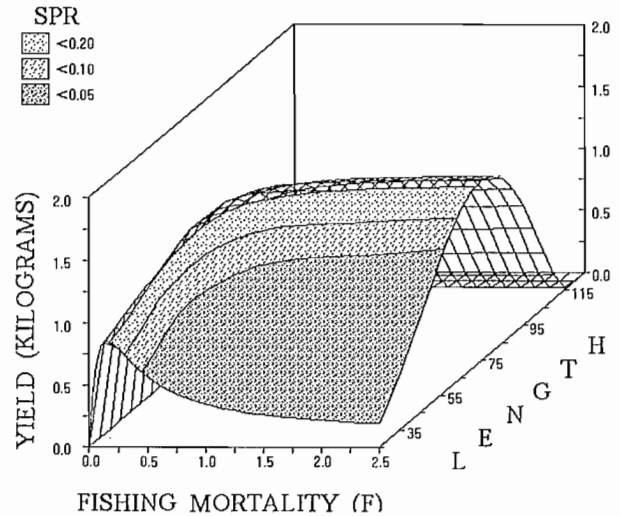


Fig. 19. Isopleths of SPR on the surface of yield per recruit as a function of length at recruitment and fishing mortality assuming no fishing-induced mortality of sublegal fish.

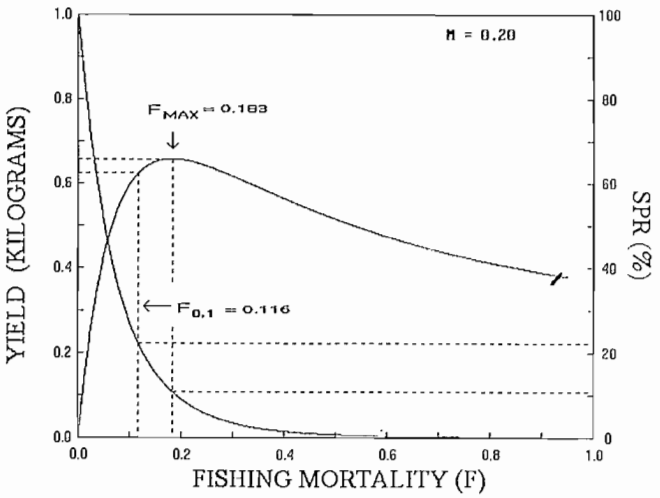


Fig. 21. Estimates of F_{max} and $F_{0.1}$ for Gulf of Mexico red snapper and corresponding levels of SPR.

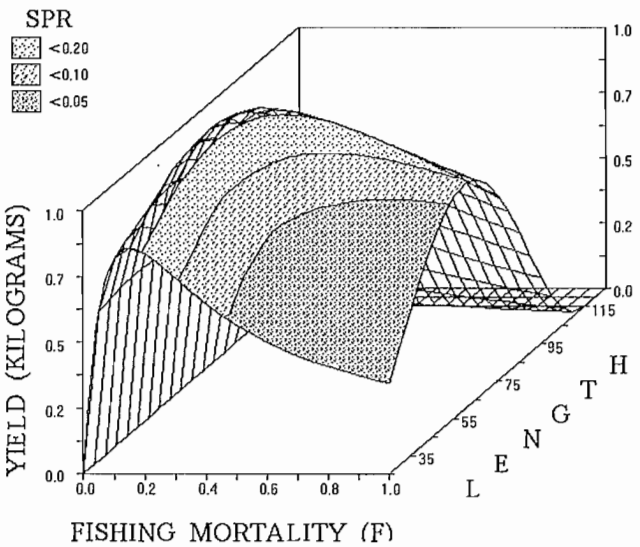


Fig. 20. Isopleths of SPR on the surface of yield per recruit as a function of length at recruitment and fishing mortality assuming 10% of released sublegal fish die as a result of the experience.

has profound implications for the expected lifetime reproductive contribution of a recruit. Because of these implications $F_{0.1}$, originally proposed as an economic consideration, has been widely adopted as management goal for fishery conservation. Because the contour for $F_{0.1}$ lies on the far side of the contours of Figures 19 and 20, it cannot be plotted on the same perspective. However, a two-dimensional presentation of the same information permits simultaneous presentation of $F_{0.1}$, F_{max}, and isopleths of yield and of SPR as a function of minimum size and fishing mortality. Examples of two-dimensional presentations of the data of Figures 19 and 20 and $F_{0.1}$ and F_{max} are presented in Figures 22 and 23. The shaded contours represent levels of SPR that decrease from upper left to lower right in the order 0.3, 0.2, 0.1, 0.05, and 0.01. The yield contours for the Fig. 22 increase from left to right and for Fig. 23 they increase toward the innermost concentric circle. For both figures the yield contours increase in the order 0.25, 0.50, 0.75, 0.9, 0.95 and 0.99 of the maximum yield attainable within the parameter space.

Experimentation with parameters from a number of stocks suggest that the 20–30% SPR contours often lie very near or slightly above profiles of fishing mortality rates that produce $F_{0.1}$. Consequently, selection of critical levels for SPR within this range often produce maximum to near maximum yield per recruit, and little biomass yield for each recruit will be lost by adopting SPR goals in this range. Inspection of the behavior of $F_{0.1}$ and F_{max} in Figures 22 and 23 reveals a sometimes overlooked feature of their definitions: neither is a constant for the population. The values taken by both measures are functions of the distribution of fishing mortality among ages in the population. If this distribution changes because the fishery changes to target younger fishes because of declining stock, or if it changes because of conservation actions, the value of the measure shifts. Although commonly employed, $F_{0.1}$ is a "backdoor" approach to preservation of spawning potential since it does not directly consider the spawners. Further, since the actual value of $F_{0.1}$ depends on the characteristics of the fishery, it is not uniquely related to any attribute of the spawning potential of the stock. Any changes resulting from modifications of the selectivity of gears employed or caused by the imposition of conservation measures that affect the age distribution of the harvest will cause a change in the value of $F_{0.1}$. These changes are independent of their effect on the spawning potential of the stock. Deriso (1987) defined an optimal $F_{0.1}$ as a combination of the size or age at entry of fish into a fishable population that maximizes the yield obtained from fishing at $F_{0.1}$. This value is unique for the simple yield-per-recruit theory in Beverton and Holt (1957). However it is not well suited to the management of fish stocks exposed to competing fisheries with different ages at entry and different objectives.

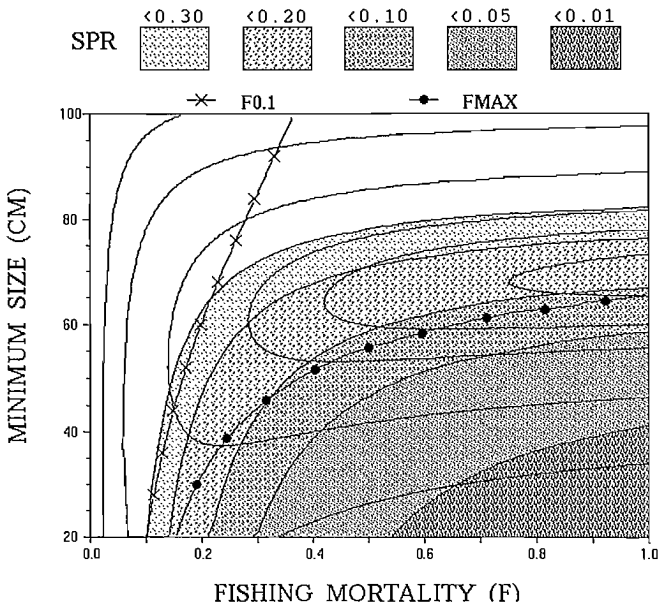


Fig. 22. SPR, yield per recruit, F_{max} and $F_{0.1}$ as a function of length at recruitment and fishing mortality for Gulf of Mexico red snapper assuming no fishing-induced mortality for sub-legal fish.

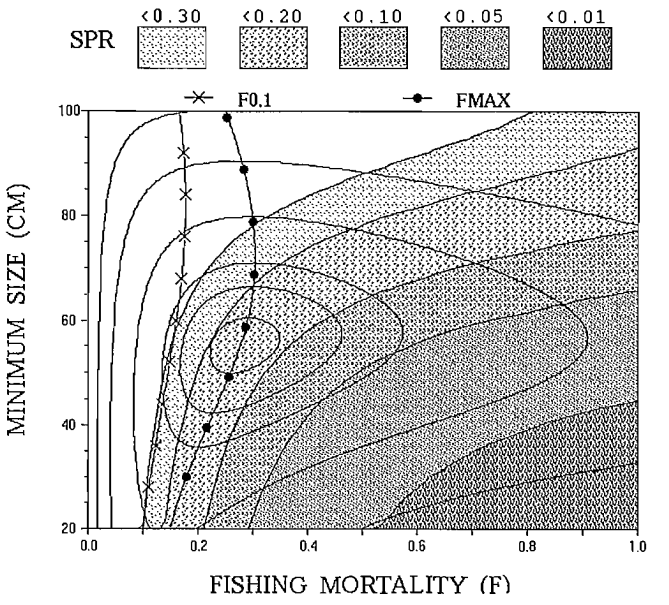


Fig. 23. SPR, yield per recruit, F_{max} and $F_{0.1}$ as a function of length at recruitment and fishing mortality for Gulf of Mexico red snapper assuming a 10% mortality for fish caught and released below the minimum size.

Optimal Spawning Stock

One possible biological objective for management is to manage fishing mortality so that the spawning stock is maintained at some particular absolute (non-zero) level. This option has the advantage that (if successful) it directly preserves the stock at a level at least equal to the objective. The level of the objective itself may be based on surplus production models, spawner recruitment models, or similar considerations. Inadequate data are a principal impediment to the application of this approach for many stocks. However, where sufficient information is available and the population renewal processes conform adequately to the assumptions of such models, the level of spawning stock can be set to achieve maximum surplus production, which leads to such things as maximum sustainable yield, or other similar goal. Because the objective involves maintaining spawning stock at a constant level, successful implementation ensures stock persistence. Whether the yield is the maximum (or optimum) attainable from the stock depends upon the adequacy of the model and estimates of its parameters. If the assumptions are met, then managing for a spawning stock that produces the maximum average recruitment will maximize long-term yield (Reed 1979). Thus, managing for a fixed, optimum spawning stock in some fisheries may permit higher average yields than could be attained though a management scheme based on SPR.

Fixed Rate Criteria

Although fixed escapement policies have been shown to maximize yield for a large number of population models (Reed 1979), they are less frequently employed in fisheries management than other schemes, in part because they can result in large annual variations in allowable harvest. In contrast, fixed harvest rate policies allow more stable harvests so long as the rate is low enough to permit population persistence. Management based primarily on F_{MSY}, F_{rep}, F_{max}, $F_{0.1}$, or F_{mmy} (Clark 1991) are fixed rate policies.

Management based on SPR also employs the concept of a fixed rate and shares the good (and bad) attributes of such policies. SPR, however, differs from the other fixed rate approaches in that it is not defined in units of fishing mortality or associated yield. This is of no disadvantage since SPR can be readily evaluated for any realized ogive of fishing mortalities. However, it provides the advantage of separating harvest objectives from conservation objectives. This characteristic allows a unique value of SPR to be employed as a conservation standard without the prejudice of the nature of harvesting strategy subsumed by the definitions of F_{MSY}, F_{rep}, F_{max}, $F_{0.1}$, and F_{mmy}. The advantage to decoupling the conservation standard from the harvesting strategy becomes immediately apparent if there are several distinct fisheries competing for a resource. This is a common problem when there is competition between recreational and commercial fisheries. These fisheries often seek the same species with different gears, at different places, and at different ages. Further, their objectives tend to be quite different. Given estimates of each component's contribution to fishing mortality by age, it is possible to apportion realized SPR among the fisheries. Similarly it is possible for management to allocate fractions of the allowable reduction of SPR among the fisheries according to any desired scheme. Within that framework, each fishery could be managed according to harvesting strategies designed for that fishery. This approach provides a mechanism to avoid undesirable reallocations of allowable harvest by imposing regulations which unintentionally change the relative availabilities of legally harvestable fish among the fisheries.

Current and Future Usage

The 1990 amendments to the Magnuson Fishery Conservation and Management Act (U.S. Congress 1990) required definitions for overfishing for each fishery management plan administered under its Authority. This action was intended to be risk-aversive to avoid depletion of stocks. Most of the overfishing definitions submitted to date have been based on SPR, usually in the context of a percentage of the unfished SSBR (Mace and Sissenwine 1993). The critical values are intended to be minimum acceptable levels above which stocks maintain acceptable productivity and have typically been set in the range of 0.2 to 0.3 primarily based upon the experience in the Northwest Atlantic. This approach to management is too recent to evaluate its ultimate utility based on its empirical track record. However, there is reason to believe that it will protect against stock depletion as well as any other approach so long as the critical levels are set within acceptable ranges. It also provides a straightforward basis for the evaluation of persistent non-fishing anthropogenic sources of mortality against the spawning stock objectives for managed fisheries. Given the tendency for adoption of SPR minima in definitions of overfishing in U.S. FMP's and elsewhere, the strengths and weaknesses of this criterion for management should become readily apparent. Its ultimate utility will probably be assessed in some future historical retrospective of how it fares in the context of actual management activities: the restoration of now depleted stocks and the continued productivity of those now robust.

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The Development of a Management Procedure for the South African Anchovy Resource

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Exploitation of the South African anchovy resource commenced in 1963. Until 1983, the basis for management of this resource very roughly resembled a constant catch strategy, as the TAC for all pelagic species combined was kept virtually unchanged. Hydroacoustic surveys of the resource commenced in 1983, and showed it to be larger and less heavily exploited than had been thought previously on the basis of analysis of data from commercial catches alone. These surveys led to management on a constant escapement basis over the 1987–1990 period; the actual criteria applied involved pre-specified probabilities of escapement falling below 50% of the average spawning biomass level in the absence of exploitation. This procedure initially found general favour, when TAC's were increased markedly for 1987 and 1988 following good recruitment. However, it was quickly overridden by decision makers when it indicated fishery closures following poor recruitment in 1989 and again in 1990. It has now been replaced by a management procedure based on the constant proportion strategy, which was selected on the basis of the results of simulation testing over a twenty year time horizon. A key requirement for sound interpretation of the results of this process is an appropriate definition of risk, which must also be readily meaningful to layman decision makers. This is closely related to the question: at what levels of spawning stock depletion is the probability of recruitment success likely to become substantially impaired? TAC's should ultimately be set by means of such simulation-tested management procedures, and not by the present widely used practice of linking annual "best" assessments of resource status to biological reference points. The future role of assessments is rather to condition the underlying models used for such simulation tests, as management procedures are revised at (typically) three to five year intervals.

L'exploitation de l'anchois d'Afrique du Sud a commencé en 1963. Jusqu'en 1983, cette ressource était gérée dans l'ensemble selon une stratégie de prise constante, car le TPA de toutes les espèces pélagiques combinées était pratiquement immuable. Des relevés hydroacoustiques de la ressource, entrepris en 1983, ont montré qu'elle était plus importante et moins fortement exploitée qu'on l'avait cru précédemment d'après l'analyse des données des prises commerciales seules. On a donc décidé de gérer cette ressource selon une échappée constante pendant la période de 1987 à 1990; les critères appliqués actuellement comprennent des probabilités pré-établies de l'échappée qui sont inférieures à la moitié de la biomasse moyenne du stock reproducteur en l'absence d'exploitation. Dans l'ensemble, cette méthode a été bien accueillie au début lorsque les TPA ont été sensiblement augmentés en 1987 et 1988 à la suite d'un bon recrutement. Toutefois, les décisionnaires l'ont rapidement abandonnée lorsqu'on a signalé des fermetures de pêche après un faible recrutement en 1989 et de nouveau, en 1990. Cette méthode est maintenant remplacée par une méthode de gestion fondée sur une stratégie à proportion constante choisie selon les résultats d'un essai de simulation sur une période de vingt ans. Une bonne définition du risque, qui doit aussi être facilement intelligible pour des décisionnaires non spécialistes, est une exigence clé pour une interprétation judicieuse des résultats obtenus par cette méthode. Cette exigence est étroitement liée à la question suivante: à quel taux d'affaiblissement du stock reproducteur la probabilité de la réussite du recrutement risque-t-elle d'être sensiblement altérée? Les TPA devraient finalement être fixés par des méthodes comme des méthodes de gestion vérifiées par des simulations, et non par la méthode largement utilisée qui consiste à lier les «meilleures» évaluations annuelles de la ressource à des points de référence biologiques. Dans les prochaines années, le rôle des évaluations sera plutôt de déterminer les modèles sous-jacents utilisés pour ces essais de simulation, étant donné que les méthodes de gestion sont examinées (en général) à intervalles de 3 à 5 ans.

The anchovy (*Engraulis capensis*) fished off the west and south coasts of southern Africa (see Fig. 1) by purse-seiners, is a small and short-lived pelagic species, which is used primarily for the production of fish meal and oil. Few of these anchovy live beyond 3 years of age, at which time their mass scarcely exceeds a mere 20 g. The fisheries off Namibia and South Africa are well separated spatially, with very little catching taking place between

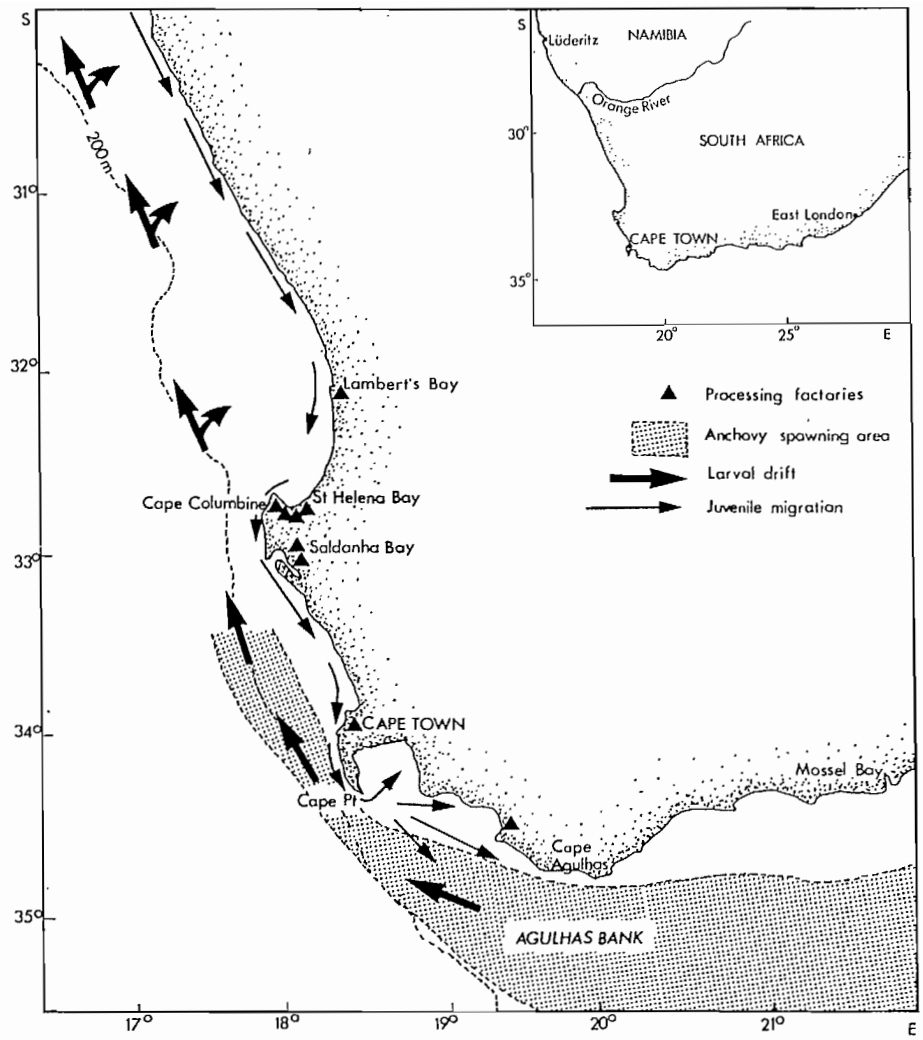


Fig. 1. A map of southern Africa, showing the distributional dynamics of the South African anchovy population schematically (reproduced with slight modification from Waldron et al. 1989).

Luderitz in Namibia to well south of the Orange River off South Africa. Traditionally, the two fisheries are regarded as exploiting different populations, and this paper considers only the South African "stock".

Anchovy distribution dynamics have only become properly understood following the commencement in 1983 of hydroacoustic surveys by a specialist research vessel (Hampton 1987). The anchovy spawn over the summer months in the vicinity of the Agulhas Bank to the south of the country. Eggs and larvae are transported by currents up the west coast (there is also some transport towards the east, but this seems to be of little consequence in the context of the fishery) (Shelton and Hutchings 1982). Recruitment of 0-year-old anchovy takes place off the west coast, and the fish work their way back down to the Agulhas Bank, keeping close to the coast, to spawn at the end of the year (anchovy are mature at an age of one year). Most of the fishing takes place on these juveniles during this southward migration, which has typically developed sufficiently to allow such fishing to commence sometime in April. Usually the fishing season finishes near the end of July when the Total Allowable Catch (TAC) has been taken (weather conditions for fishing deteriorate after this time). The adult stock appears to remain offshore on the Agulhas Bank for most of the year, and is usually relatively unavailable to the small purse-seiners which fish close to the coast. Thus, although the fishery opens in mid-January, in most years only light fishing on adults will occur for the first three months, so that an average of some 70% by mass (and more by number) of the anchovy catch comprises 0-year-old fish with an average mass of some 7 g. Occasionally, however, oceanographic conditions are such as to concentrate the adult fish closer to the coast, resulting in rather larger catches being made off the south and southwest coasts during the early stages of the season.

The South African pelagic fishery developed after World War II, primarily on the basis of exploitation of the pilchard (*Sardinops sagax*) (see Fig. 2). Initially, the legislated minimum net mesh size was too large to retain anchovy. However, with the onset of the collapse of the pilchard fishery in 1963, smaller 12.7 mm mesh nets were introduced. By 1965, all vessels were using these nets, and an anchovy fishery was

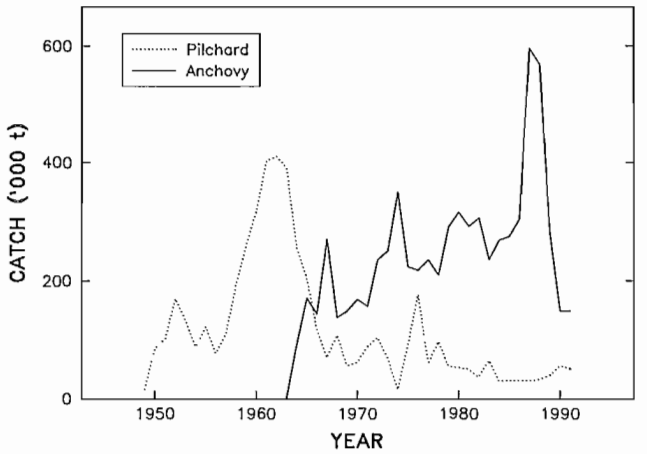


Fig. 2. Annual landings of anchovy and pilchard in South Africa from 1949 to 1991.

well established. The anchovy has since become the mainstay of the South African pelagic fishery, making the largest contribution of any species to pelagic landings in every year since 1966.

During the 1960's, the only restrictions on the pelagic fishery involved limitations on vessel and processing capacity. In 1971, a global (all pelagic species combined) TAC was introduced, presumably in reaction to the dramatic collapse of the adjoining Namibian pilchard fishery at that time. Management continued on that basis until 1983, with the global TAC fixed at values ranging between 360 and 450 thousand tonnes. The initial basis for choosing these values was entirely intuitive, though a combined-species MSY estimate of 360 thousand tonnes was suggested in 1974 on the basis of a CPUE-effort regression (a method of analysis now appreciated to be flawed). More detailed reviews of the ecology of the anchovy and of the associated fishery may be found in Butterworth (1983) and Crawford et al. (1987).

By 1983, these management measures were under criticism for two reasons. Based on analysis by Crawford (1979), the scientific recommendation at the time was to decrease the global TAC from the then current 380 thousand tonnes to 325 thousand tonnes, to make allowance for the virtual disappearance of chub mackerel (*Scomber japonicus*) and horse mackerel (*Trachurus trachurus*) from the pelagic catch. Furthermore, there was concern that the anchovy contribution to the global TAC had climbed to some 300 thousand tonnes. Virtual Population Analysis (VPA) assessments (e.g., Butterworth 1983) suggested a very high level of fishing mortality on 0-year-old anchovy, and indicated an MSY in the 250 thousand tonnes region. In the absence of any direct survey estimates of abundance at that time, the anchovy VPA was tuned using pelagic effort data (whose proportional relationship to fishing mortality is somewhat questionable because of the shoaling behaviour of the fish).

These analyses contributed to a decision in 1983 to change to species-specific TAC's. The anchovy catch level was reduced, although for reasons more complicated than merit discussion here (see Anon. 1983). However, these events were rapidly overtaken by the results of hydroacoustic surveys carried out by the newly acquired research vessel *Africana* (Hampton 1987) (see Table 1). These showed that the high "total mortality" suggested by the age structure of the anchovy catch was not a reflection of high fishing mortality, but rather of emigration of the adults from the near-shore fishing grounds off the west coast to offshore on the Agulhas Bank. While VPA had indicated spawning biomasses in the 200–300 thousand tonnes range, the hydroacoustic assessment for November 1984 exceeded one million tonnes, suggesting an under-rather than an over-exploited resource.

Over the period from 1983 to 1986, confidence in the survey approach increased, the November survey was extended to allow abundance estimation by the egg production as well as the hydroacoustic method (Armstrong et al. 1988; Hampton et al. 1990; Shelton et al. 1993), and a mid-season hydroacoustic recruitment survey off the west coast was introduced. Consequently, scientific recommendations for TAC's for anchovy came to be based entirely on the results of these surveys. This paper will not detail the initial attempts along these lines [such as, for example, by Armstrong and Butterworth (1986)]. Instead, the next section describes the approach based on probabilities and a biological reference level which was developed and applied after 1987, and explains why this had to be discarded in 1990. This is followed by a discussion of the method, based on simulations of harvesting over a future 20-year period, which was then used to investigate alternative candidate management procedures, and how a selection was made between these — the procedure thus chosen being implemented in 1991.

In the final sections of the paper, future possibilities for improvement of the current procedure are discussed, together with their wider implications for risk evaluation and biological reference points, and the key lessons and outstanding problems evident from the experience gained in developing a basis for management of this fishery are summarised.

Survey-Based Anchovy TAC's: 1987–1990

The population dynamics model and estimation procedure used as the basis for scientific recommendations for TAC's over this period are detailed in Appendix A. The model assumes that annual recruitment is log-normally distributed about some median value (R_{med}). Data from the surveys are used to estimate R_{med}, an age- and year-invariant natural mortality (M), and current (as well as historic) numbers-at-age; these estimates then enable escapements (i.e., year-end spawning biomasses) under different TAC levels to be predicted. Fig. 3 shows the most recent estimates obtained from this process of annual recruitment (scaled to R_{med}) and spawning biomass B_{sp} (scaled to the average spawning biomass in the absence of exploitation, K, which is assumed not to vary systematically with time).

A Bayes-like estimation procedure was used. At the time estimation was first attempted, relatively few survey data were

Table 1. Survey estimates of biomass for the South African anchovy resource. Units are thousand tonnes, and the figures in parenthesis are survey sampling coefficients of variation. Annual catches for the years since commencement of the surveys are also shown.

Year	Catch	Time of Survey	Recruitment Hydroacoustic	Spawning Biomass Hydroacoustic	Spawning Biomass Egg Production
1983	236	November		408 (0.25)*	
1984	269	November		1067 (0.23)	
1985	275	May	310 (0.24)		
		November		975 (0.16)	616 (0.40)
1986	304	May	235 (0.18)*		
		June	466 (0.17)		
		November		1747 (0.15)	2001 (0.35)
1987	596	July	575 (0.15)		
		November		1460 (0.12)	1606 (0.30)
1988	569	July	508 (0.18)		
		November		1106 (0.31)	2048 (0.35)
1989	280	June	132 (0.19)		
		November		546 (0.13)	487 (0.34)
1990	149	May	150 (0.26)		
		June–July	137 (0.25)		
		November		335 (0.21)	882 (0.58)
1991	149	May	332 (0.23)		

* Considered to be negatively biased because of limited areal coverage.

available, so that it was desirable to attempt to improve estimation precision by incorporating auxiliary information in the estimation procedure by means of Bayesian priors (*inter alia*). For example, the fact that reasonably high anchovy catches had been sustained over the 1965–1982 period, prior to availability of survey information, indicates that R_{med} cannot be too low (or the catches made would have extirpated the resource) — the estimation procedure used was easily extended to take account of this.

Before discussing the historical sequence of TAC decisions over the 1987–1990 period, it is helpful to describe the three-stage process that came to be used for this purpose (even though it was applied in this complete form only at the end of this period). The process is also shown schematically in Fig. 4.

1. Once the preceding season had ended, survey results (including particularly the most recent recruitment survey) allowed probability distributions for numbers-at-age at the end of the year to be estimated, given information on the recent catches. These could be projected forward another year, under the assumption that the probability distribution for recruitment for that year was centered upon its median value, to provide estimates of the probability distribution for escapement at the end of that year for various TAC levels. An initial (pre-season) TAC was specified on this basis (typically in October) to assist the industry with planning.

2. When data from the November spawning biomass survey had been processed, the resultant spawning biomass estimates would allow improved estimates of numbers-at-age at the end of the year to be made. These estimates would be projected to the end of the next year as for 1 above (i.e., still assuming the distribution of recruitment for that year to be centered on its median value, as estimated from surveys in earlier years), to provide the basis for recommending a "first revision" of the TAC. A decision in this regard would usually be announced in February or March.

3. Once the recruitment survey (May or later) had taken place, a direct estimate of recruitment strength for the year in question was available, so that a much improved estimate of escapement at the end of the year could be made. A final TAC for the season would then be announced, typically in July.

The choice of the timing for the recruitment survey involved a trade-off. A later time gave a better chance that the recruitment process was near complete, so that the survey result would not be severely negatively biased. However, continuity of fishing operations required earlier notification to the industry of the final TAC, to avoid any need to perhaps halt and then later recommence operations at a time when weather conditions were less favourable for fishing, should an increased TAC be announced. Earlier finalisation was also desirable to render less likely the chance that the final scientific TAC recommendation would be less than the tonnage already taken by that time.

It is also important to clarify the sources of the scientific recommendations for TAC's over this period. Prior to 1988, these were provided by scientists from the pelagic section

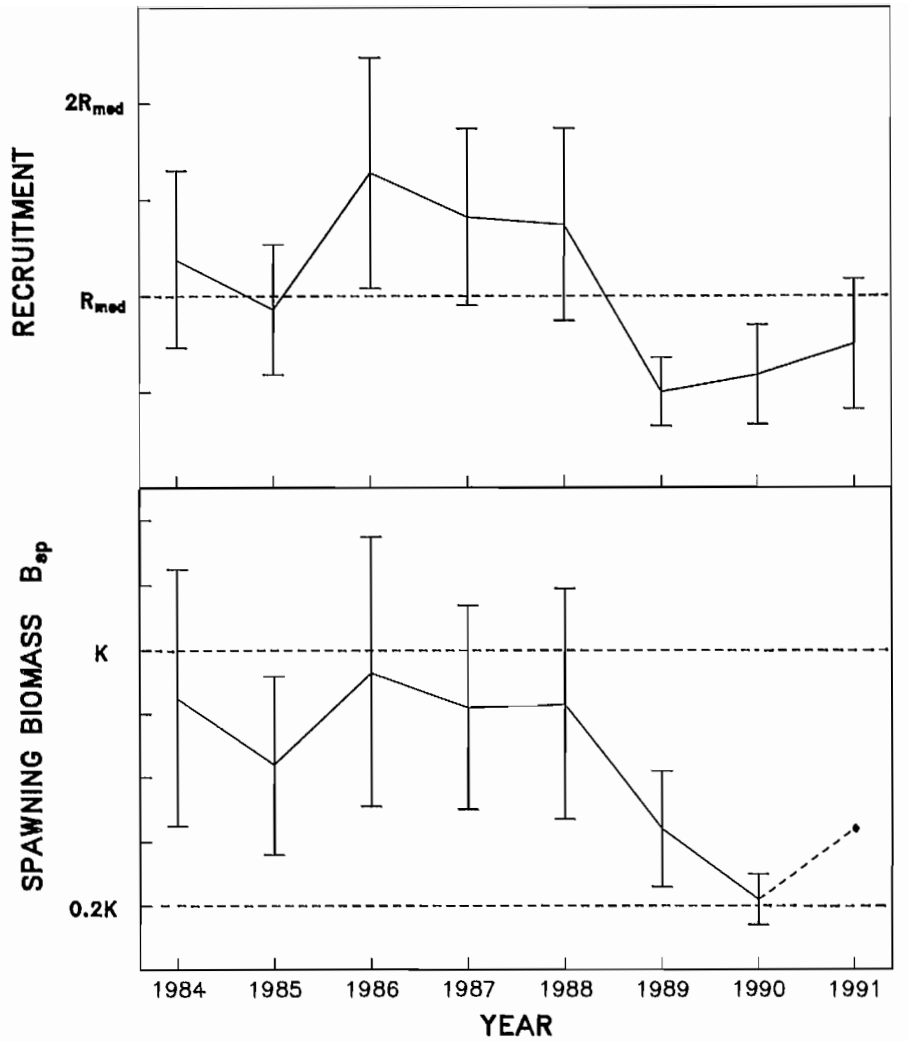


Fig. 3. Estimates of annual recruitment and end-of-year spawning biomass for the South African anchovy resource from 1984 to 1991. These are the most recent estimates obtained from survey data upon application the assessment procedure of Appendix A. The estimates are scaled to estimates of median recruitment (R_{med}) and average spawning biomass in the absence of exploitation (K) respectively. The error bars reflect ±2 standard errors. The spawning biomass estimate shown for 1991 is a projection, given the 1991 catch taken.

of the Sea Fisheries Research Institute (SFRI), a Government Agency within the Department of Environment Affairs. From late in 1987, scientists from this section met informally with scientists based at the University of Cape Town (who were engaged in research on this topic as a component of the Foundation for Research Development's Benguela Ecology Programme) to discuss the basis and then formulate scientific recommendations for anchovy TAC's to the Director of the Institute. Some two years later, this process was formalised by the annual appointment by the Director of Working Groups (see Fig. 5), consisting of scientists from both within and outside the Institute, with responsibilities which included formulating such recommendations for the various marine species harvested off South Africa.

Returning to the historical TAC sequence, the initial TAC for 1987 was set at 300 thousand tonnes, essentially on the conservative basis of making little change to the status-quo. However, two factors soon necessitated an early revision. First, the November 1986 survey results (see Table 1) suggested a spawning biomass some double that of the previous two years, as a result of a particularly good recruiting cohort for 1986. Secondly, unusual oceanographic conditions made these adult fish readily available to the purse-seiners on the south and southwest coasts during the early months of the 1987 season, so that the cumulative catch grew much faster than normal. The extent of this growth was such that the initial TAC might have been filled before fishermen on the west coast had any chance to make their traditional catch from the recruiting fish that become available only later in the season.

As a result, the TAC for 1987 was increased to 450 thousand tonnes. Later in the year, once the recruitment survey indicated that the 1987 recruitment level was quite satisfactory, the TAC was increased further to 600 thousand tonnes — a figure probably close to the industry's maximum processing capacity. These TAC levels were based on deterministic calculations which suggested that they would ensure an escapement

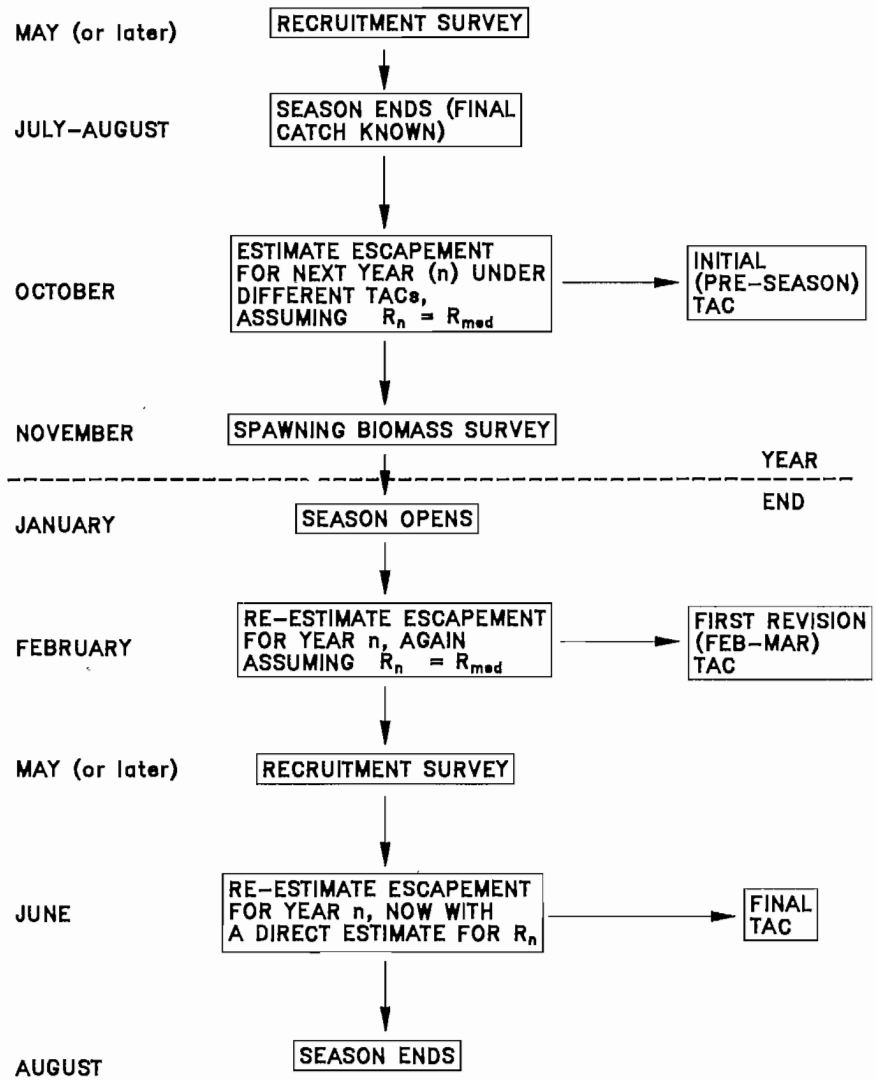


Fig. 4. Schematic representation of the procedure used to set and then revise annual anchovy TAC's over the 1987–1990 period.

(spawning biomass) in excess of one million tonnes. This was considered sufficiently large, because similar spawning biomass levels for the previous two years had provided adequate subsequent recruitment. Furthermore, decision makers were becoming sufficiently confident in the surveys and their results to be comfortable about adjustments to the TAC on that basis. Nevertheless, the TAC's allocated reflected essentially *ad hoc* decisions, rather than ones based on an a priori choice of a biological reference point.

Faced with the task of providing scientific recommendations for future TAC's in these circumstances, the scientists concerned took the view that the 1987 TAC decisions should be interpreted as defining the level of risk acceptable to the decision makers (ultimately, the Minister of the responsible Government Department — see Fig. 5). The biological reference point chosen was a spawning biomass of $0.5K$, on the basis that this corresponded to MSY level for the Schaefer model which is used frequently to provide the basis for fisheries management decisions. The estimation procedure of Appendix A provided not only point estimates of escapement under future TAC's, but also the distributions for these estimates. Such calculations then indicated that:

I. the initial 1987 revision of the TAC to 450 thousand tonnes corresponded to a probability of about 20% of an eventual escapement less than $0.5K$, and

II. the final revision (after the recruitment survey) to 600 thousand tonnes corresponded to a probability of about 10% of the same occurrence.

These scientists decided, therefore, to use these prescriptions as the basis for future scientific recommendations, so as to provide a consistent approach to management. Both prescriptions corresponded quite closely to point estimates of escapement of $0.7K$, but the reference points based on the lower tails of the distributions were preferred because they would take some account of varying levels of precision for

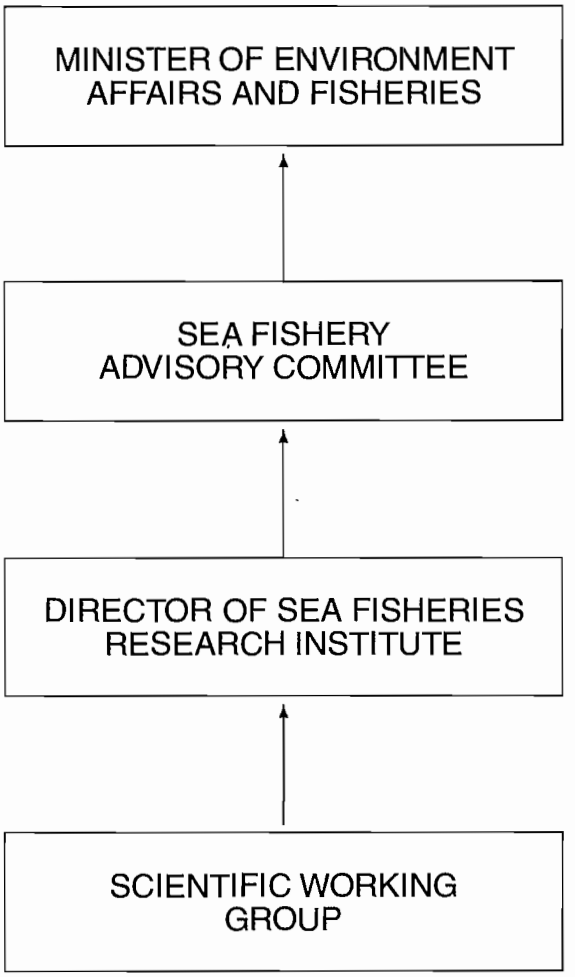


Fig. 5. The decision making process for the allocation of annual TAC's for South African fisheries. The Sea Fishery Advisory Committee consists primarily of senior officials of the Department of Environment Affairs and appointees from the Fishing Industry. The Scientific Working Groups for different marine species groups are comprised of scientists from the Department's Sea Fisheries Research Institute, supplemented by invited specialist scientists from other organisations. Prior to late-1989 when these groups were first constituted, there was no formal basis for scientists other than those from the Department to contribute at this level. The Working Groups are directed to consider biological aspects only; economic and social implications of management decisions are taken into account only at a higher level.

survey estimates of biomass, as would be appropriate in the context of risk. The choice of a reference point related to K, rather than to some absolute spawning biomass level such as one million tonnes, was considered to be more appropriate from a biological viewpoint, appreciating nonetheless that estimates of K would change as the acquisition of further survey data led to updated estimates of R_{med} and M.

This process for scientific recommendation of TAC's continued smoothly through 1988 and for the first revision in 1989, with the recommendations provided (which were rounded to the nearest 50 thousand tonnes) being implemented without change by decision makers (see Table 2). However, problems started when the 1989 recruitment survey showed that the recruitment for that year was substantially below average (see Table 1 and Fig. 3). This meant that even had no catch been taken that year, the target of no more than a 10% chance of an escapement below $0.5K$ was unachievable. For the catch already taken by that time, this probability approached 75%. The decision was made to close the fishery immediately, even though the cumulative catch at that time of 280 thousand tonnes was less than the first revision of the TAC of 350 thousand tonnes.

This early closure in 1989 was not that problematic from the industry's viewpoint. A reasonable level of catch had been achieved, and their own vessels' experiences were consistent with the poor level of recruitment indicated by the June hydroacoustic recruitment survey. The real problems were to arise in 1990. For the first time, the full three-stage process described above was applied, with the following results.

1. Forward projection of the 1989 recruitment results indicated a 100 thousand tonnes TAC for a 20% chance of 1990 escapement less than $0.5K$ [prescription I above]. Decision makers opted to allocate an initial TAC of 150 thousand tonnes.

2. The November 1989 spawning biomass indicated that the situation was somewhat worse than had been projected previously, and that the TAC corresponding to prescription I was zero. The SFRI scientific Working Group structure (Fig. 5) was now operative, and the responsible Group recommended immediate closure of the fishery, but decision makers maintained the TAC at 150 thousand tonnes.

3. Two recruitment surveys were carried out during 1990, giving consistent results of a recruitment for that year almost as poor as that of 1989. Prescription II would have required a "return" of some 250 thousand tonnes of anchovy to the sea! The scientific Working Group recommended immediate closure more forcefully; decision makers retained the 150 thousand tonnes TAC unchanged.

Morals

A number of important inferences and lessons for the future are evident from this series of events.

- The decision makers really paid little attention to the biological reference point and probability level choices used as the basis for scientists' TAC recommendations. Though tacitly accepting these as the basis for setting TAC's while they gave "acceptable" answers, the decision makers readily discarded them once they "failed" to do so.

Table 2. Scientific recommendations for TAC's (units: thousand tonnes) for the South African anchovy over the period 1986–1991. The probabilities shown correspond to the chance assessed at that time that the TAC recommended would result in a spawning biomass below $0.5K$ at the end of that year. In cases where the final TAC allocated differed from the scientific recommendation, the allocation and associated probability are shown in parenthesis.

	Pre-season		Feb.–March revision^{+}		Late-season revision^{++}		
Year	TAC	Prob.	TAC	Prob.	Month announced	TAC	Prob.
1986	280				July	300	
1987	300		450	0.18*	July	600	0.08*
1988	300		500	0.18	July	600	0.07
1989	300		350	0.19	July	300^{+++}	0.75
1990	100	0.20	0	0.20	June	0	0.43
	(150)	(0.24)	(150)	(0.30)		(150)	(0.64)
					July	0	0.45
						(150)	(0.87)
1991	150		100		June	150	

* Calculated only after the event.
$^{+}$ Based on results from preceding November spawning biomass survey.
$^{++}$ Based on results of recruitment survey during current season.
$^{+++}$ corresponding to closure.

- Basing recommendations on some fixed level of biologically-defined "risk" appears an attractive and consistent approach to scientists. Yet during 1990, decision makers readily changed their acceptable "risk" level from the *a priori* 20% of escapement below $0.5K$, to successively 24%, 30%, 64% and 87% (see Table 2).

- One reason for these actions was that the responsible Minister and his advisory committee (see Fig. 5) were quite properly concerned with the economic viability of the industry and the social dislocations (unemployment) that would have resulted from a TAC lower than 150 thousand tonnes. This TAC represented a reduction of some 50% in the recent average, and 75% from that of two years before, so these decision makers certainly had taken cognisance of the poor recruitment indicated by the 1989 survey results and had shown that they were prepared to take quite drastic action. Nevertheless, it is of concern that successively poorer appraisals of resource status saw them unprepared to make any further TAC reduction whatsoever to partially compensate for the increasing risk being taken. [It is of interest that in a survey of industry views on priorities for pelagic resource management objectives carried out shortly beforehand (Stewart 1990), a low risk of severe resource depletion was accorded the highest priority, while (low) inter-annual TAC variability featured only well down (sixth) on the list. Behaviour in reality (as reflected above) suggests otherwise, and serves to illustrate the problem that preferences expressed in the abstract may bear little relation to those that become evident in a real situation.]

- Another probable reason for these actions was that the definition of "risk" used (probability of escapement less than $0.5K$) was not particularly meaningful to the decision makers — not in the sense that they were unable to understand it, but rather because they lacked comparable experiences to which to relate it and to contextualise the target probability levels chosen.

- What occurred in 1990 had not been unanticipated by scientists. Simulation studies (along similar lines to those reported in the next section) carried out beforehand, had indicated that application of prescriptions I and II would result in a zero TAC on an average of one year in four. [It is, of course, quite accidental that it was in the fourth year of application of the procedure that a zero TAC recommendation eventuated!] The series of decisions actually taken suggests that managers paid no real attention to these results. Presumably an overoptimistic attitude had been generated by the larger-than-expected anchovy spawning biomass revealed by the surveys, together with the good recruitments in 1986 and 1987 which were followed by much larger TAC's than in previous years. The fact that decision makers seem to have been unaware either of the questions which they should have asked in respect of prescriptions I and II, or that techniques were available (and had been applied!) to answer them, is indicative of an education need and a communication gap. These need to be better addressed in future to avoid the process of learning only by mistakes, as occurred in this case (partial progress has already been made in this regard with the constitution of scientific Working Groups — see Fig. 5 caption).

- The anchovy TAC reduction to 150 thousand tonnes in 1990 (and subsequently also in 1991) severely compromised scientists' credibility in the eyes of the industry. The argument advanced by some members of the industry was superficially both straightforward and damning. Industry had previously operated happily under a "steadyish" annual anchovy catch of some 300

thousand tonnes without problems. By advocating a doubling of the TAC for 1987 and 1988, scientists were seen to have been responsible for the overexploitation and resultant collapse of the resource. However, the stock-recruitment plot (Fig. 6) shows that the poor recruitment in 1989 was certainly not a consequence of a spawning biomass which had been heavily depleted as a result of excessive preceding catches. Furthermore, in all likelihood some of the pre-1984 catches were taken from a stock which had fluctuated to a low level following poor recruitment, so that these catches involved a high degree of risk; however, in the absence of surveys, this went unappreciated at the time. Thus this argument from (some quarters of) the industry is clearly flawed, but its attractive simplicity makes it difficult to counter.

The Present Management Procedure

The 1990 experience related above demonstrated quite clearly that continued application of prescriptions I and II to provide scientific recommendations for anchovy TAC's was not viable. The inevitable initial TAC recommendation of zero for 1991 would simply have been ignored by decision makers, who would then have been left to decide upon a TAC with little biological guidance. The prescriptions were in any case extremely risk-averse by international fishery standards, and an approach allowing for larger catches would not have been unsound biologically. However, simply redefining the probabilities or reference level in these prescriptions [20%/10% or $0.5K$ — see I and II above] introduced other problems. What alternative levels to choose, and on what basis? More importantly, a particular concern of the scientists involved was that these "risk" levels might be chosen by decision makers simply to give what they considered for other reasons to be a desirable level of catch for 1991 — this would have created a precedent for continued annual redefinition, with inevitable catastrophic longer term consequences for the conservation of the resource.

The selection of a trade-off between risk (to the resource) and reward (in terms of catches made) is ultimately one for the responsible politician, not scientists. Nevertheless, it is important (to avoid the catastrophic scenario alluded to above) that scientists attempt to ensure that this selection is made on an *a priori* basis related to longer term concerns, rather than be dominated by a short term consideration such as the TAC level for the forthcoming season. Accordingly, analyses were carried out to indicate the different trade-offs between competing objectives for alternative candidate management procedures (i.e., algorithms for calculating scientific TAC recommendations), for a twenty year planning horizon.

The simulations concerned thus involved a twenty year projection of the resource dynamics, commencing from the assessed status of the anchovy population at the end of 1990, under different scenarios for specifying future TAC's. They were carried out on a myopic Bayes basis, i.e., assuming that the present estimates and levels of uncertainty about stock status and population dynamics parameters (R_{med} and M) are not updated and improved with the acquisition of data from future surveys. Such future data are thus used only in catch control law formulae for TAC's, and not for re-estimation purposes; this contributes a positive bias to the risk levels assessed. The Monte Carlo estimation procedure of Appendix A provides a numerical realisation of the joint probability distribution required for the computations. This distribution automatically incorporates the covariances between the estimates of the quantities pertinent to the projections. Given that the past recruitment series shown in Fig. 3 is suggestive of positive serial correlation (the point estimate for which is about 0.35), the estimated serial correlation for each Monte Carlo realisation of an historic recruitment series was used for the forward projection of that particular series, rather than ignoring possible future correlation.

A key factor in these calculations is the stock-recruitment function used. This was of the form:

$$R_y = \begin{cases} R_{\text{med}} \exp(\varepsilon_y) & \text{if } D_{y-1} > D_{\text{thr}}; \\ \left(\frac{D_{y-1}}{D_{\text{thr}}}\right) R_{\text{med}} \exp(\varepsilon_y) & \text{if } D_{y-1} \leq D_{\text{thr}}. \end{cases}$$

where $D_y = B_{\text{sp}}(y)/K$,

D_{thr} is a threshold level of depletion (D) below which median recruitment becomes less than R_{med},

$B_{\text{sp}}(y)$ is the spawning biomass at the end of year y, and

ε_y is a random variate from a normal distribution with zero mean and variance $\sigma_R = 0.4$ (see Appendix A).

This relationship (ignoring the stochastic component) is illustrated in Fig. 7 for the baseline selection for computations of $D_{\text{thr}} = 0.2$.

The results of computations of this nature must always be tested for robustness to modifications of the parameter values and assumptions of the underlying model of resource dynamics. Amendments to this stock-recruitment relationship were found to be by far the most influential of such modifications (more so than increasing the serial correlation in recruitment estimated from the data, for example). Figure 7 therefore also shows an alternative relationship (with $D_{\text{thr}} = 0.5$) for which computations were repeated.

The computations assumed unbiased estimates of (absolute) biomass to be available from both the spawning biomass and recruitment surveys, each with a coefficient of variation of 25% (a little less precision than has been achieved historically — see Table 1; the calibration provided by past egg-production surveys allows the hydroacoustic survey results to be considered as estimates of absolute abundance, despite uncertainty concerning target strength). An initial TAC would be set on the basis of the spawning biomass estimate, assuming (if required by the management procedure under investigation) that the forthcoming year's recruitment would be equal to its median value. This calculation would be updated during the simulated season once the recruitment survey estimate became available.

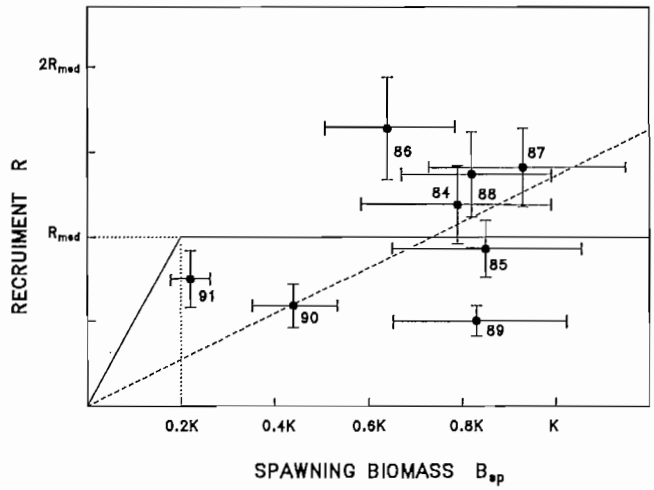


Fig. 6. The anchovy recruitment and spawning biomass estimates of Fig. 3 shown in the form of a stock-recruitment plot. Note that the point plotted for year n refers to recruitment in year n and spawning biomass at the end of year $n-1$. The error bars reflect ± 1 standard error. The solid line shows the estimate of median recruitment (R_{med}), and the form of the stock-recruitment relation assumed for projection purposes ($D_{\text{thr}} = 0.2$). The dashed line corresponds to the mean recruits per unit spawning biomass ratio, which was used to predict recruitment for the early-season revision of the 1991 anchovy TAC (see text).

Typically, the anchovy fishing season runs over no more than eight months: January–August. In the first three of these months, the simulations assumed that 30% (the historical average) of the initial TAC would be caught, and that this would comprise 1-year-old anchovy only. In practice, the time required for analysis of the recruitment survey data and the decision making process, hardly permits announcement of a TAC revision before the end of June. Thus it was assumed further that 60% of the remainder of the initial TAC would have been caught (as 0-year-olds) before the TAC was revised on the basis of the recruit survey estimate (i.e., that, in total, 72% of the initial TAC would be caught before this revision). Accordingly, situations would arise in the simulations where more than the TAC finally desired had been caught before the revised value was known (i.e., whenever $\text{TAC}_{\text{final}} < 0.72\,\text{TAC}_{\text{initial}}$). Since such situations can arise in reality for this fishery (and, indeed, did so in 1989 and 1990), the evaluation of the performance of alternative candidate management procedures must take their occurrence into account.

This late announcement of the TAC could, in practice, also lead to situations where an increase proved too large for the industry to be able to catch it, given the poor weather conditions customary at that time of the year. However, no attempts have as yet been made to take account of this factor in the simulations.

In most instances, long-term management of a renewable marine resource has three general aims:

1. to maximise the total catch to be made over a period of time,
2. to minimise the chance that harvesting unintentionally depletes the resource to a level at which future recruitment success is impaired (i.e., that the resource "collapses"), and
3. to minimise variation in TAC's from one year to the next for reasons of industrial and social stability.

These aims are mutually exclusive, so that some trade-offs have to be accepted. To illustrate these trade-offs as clearly as possible, summary statistics of the results of the computations need to be chosen which reflect the performance of a candidate management procedure in relation to these aims. Those chosen in this case were:

- $\overline{C}$ - the average annual catch over the twenty year time horizon,
- "Risk" - the probability that the spawning biomass B_{sp} drops below $0.2\,K$ at least once within this time horizon, and
- V - the mean annual change in TAC as a proportion of $\overline{C}$.

The harvesting strategies for which these statistics were computed were constant catch, constant escapement and constant proportion, together with upper and lower bounds on the size of the TAC, and restrictions on the extent to which it might change from one year to the next, as described in more detail below.

Results

Selected results of the computations which were carried out, are shown in Table 3 and Fig. 8. The table reports values of a statistic for mean escapement over the projection period ($\overline{B_{\text{sp}}/K}$), as well as the three summary statistics defined above.

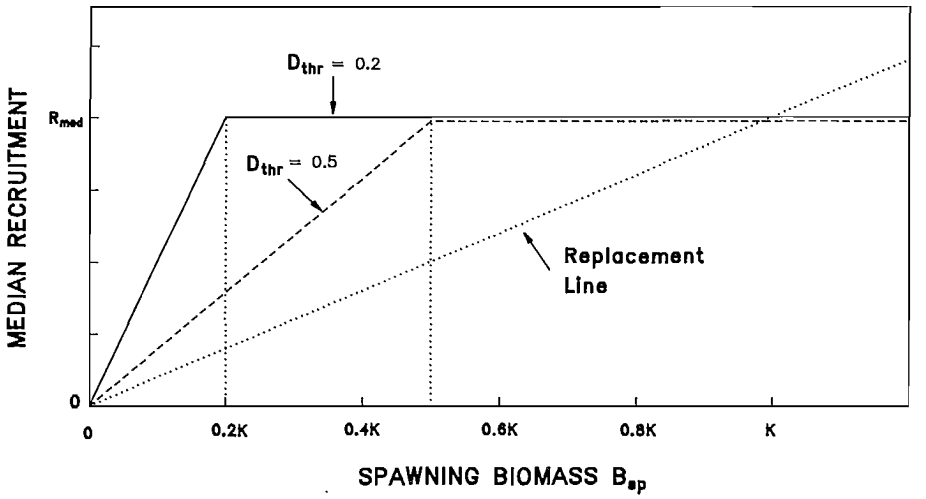


Fig. 7. The solid line shows the underlying stock-median recruitment relationship (i.e., ignoring the log-normal stochastic contributions) which was assumed for testing the performance of alternative candidate anchovy management procedures. This form has $D_{\text{thr}} = 0.2$. An alternative form with $D_{\text{thr}} = 0.5$, which was used for testing robustness, is shown by the dashed line.

The harvesting strategy adopted between 1987 and 1990 for the anchovy resource closely approximates constant escapement. Somewhat more loosely, the strategy before that time might be categorised as constant catch. Figure 8 contrasts the performances of these two strategies in terms of "risk" plotted against average catch ($\overline{C}$), for various choices of target escapement and constant catch respectively. The reason that these "risk-versus-reward" curves bend back on themselves is that target choices which lead to large catches being taken in the first few years, rapidly deplete the resource below D_{thr}. Future recruitment success is then impaired, so that only small catches can be taken for the balance of the projection period; consequently as the "risk" level increases, the average catches fall.

The 1987–1990 strategy was virtually constant escapement with a target of $0.7K$. Both Fig. 8 and Table 3 show that a constant catch strategy with a similar "risk" level achieves about one third less catch on average than this constant escapement strategy, so that the latter seems the superior choice. However, Fig. 1 hides the fact that this higher catch can only be achieved at the expense of substantial variation in TAC levels from one year to the next — by some 90% on average.

The constant escapement strategy also occasionally allocates very high TAC's — well beyond the current annual processing capacity of the industry, which probably is not much above 600 thousand tonnes. Placing this upper bound on TAC's decreases average catches somewhat as would be expected, but also decreases risk dramatically; however, the inter-annual TAC variability remains high at some 60%.

Industrial and socio-political concerns about upward and downward changes to TAC's are usually not symmetric. There is seldom much resistance to a proposed TAC increase. [It is interesting to note that while one industry spokesman had claimed in earlier years that TAC changes exceeding 15% (in either direction) could not be accommodated, the industry offered no objection to the 100% anchovy TAC increase of 1987.] In contrast, recommendations to decrease a TAC are usually difficult to implement in practice. Unless this aspect of reality is captured in simulations, an overoptimistic impression can be given of the risk level associated with a particular management procedure. For this reason, further computations were carried out placing no bound on the extent to which the TAC could be increased upwards in any year (except for the 600 thousand tonnes limit), but restricting any decrease from one year to the next to a maximum of 25%. Table 3 indicates that this results in improvements in both average catch and catch variability for the constant escapement strategy, but at the expense of an increase in risk.

The next question which arises is whether other harvesting strategies can provide improved performance overall to that of the constant escapement strategy (with the constraints discussed above). The alternative investigated was a "constant proportion" strategy, where the target proportion shown in Table 3 pertains to a TAC set as a proportion of the estimated escapement in the absence of any catch that year. For a target proportion of 25%, this strategy dominates the corresponding constant escapement strategy, manifesting both lower "risk" and lower TAC variability for the same average catch.

Table 3 shows some further results for the constant proportion strategy. A lower target proportion of 20% secures a lower risk at the expense of a decreased average catch. The incorporation of a lower bound of 150 thousand tonnes makes negligible difference to performance statistics. (This choice for a lower bound was based on the experiences of 1990 related above, which suggested that the pelagic industry would find extreme hardship in any allocation less than this tonnage.)

Performance statistics change markedly if the underlying stock-recruitment relationship differs from that assumed for the baseline calculations ($D_{\text{thr}} = 0.2$). For $D_{\text{thr}} = 0.5$, average catch falls; but, more importantly, "risk" as defined increases substantially. The final set of results shown in Table 3 indicates the advantage to industry of greater flexibility, in the sense of an ability to accommodate greater proportional TAC reductions. If annual decreases of up to 40%, rather than 25%, are acceptable, average catches can be improved by 7%.

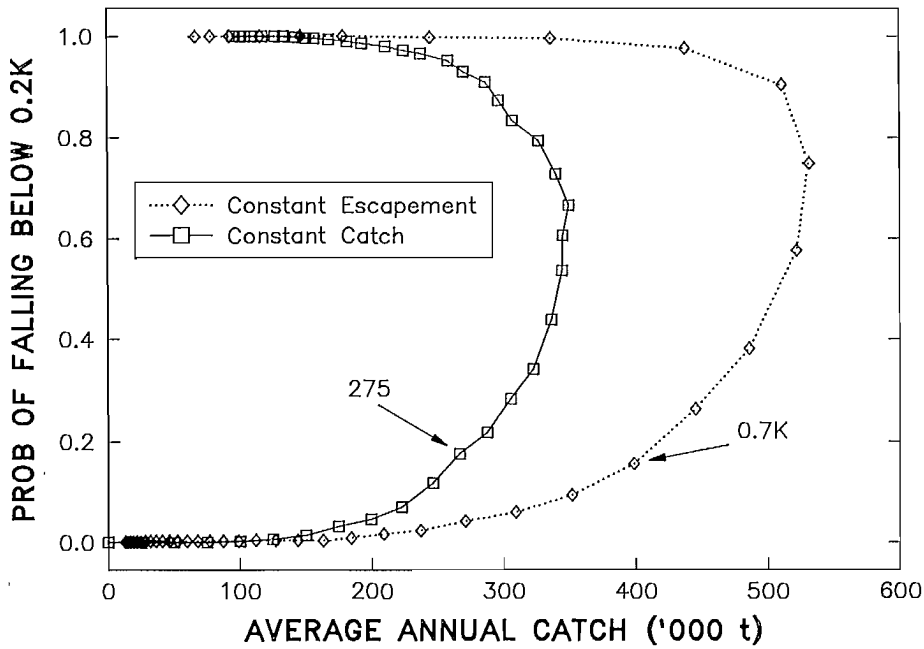


Fig. 8. Plots of "risk" against average annual catch ($\overline{C}$) in thousand tonnes for various target levels for the constant catch and constant escapement harvesting strategies. The results for the latter correspond to evaluations without restrictions on either the size, or the change from one year to the next, of the TAC. The points with arrows correspond to the management procedures whose performance statistics are listed in the first two rows of Table 3.

Implementation

The revised procedure for scientific TAC recommendations for the South African anchovy was selected essentially on the basis of the results shown in Table 3. The procedure chosen was that shown in the last line but three of that table. The underlying rationale for the choice was along the following lines.

1. The results indicated that the constant proportion strategy dominates the constant escapement strategy (at least in the region of target space of interest, and subject to the constraints of a 600 thousand tonnes maximum TAC and a maximum annual TAC reduction of 25%).

2. While the "risk" level associated with a target proportion of 25% appeared acceptable given the baseline choice for a stock-recruitment function ($D_{\text{thr}} = 0.2$), this level increased markedly for robustness tests with a higher D_{thr} value. Accordingly, a target proportion of 20% was preferred.

3. The minimum TAC of 150 thousand tonnes, and the other specifications, were accepted subject to an "except in special circumstances" caveat.

This selection was made by the Director of the Sea Fisheries Research Institute in the light of advice received from his scientific Working Group (see Fig. 5). The alternative options were presented to the Sea Fishery Advisory Committee, but seem not to have received close attention. Thus ratification of this revised management procedure by this Committee and the responsible Minister appears to be implicit rather than explicit, so that an element of vagueness about its formal status remains.

The first real application of the revised procedure took place after analysis of the results of the November 1990 spawning biomass survey. Following continued poor recruitment in 1990, the procedure indicated a TAC of 170 thousand tonnes for the whole season, which satisfied the "guarantee" [see 3 above] of a minimum 150 thousand tonnes TAC. However, scientific reservations were expressed concerning such a recommendation. The point estimate of escapement at the end of 1990 was $0.22K$, uncomfortably close to the $0.20K$ level which the management procedure was designed to avoid, and sufficiently low to suggest to the scientists in the Working Group responsible that recruitment strength for 1991 might be impaired.

This was considered sufficiently serious by those scientists to merit invoking the "except in special circumstances" caveat — despite thereby risking scientific credibility in the eyes of decision makers, by recommending departure from newly "adopted" procedure on its virgin implementation. The *ad hoc* approach applied in these circumstances was to estimate the 1991 escapement using the mean (R/B_{sp}) ratio [analogous to the F_{med} approach (Beverton 1984) — see also Fig. 6], rather than R_{med}, to estimate anticipated recruitment for 1991 by multiplying the mean ratio by the estimate of B_{sp} for November 1990. A resultant point estimate of a 100 thousand tonnes TAC (to result in 1991 escapement no worse than that in 1990) was recommended and subsequently allocated.

Fortunately, the 1991 recruitment survey indicated improved recruitment over that of the two previous years (see

Table 3. Performance statistics for alternative management procedures for the South African anchovy resource. The method used for the calculation is described in the text. The statistics are defined as follows:

$\overline{C}$ - average annual catch over the next 20 years (in thousand tonnes),

Risk - probability that $B_{sp} < 0.2K$ at least once within these 20 years,

$\overline{B_{sp}/K}$ - average spawning biomass level over the 20 years, and

V - mean annual change in TAC as a proportion of $\overline{C}$.

D_{thr} is the value of B_{sp}/K below which median recruitment decreases as B_{sp} falls (see Fig. 7), and the table shows the numerical value which was assumed for the results shown. Entries (in thousand tonnes) under "Max." and "Min." indicate whether bounds have been placed on the TAC, and "Max. reduction" reflects a constraint on the proportion by which the TAC can be reduced from one year to the next (there is no similar constraint on an increase).

		TAC				Performance statistics			
Strategy	Target	Max.	Min.	Max. reduction	D_{thr}	$\overline{C}$	Risk	$\overline{B_{sp}/K}$	V
Constant catch	275	–	–	–	0.2	267	0.18	0.71	0.04
Constant escapement	$0.7K$	–	–	–	0.2	398	0.16	0.61	0.91
	$0.7K$	600	–	–	0.2	317	0.00	0.69	0.61
	$0.7K$	600	–	25%	0.2	372	0.12	0.63	0.31
Constant proportion	0.25	600	–	25%	0.2	373	0.09	0.64	0.23
	0.20	600	–	25%	0.2	314	0.04	0.70	0.24
	0.20	600	150	25%	0.2	314	0.04	0.70	0.24
	0.20	600	150	25%	0.5	250	0.40	0.54	0.26
	0.25	600	150	25%	0.5	241	0.63	0.39	0.30
	0.225	600	150	40%	0.2	337	0.04	0.67	0.26

Table 1 and Fig. 3). This provided a basis to withdraw the "special circumstances" reservation and revert to the new management procedure for the final TAC recommendation. The resultant figure of 150 thousand tonnes was subsequently allocated.

Future Improvements

There are three aspects of the current management procedure that are earmarked for further investigation in the near future.

- Re-evaluation of performance statistics for the procedure on a feedback-control (i.e., annual re-estimation of parameter values) rather than a myopic Bayes basis. The computationally intensive Bayes-like estimation procedure will be replaced by a simpler least squares approach for this purpose (as there is now a time series of survey data that is sufficiently long to render this approach viable).

- Development and simulation in testing of decision rules within the management procedure which specify *a priori* how to react when low spawning biomass levels occur (as at the start of 1991), or surveys have coefficients of variation exceeding the 25% assumed for simulation tests of the procedure, so as to avoid invoking "special circumstances" with *ad hoc* fixes.

- Improving interaction between scientists on the one hand, and decision makers and members of the industry on the other, in the process of developing future versions of the procedure. The results of stochastic simulations form the essential basis for selecting management procedures for highly variable pelagic resources, so that scientists need to develop methods of presenting these results to laymen in a manner that makes them more easily comprehended. In the first instance, scientists need to explain the current procedure and its likely implications better, and to obtain feedback on whether a summary performance statistic such as V (which measures the extent to which the TAC varies) is capturing industrial concerns in this regard in the most effective manner. Such enhanced interaction should lead to an improved process of choice between candidate updated procedures to be put forward in the future.

Rigorous independent quantification of socio-economic factors has been suggested as a means to facilitate the process of choice between alternative procedures in the future. However, this encounters the problem that individual companies consider some of the key information required for such an exercise as necessarily commercially secret. Initial reaction from members of the industry is a preference for the presentation of options (performance statistics for a number of candidate procedures), which would allow them to perform the economic calculations themselves.

Risk Evaluation and Biological Reference Points

The most important aspect of risk would seem to be finding an appropriate definition, particularly one that is more immediately meaningful to intelligent laymen (i.e., the typical decision makers in the fisheries management process). Experience with the South African anchovy fishery indicates that expressing risk as "the probability of depletion below a specified fraction of B_{sp}/K" may not be the most useful approach. Perhaps presentation analogous to the commonly used, and presumably meaningful, "bridge built to withstand a one in n years flood" approach should be attempted.

Nevertheless, in general, scientists' technical calculations will still revolve around addressing the issue: "a probability of less than $x\%$ of reducing the resource below $y\%$ of K within a period of z years" [cf. Beddington and Cooke (1983), who choose $x = 10$, $y = 20$ and $z = 20$ when presenting their results]. The problem here is that three quantities need to be specified, rather than the single number required to designate within a set of deterministic strategies (such as $F_{0.1}$ within the $F_{0.n}$ set of harvesting strategies). Ultimately the justification or otherwise for such strategies is that they "work" in practice, i.e., they allow reasonable levels of catch without collapsing more than a small proportion of the stocks to which they are applied. Future evaluation of the appropriateness of stochastic-evaluation-based strategies would be greatly assisted by a common international convention for the choices of x, y and z.

A particular message of this paper is that biological reference points such as $F_{0.1}$ and F_{med}, in the sense that they are conventionally applied to develop scientific recommendations for fisheries management, are at best of interim value, and irrelevant in the longer term. This outlook has its roots in the process adopted by the International Whaling Commission (IWC 1989a,b) for the development of a revised management procedure for baleen whale stocks. Essentially, this views the aspects of estimation ("assessment"), estimation error, and the formulae used to set, say, TAC's as inextricably interwoven. There is therefore little sense in attempting to separate the components and debate their individual merits. Rather, the management procedure (data-estimator-catch control law) can only be evaluated in its totality, as a feedback control process implemented over a number of years, rather than for a single year only (in particular because risk is usually only meaningful for a multi-year rather than a single year time horizon). Thus procedure selection is related to concepts more readily meaningful to decision makers, such as average catch and inter-annual catch variability. The biological reference points that perhaps provided the motivation for some of the catch control laws that were investigated, retain no essential role and can be discarded — all that matters is medium-term procedure performanee in terms of selected summary statisties. The long term scientific goal for all the major South African commercial fisheries is therefore to convert the basis for their management to one similar to that now used for the anchovy.

Where (in a different sense) biological reference points do have a role, would seem to be in the definition of risk. The really important question here would seem to be: "Below what level of B_{sp}/K should one become concerned about recruitment overfishing?". Medium-term summary statistics from feedback control simulations can be directly related to this question. Management procedure selection for the South African anchovy, and most other fisheries, depends crucially on the answer to this question: 0.2?, 0.5? — what is the accumulated wisdom from international experience? A lower value than 0.2 would mean that the current choice of management procedure for the anchovy reflects unnecessary resource wastage — a rather higher value that it involves irresponsible risk-taking. Linked to this is the question of which is the most appropriate baseline choice of the underlying stock-recruitment function for the simulations; and further, what range of robustness tests concerning this choice requires exploration?

Finally, what is the future role of the standard annual fisheries assessment exercise that underpins the setting of TAC's in so many fisheries, in a management procedure context? In the present anchovy management procedure, this complex process is replaced by a simple linear relationship between TAC and the point estimate of abundance from the most recent hydroacoustic survey (provided that this meets certain precision requirements). The key aspect of the estimator component of a management procedure is not that it produces the "best" estimate of the real status of the resource, but that it has been tested to be robust against the wide range of the uncertainties inevitable in the assessment process. But robustness requirements impact procedure "efficiency" — to achieve a specified risk level, the wider the range of robustness requirements, the lower in consequence are the catches achievable from the resource (i.e., the lesser the efficiency).

The increasing data base that contributes to a time-series of assessments serves to reduce the extent of uncertainty about the resource, i.e., to place greater limits on the robustness requirements and therefore to make more efficient management procedures possible. Management procedures should not be set in stone, but rather redeveloped typically every three to five years, i.e., at an interval which allows sufficient time for notable improvements to the state of knowledge of the resource. In this context then, the future role of assessments is not to provide TAC recommendations through linkage to biological reference points, but rather to condition the underlying models used for the feedback-control simulation studies, whose results provide the basis for selection of management procedures.

Conclusions

The key implications of the experiences gained in attempting to develop a management procedure for the South African anchovy resource seem to us to be as follows.

- An international convention for the definition of risk in the context of simulation testing of management procedures for fisheries (i.e., a stochastic analog of, for example, $F_{0.1}$) would be valuable. An important component of this definition is related to the question: at what level of depletion of spawning biomass below average pristine size, is the chance of successful recruitment likely to become substantially impaired?

- Unless risk evaluations are presented to layman decision makers in terms of statistics found readily meaningful, they will be at best misinterpreted, and at worst ignored.

- The likely extent of TAC variability from one year to the next is a more important performance attribute for a management procedure than responses in the abstract from decision makers might indicate.

- TAC's should ultimately be set by means of simulation tested feedback-control management procedures, not annual "best" assessments linked to biological reference points (a practice which can tend to exacerbate TAC fluctuations and delay the decision making process without compensating benefits, and which also frequently fails to take account of precision and uncertainty considerations). Biological reference points are relevant rather to performance criteria under the simulation tests.

- The future role of assessments should not be to provide the basis for annual TAC decisions. Rather, it is to condition the range of underlying models used to simulation test candidate management procedures for a particular resource, as such procedures are reviewed and revised at (typically) three to five year intervals.

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Appendix A

The Assessment Procedure Used for the South African Anchovy

The assessment procedure used to obtain the estimates of annual recruitment and spawning biomass shown in Fig. 3 is an extension of that described by Bergh and Butterworth (1987). This (extended) procedure was used to obtain the probability estimates shown in Table 2, on which scientific recommendations for TAC's over the 1987–1990 period were based, and also provided the distributions for the parameters of the underlying model used for myopic Bayes simulation testing of the constant proportion management procedure now used to provide scientific TAC recommendations for this resource.

The model underlying the assessment procedure makes the following assumptions:

1. the adult stock consists of 3 age classes,
2. all fish have a November birth date,
3. fishing is approximated by two pulses — one in February comprising fish of 15 months of age only, and one in May made up entirely of recruits of 6 months of age, and
4. recruitment is log-normally distributed about a median value (R_{med}) which is unaffected by the size of the spawning biomass.

[Assumption 4 is questionable for 1991, following the low spawning biomass at the end of 1990. However, these most recent survey data were not incorporated in the calculations to estimate R_{med}.]

The equations describing this model are as follows:

$$N_{y+1,1} = \left[R_{\text{med}} \exp(\varepsilon_y)\, e^{-M/2} - \frac{C_{y,0}}{w_c(y)}\right] e^{-M/2}$$

(A1)

$$N_{y+1,2} = \left[N_{y,1} e^{-M/4} - \frac{C_{y,1}}{w_{1c}(y)}\right] e^{-3M/4}$$

$$N_{y+1,3} = N_{y,2}\, e^{-M}$$

$$B_{\text{sp}}(y) = \sum_{i=1}^{3} w_i(y+1)\, N_{y+1,i}$$

where

$N_{y,a}$ is the number of anchovy of age a at the "start" (in November) of year y,

R_{med} is the median recruitment,

ε_y is distributed according to a normal distribution with zero mean and variance σ^2,

M is the natural mortality (independent of age and year),

$C_{y,0}$ is the catch by mass of 0-year-olds (assumed equal to the April+ catch) in year y,

$C_{y,1}$ is the catch by mass of 1-year-olds (assumed equal to the January–March catch) in year y,

$w_c(y)$, $w_{1c}(y)$ are the average masses of the 0 and 1-year-olds, respectively, caught in year y,

$w_1(y)$, $w_2(y)$, $w_3(y)$ are the average masses of 1, 2 and 3-year-old anchovy, respectively, at the "start" of year y (i.e., in trawl samples taken during the November survey), and

$B_{\text{sp}}(y)$ is the spawning biomass at the "end" of year y (i.e., the "start" of year $y + 1$).

Thus the recruitment in year y is:

$$R_y = R_{\text{med}} \exp(\varepsilon_y) \qquad \text{(A2)}$$

The motivation underlying the Bayes-like estimation procedure used is as follows. Say prior distributions $p_i(a)$ and $p_i(b)$ are available for two quantities a and b. Such priors are provided, for example, by the survey estimates of anchovy biomass and their associated standard errors. Say further that there is known to be a relationship between the true values of a and b: $f(a, b) = 0$ — analogous in this case to the population dynamics relationships of equations (A1). The estimation algorithm then provides a joint posterior distribution for a and b [$p_f(a)$ and $p_f(b)$ — which are correlated, unlike $p_i(a)$ and $p_i(b)$] as that subspace of the joint prior distribution consistent with the $f(a, b) = 0$ constraint. For the simple case $f(a, b) = a - b$, it is easily shown that this prescription is equivalent to the maximum likelihood estimation result of taking an inverse-variance-weighted average. The scheme generalises readily given additional quantities and/or relationships.

A numerical implementation of this scheme is as follows. First, generate a set of data pairs $S = \{(a_j, b_j) : j = 1, \ldots, N\}$ at random from the prior distributions $p_i(a)$ and $p_i(b)$. A numerical realisation of the joint posterior distribution is then provided by the subset of S for which $f(a_j, b_j) = 0$ (within an appropriate tolerance, for practical purposes).

While the numerical implementation is more readily explained in this way, an equivalent and computationally far more efficient method of generating the posterior distribution

is as follows. First generate a_k at random from $p_i(a)$, and solve the equation $f(a_k, b_k) = 0$ to obtain b_k. A numerical representation of the joint posterior distribution is then provided by the set:

$$S^* = \{[(a_k, b_k);\ w_k] : k = 1, \ldots, N^*\}$$

where $w_k = p_i(b_k)$ is the weight given to the (a_k, b_k) pair.

This estimation scheme is applied to the problem at hand in the following way. Equations (A1) are rewritten:

$$\begin{aligned} B_{sp}(y) \quad = \quad & \left[R_{med}\, e^{\varepsilon_y} e^{-M/2} - \frac{C_{y,0}}{w_c(y)}\right] e^{-M/2} w_1(y+1) \\ & \times \left[\left\{R_{med}\, e^{\varepsilon_{y-1}} e^{-M/2} - \frac{C_{y-1,0}}{w_c(y-1)}\right\} e^{-3M/4}\right. \\ & \left. - \frac{C_{y,1}}{w_{1c}(y)}\right] e^{-3M/4} w_2(y+1) \\ & + \left[\left\{R_{med}\, e^{\varepsilon_{y-2}} e^{-M/2} - \frac{C_{y-2,0}}{w_c(y-2)}\right\} e^{-3M/4}\right. \\ & \left. - \frac{C_{y-1}}{w_{1c}(y-1)}\right] e^{-7M/4} w_3(y+1) \end{aligned} \tag{A3}$$

A composite parameter vector $[B_{sp}(y)_k;\ \varepsilon_{y,k};\ M_k; \ldots; w_3(y+1)_k]$ is generated at random from the prior distributions for these quantities. Equation (A3) is then solved for $R_{med,k}$. This process is continued using equation (A1) with y increased to $y+1$. Requisite values of new parameters are generated from their priors, but since a value for $R_{med,k}$ is already available, the equation is solved instead for $\exp(\varepsilon_{y+1,k})$. The weight associated with the composite parameter vector for the joint posterior distribution is then the value of the prior for ε_{y+1} : $w_k = p_i(\varepsilon_{y+1,k})$. [Naturally, $w_k = 0$ if $\exp(\varepsilon_{y+1,k}) < 0$.] Extending to the surveys for subsequent years, an overall weighting factor is developed:

$$\tilde{w}_k = p_i\left(\varepsilon_{y+1,k}\right) \times p_i\left(\varepsilon_{y+2,k}\right) \times \cdots \tag{A4}$$

Additional information is incorporated by further multiplicative modifications to this overall weighting factor. For example, say $B_{sp}(y)_k$ was derived from the hydroacoustic survey result for that year. If the egg-production survey for the same year provided an estimate $B_{egg}(y)$ with standard error $\sigma_{egg}(y)$, the modification is then:

$$\tilde{w}_k \rightarrow \tilde{w}_k \exp\left[\frac{\{B_{egg}(y) - B_{sp}(y)_k\}^2}{\{2\sigma^2_{egg}(y)\}}\right] \tag{A5}$$

Biomass estimates from recruitment surveys, and estimates of the 1-year-old proportion of the spawning stock from the spawning biomass survey samples, are taken into account in a similar manner. Estimates known to be biased in a particular direction can also be incorporated in this structure. For example, if the egg production survey just referenced is known to be negatively biased (because of incomplete areal coverage perhaps), then:

$$\tilde{w}_k \rightarrow \begin{cases} \tilde{w}_k & \text{if } B_{sp}(y)_k \geq B_{egg}(y)_k; \\ 0 & \text{if } B_{sp}(y)_k < B_{egg}(y)_k. \end{cases} \tag{A6}$$

where $B_{egg}(y)_k$ is generated as a normal random variate with mean $B_{egg}(y)$ and variance $\sigma^2_{egg}(y)$.

The scale of the eventual posterior estimates of biomasses is controlled primarily by the egg production survey results, which are taken to be unbiased estimates of absolute abundance. The prior distributions for the results of the hydroacoustic surveys incorporated judgments by the scientists responsible for these surveys of the likely range of uncertainties in target identification, fish missed near the surface, calibration of acoustic equipment and target strength estimates, as well as the estimated survey sampling error. The target strength uncertainty alone is so large — effectively a [0.5, 2.0] multiplicative range — that the hydroacoustic estimates really function only as relative rather than absolute measures. In fact, the combination of the potentially high bias, but low variance hydroacoustic estimates, with the low (assumed zero) bias but high variance egg production estimates for the spawning biomass surveys [see Table 1, and also Armstrong et al. (1988), Hampton et al. (1990) and Shelton et al. (1993)], provides the desirable end result of both low bias and low variance. (The target strength uncertainty for the hydroacoustic surveys is termed a bias because the true target strength, though unknown, is the same for all surveys. Thus the results of these surveys provide a precise index of relative abundance.)

The recruitment surveys are taken into account allowing for the catch of recruits before the survey, uncertainties in separating the adult and recruiting components of the fish distribution, and ranges of likely negative bias because the recruitment process is not complete at the time of the survey. In principle, different natural mortalities for recruits and 1+ anchovy could be estimated, but this was not attempted as any difference in these estimates is confounded with the extent of this last-mentioned bias, for which specifications for the possible range of the uncertainty are rather poorly based. The prior distribution assumed for M is uniform over the range $[0.5, 1.5]\text{yr}^{-1}$.

The prior distribution for the log-recruitment fluctuation variables ε_y is chosen to be normal with zero mean and variance σ^2_R.

The assessment was implemented with $\sigma_R = 0.4$, based on values for other species listed in Beddington and Cooke (1983), although the estimation procedure could also be applied with σ_R generated from some prior distribution.

This assessment procedure provided the following means and standard deviations (in parenthesis) of posterior distributions for key population dynamics parameters:

$$\begin{aligned} M &= 1.33\ (0.11)\ \text{yr}^{-1} \\ R_{med} &= 271\ (51)\ \text{billion}\ (10^9) \\ \sigma_R &= 0.42\ (0.05) \\ K &= 1356\ (230)\ \text{thousand tonnes.} \end{aligned}$$

How Much Spawning per Recruit is Enough?

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In order for fish stocks to persist, successive generations must replace each other on average. This means that fishing should not reduce the egg production or amount of spawning per recruit (SPR) below a threshold level that is necessary for replacement. For each fishery under Federal management, the United States requires definitions of overfishing that at the least guard against recruitment overfishing; 60% of these definitions are based on threshold levels of SPR, which have typically been chosen arbitrarily from the range 20–35% of the level that would occur if there were no fishing (referred to as the %SPR). Threshold replacement levels can be estimated from spawning-recruitment (*S-R*) data, using a conservative hypothesis that the median of the observed survival ratios is an estimate of the maximum average survival ratio. A survey of 91 sets of *S-R* data for fisheries resources of Europe and North America indicated that the replacement %SPR averaged slightly less than 20% overall. Some species (e.g., Atlantic cod and most flatfish) exhibit consistently low levels of replacement %SPR (high resilience to fishing), while the smaller gadoids and many of the small pelagic species have values as high as 40–60% (low resilience). Replacement %SPR was positively correlated with natural mortality and negatively correlated with both the maximum average body weight and the body weight at 50% maturity. Replacement %SPR is recommended as a reference point for defining overfishing since it is based on the premise of stock replacement, it can be estimated from standard stock assessment information and, where such information is lacking, taxonomic affiliation and life history parameters can be used to select preliminary estimates.

Pour que les stocks de poissons persistent, les générations successives doivent se remplacer les unes les autres en moyenne. Cette observation signifie que la pêche ne doit pas faire descendre la production d'oeufs ou la ponte par recrue (SPR) au-dessous du seuil nécessaire pour assurer le remplacement. Pour chaque pêche gérée par le gouvernement fédéral, les États-Unis exigent des définitions de la surpêche qui au moins assurent une protection contre la surpêche du recrutement; 60 % de ces définitions reposent sur des seuils de ponte par recrue qui ont été en général choisis arbitrairement dans la plage de 20 à 35 % du niveau qui existerait en l'absence de pêche (désigné comme % SPR). On peut estimer les seuils de remplacement à partir des données de ponte-recrutement, en appliquant une hypothèse prudente selon laquelle la médiane des taux de survie observés est une estimation du taux maximum moyen de survie. Une étude de 91 ensembles de données de ponte-recrutement pour les ressources halieutiques d'Europe et d'Amérique du Nord a montré que le %SPR de remplacement s'établissait en moyenne à un peu moins de 20 % dans l'ensemble. Chez certaines espèces (par exemple, la morue atlantique et la plupart des poissons plats), on observe constamment de faibles taux de %SPR de remplacement (forte résilience à la pêche), tandis que les gadidés de taille plus petite et de nombreuses espèces pélagiques de petite taille ont des valeurs qui peuvent atteindre 40 à 60 % (faible résilience). Le %SPR de remplacement présentait une corrélation positive avec la mortalité naturelle et une corrélation négative avec le poids corporel maximal moyen et le poids corporel à maturité de 50 %. Le %SPR de remplacement est recommandé comme point de référence pour définir la surpêche puisqu'il repose sur l'hypothèse du remplacement du stock; il peut être estimé à partir de données sur l'évaluation du stock normalisée et, en l'absence de ces données, des paramètres d'affiliation taxonomique et de cycle biologique peuvent être utilisées pour choisir les estimations provisoires.

Population persistence requires that successive generations replace or surpass each other on average. This means that year-classes must produce sufficient spawning units (usually expressed in terms of biomass or eggs) per recruit (SPR) over their lifespan to correspond to the average number of recruits (R) produced by a unit of spawning (S). The observed ratio of R/S is referred to as the "survival ratio", s. Thus, persistence requires that SPR $\geq 1/s^*$, where s^* is the average value of s. It is only recently that spawning per recruit analysis has begun to be calculated routinely and SPR criteria have begun to be given explicit consideration in fisheries management objectives. Yet in the last few years, SPR has become the most common basis of overfishing definitions for U.S. marine fisheries.

In spite of the actual and potential importance of SPR criteria as a basis of fisheries management, relatively little attention has been given to the question of "how much spawning per recruit is enough?". There are considerable data from a wide variety of fisheries that can be used to answer this

question. This paper reports the most comprehensive compilation and analysis of such data to date. As background, it also reviews recent publications and events that have focused attention on the use of SPR criteria for defining overfishing.

Background

Development of SPR Reference Points

Spawning per recruit (SPR) analysis is a simple extension of the "dynamic pool" model used to calculate yield per recruit (YPR). Beverton and Holt (1957) give the classic derivation of the model, and Gabriel et al. (1989) specify the computational details. While maximum YPR usually occurs at an intermediate fishing intensity, SPR declines monotonically as fishing intensity increases (Fig. 1a). Biological reference points derived from YPR analysis ($F_{0.1}$ and F_{max}) have been widely applied, but until recently there were no reference points associated with SPR analysis.

Goodyear (1977, 1980) first developed a "compensation ratio" based on SPR criteria (defined as the ratio of SPR for an unfished population to the SPR for the fished population), and used it as an index of the resilience of a population to fishing and other stresses. However, Shepherd (1982) is generally credited with sparking interest in the development and application of SPR reference points by showing how a standard SPR analysis could be combined with spawning-recruitment (S-R) observations to generate reference fishing mortality rates. The relationship between the two types of information (Fig. 1) is straightforward: for any constant F, there is a corresponding SPR level from SPR analysis (Fig. 1a) that can be inverted and used as the slope of a straight line through the origin of the S-R scatterplot (Fig. 1b). Points along the line represent the average survival ratio (R/S) required to support that particular constant F. Observed survival ratios can therefore be used to define threshold and target levels of F, which can then be translated back to the SPR scale.

One potential candidate for a threshold F is the extinction threshold fishing mortality rate, which will be referred to here by the symbol, F_τ. For most of the commonly-used (non-depensatory) S-R relationships, F_τ is determined by the slope (survival ratio) at the origin of the relationship. Shepherd (1982) proposed that the observed 90th percentile survival ratio could be used as an estimate of the slope at the origin, and suggested that the corresponding F could be treated as a threshold that should not be exceeded. Subsequently, Sissenwine and Shepherd (1987) pointed out that the 90th percentile may overestimate the slope at the origin for two reasons. First, S-R observations with higher survival ratios may just reflect anomalously favourable environmental conditions, not the ability of the population to sustain fishing under average environmental conditions. Second, if the underlying S-R relationship exhibits little compensation (density-dependence) and/or the S-R observations are restricted to low

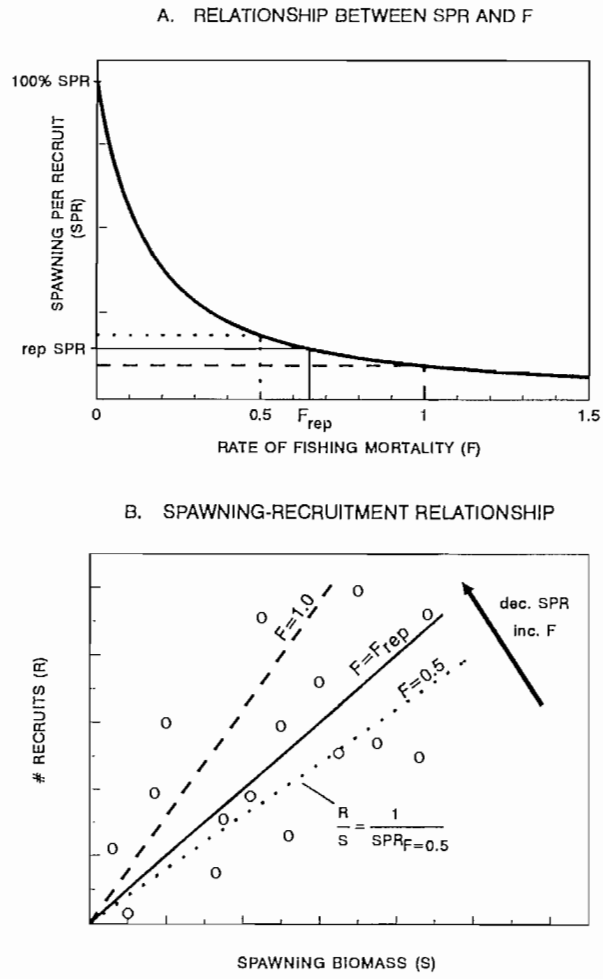


Fig. 1. Relationship between (A) spawning per recruit (SPR) levels derived from SPR analysis and (B) survival ratios (R/S) estimated from spawning-recruitment (S-R) scatterplots. A particular constant rate of fishing mortality (e.g., $F = 0.5$, Fig. 1a) corresponds 1:1 with an associated SPR level ($SPR_{F=0.5}$) which can be inverted and used as the slope of a straight line through the origin of the S-R scatterplot (dotted line, Fig. 1b). This line represents the average survival ratio required to support the corresponding (constant) F. Higher levels of F require higher survival ratios (higher slopes). The fishing mortality rate corresponding to the replacement line is referred to as F_{rep} (solid line, Fig. 1b); the corresponding level of SPR is called the replacement SPR (Fig. 1a). See text for further explanation.

stock sizes where the S-R relationship can be approximated by a straight line through the origin, fishing at a rate corresponding to the 90th percentile survival ratio will generally result in insufficient recruits to replace the spawning stock that produced them, and the stock will decline.

Following on from an earlier discussion in Beverton et al. (1984), Sissenwine and Shepherd (1987) proposed an alternative approach based on the concept of replacement. If fishing occurs at a low rate such that the observed survival ratios tend to be mostly above the line with slope corresponding to the SPR for that F (lower broken line, Fig. 1b), then the average size of the population will increase; if fishing occurs at a high rate such that the observed survival ratios are mostly below

the line (upper broken line, Fig. 1b), the population will tend to decrease. The replacement line is defined as the line with a slope equal to the observed average survival ratio. Sissenwine and Shepherd referred to the fishing mortality rate that corresponds to the average survival ratio as F_{rep} (F-replacement). They proposed estimating F_{rep} from the median survival ratio (solid line, Fig. 1b), in which case it may often be referred to, by the symbol, F_{med} (F-median).

They also defined recruitment overfishing as occurring when fishing mortality exceeds the level associated with the slope at the origin of the S-R relationship, and proposed that F_{rep} could be used as a biological reference point for defining recruitment overfishing (i.e., in the terminology used here, they proposed that F_{rep} could be used as an estimate of F_τ). The validity of this proposal depends on the degree of compensation in the underlying S-R relationship, the range of stock sizes covered by the observations, and environmental conditions and other factors that affect survival rates. It is most applicable in the cases where there is either little compensation over the entire range of the S-R relationship, or the S-R observations are restricted to low stock size where the relationship is approximately linear. This situation is referred to as the null model (no density-dependence over the observed range). If the null model of no density-dependence applies, fishing at F_{rep} should result in S varying without trend about recent levels. If the data exhibit compensation (density-dependence), then fishing at F_{rep} should result in S varying with a tendency toward the mid-range of the observed values of S. In either case, F_{rep} is an estimate of the fishing mortality that, on average, allows for replacement of successive generations over the observed range of S-R data.

It should be noted that use of the null model of no density-dependence does not necessarily mean that compensation does not exist; rather that it may not occur at all levels of S (particularly the levels of S observed in many heavily-exploited fisheries), or it may be obscured by recruitment variability. Rejection of the null model does not affect the validity of the replacement concept; however, for data that exhibit compensation, F_{rep} is likely to be a conservatively-biased estimate of F_τ. There are some circumstances where F_{rep} may even be more appropriate as a fishing target, rather than an overfishing threshold; e.g., stocks with moderate or strong compensation where the S-R observations are restricted to a range of stock sizes that are near or above optimal levels (e.g., near or above B_{msy}). The preceding statements are also contingent on the stability of environmental conditions: F_{rep} may over- or under-estimate F_τ if environmental conditions during the period of observation have produced unusually high or low survival rates, respectively.

Application of SPR Reference Points

Application of these and other SPR reference points has been sporadic. Following Shepherd's (1982) paper, the International Council for the Exploration of the Sea (ICES) Stock Assessment Methods Working Group advocated routine calculation of F_{high}, F_{med} and F_{low}, the reference fishing mortalities corresponding respectively to the 90th, 50th (or median) and 10th percentile survival ratios calculated from S-R data (ICES 1983, 1984). However, even though SPR reference points are now part of the standard output from ICES stock assessments, they are not used explicitly as a basis for management advice. The first fisheries management organization to explicitly use the concept of replacement SPR to define overfishing was the New England Fishery Management Council (NEFMC 1985, Sissenwine and Marchesseault 1985), one of eight regional U.S. Fishery Management Councils charged with assisting the National Marine Fisheries Service to develop Fisheries Management Plans (FMP's) to manage U.S. marine fisheries. It was several more years before the concept was applied elsewhere.

The impetus for wider application of SPR criteria as a basis for fisheries management in the U.S. came with the development of regulations requiring Fishery Management Councils to establish measurable definitions of overfishing for all managed stocks. The new regulations, published in July 1989 and referred to as the "50 CFR Section 602 Guidelines", now require FMP's to

> ...specify, to the maximum extent possible, an objective and measurable definition of overfishing for each stock or stock complex covered by that FMP, and to provide an analysis of how the definition was determined and how it relates to reproductive potential.

The guidelines also state that although some types of overfishing (growth, localised and pulse) may be permissible, the definitions must, at the least, guard against recruitment overfishing.

Here there is an implicit assumption that recruitment overfishing occurs at higher fishing intensity than growth overfishing, which is usually defined as occurring when F exceeds F_{max}, the fishing mortality rate associated with maximum YPR. However, the relationship between F_{max} (or $F_{0.1}$) and reference levels of SPR has not yet been examined. There are currently 33 FMP's in the U.S. covering over 100 stocks and stock-complexes; of these more than 80% use overfishing definitions based on threshold fishing mortality rates. About 20% have used $F_{0.1}$, F_{max} or F_{msy}, while 60% have adopted threshold levels of SPR, which correspond 1:1 with threshold levels of F (but not necessarily S).

In some ways it is surprising that SPR thresholds have been so widely adopted by the U.S. Fishery Management Councils since there are few U.S. fisheries where critical levels of SPR have been calculated explicitly from actual S-R data. Moreover, there are few studies, particularly empirical studies, that can be used to provide guidelines on the values to select. Goodyear (1989) suggested that a critical minimum of at least 20% of the maximum SPR (i.e., 20% $SPR_{F=0}$) should be maintained for stocks where the spawning-recruitment relationship cannot be determined. Clark (1991) suggested that a management target of 35% SPR should be

capable of achieving high yields for a wide range of plausible spawning-recruitment relationships. Goodyear's proposal (for a management threshold) was based on limited empirical results, while Clark's proposal (for a management target) was based on a theoretical analysis. The levels chosen to represent overfishing thresholds by the Fisheries Management Councils have often been somewhat arbitrary, ranging anywhere from 5–70%, although usually falling within the range 20–35%.

Terminology

The amount of SPR that corresponds to F_{rep} is referred to in the present paper as the "replacement SPR". The term F_{rep} is used to refer to the replacement concept, whereas F_{med} is the statistic used to estimate F_{rep}. (More work is needed to determine whether the median is the best estimator of the replacement level; however, it is the estimator that has been the most widely adopted — e.g., ICES 1990a–o). Levels of SPR are often expressed as a percent of the maximum, which occurs at $F = 0$ (Fig. 1a).

It should be noted that the term percent spawning per recruit (%SPR) is analogous to the percent maximum spawning potential (%MSP) first used by NEFMC (1985) and the spawning potential ratio defined by Goodyear (1989) as the inverse of his compensation ratio. Thus, using similar logic to Goodyear (1977, 1980, 1989), the threshold replacement %SPR can be considered a measure of the total "compensatory reserve" of an unfished stock, and therefore as an index of the overall resilience of a stock to fishing pressure (with the degree of resilience being inversely related to the magnitude of the replacement %SPR). For the remainder of this paper, the replacement survival ratio, replacement SPR (i.e., the inverse of the replacement survival ratio), the replacement %SPR, and F_{rep} will be referred to collectively as the replacement reference points. In most contexts, it is preferable to talk about %SPR (or absolute SPR) rather than F_{rep} because the latter is actually a vector of numbers related to partial recruitment (PR), whereas the former is independent of PR.

Purposes of Present Study

The main purposes of the present study were to compile and summarise data on replacement %SPR and related reference points for a variety of well-studied fisheries, and to determine whether there are any consistent effects of taxonomic affiliation or life history parameters on these reference points. One of the primary reasons for investigating such relationships is that if they exist they may be extremely useful for inferring the location of overfishing thresholds for comparatively little-studied fisheries (e.g., most of the U.S. fisheries covered by FMP's), or to define default reference points that can be used until more information becomes available.

Methods

Two basic types of input were required to estimate the replacement SPR and F_{rep} using Sissenwine and Shepherd's (1987) approach: an SPR curve and a spawning-recruitment (S-R) scatterplot (Fig. 1). The replacement SPR was estimated as the inverse of the median survival ratio from the S-R scatterplots or, equivalently, the inverse of the slope of the straight line through the origin that bisects the S-R observations (solid line, Fig. 1b). The corresponding estimate of replacement fishing mortality, F_{rep}, was then obtained directly from the SPR curve (Fig. 1a). The replacement SPR was normalised by dividing by the maximum SPR ($SPR_{F=0}$), the result being referred to here as the replacement %SPR. SPR and %SPR levels corresponding to $F_{0.1}$ and F_{max} were also obtained from the SPR curve.

The main criterion used to select fish stocks for this survey was the availability of adequate stock assessment results. In cases where SPR curves and S-R scatterplots had already been provided in the stock assessment literature or by personal communication, they were used directly to calculate the replacement reference points. Otherwise, it was necessary to assemble the following data: estimates of natural mortality, a maturity ogive, and VPA-type output such as population numbers by age and year, annual mean weights at age, and age- and year-specific fishing mortality rates. The following conventions were adopted to construct S-R scatterplots and YPR and SPR analyses from these data. Spawning was always expressed in units of absolute stock biomass, except in two cases (Atlantic sea scallops) where biomass was only available as a relative index. Spawning stock biomass (SSB) was calculated by projecting forward from the standard January 1 reference time of VPA to the mid-point of the spawning season (provided this was known) both for the S-R plot and the SPR analysis; the last several years of VPA data (usually one more than the age of recruitment to the fishery) were generally omitted from S-R plots; the most recent five years of VPA results were used to calculate average partial recruitments, weights at age and maturity ogives for input to YPR and SPR; if sufficient data were available, the information used to calculate mean weights at age for the stock (e.g., survey data) differed from that used to calculate mean weights at age for the catch (e.g., commercial sampling data); and at least $3/M$ age groups (Anthony 1982) or, more commonly, a plus group (essentially an infinite number of age groups) was used to calculate YPR and SPR. (These conventions were based on the most common approaches adopted by the stock assessment groups that routinely produce S-R scatterplots and SPR analysis.) The sequence of calculations for YPR and SPR was identical to that outlined in Gabriel et al. (1989).

A factor that could have a substantial influence on the validity of the replacement reference points as indices of recruitment overfishing thresholds is the degree of compensation or concavity exhibited by the S-R data. An "index of concavity" was derived to test the null hypothesis that the S-R data could be adequately represented by a straight line through the origin (the median survival ratio) over the observed range of S. Since numeric values were not available for all sets of S-R observations, the index was based on a non-parametric rank sum method. It was calculated by ranking the observations

Table 1. Classification of species included in the survey. Numbers of stocks from each species are given in brackets. Superscripts indicate the number of stocks excluded from analyses involving the replacement reference points.

ORDER CLUPEIFORMES

Family Clupeidae - herrings

1. $(17)^{-1}$ Atlantic herring (*Clupea harengus*)

2. (1) Sardine or pilchard (*Sardina pilchardis*)

3. (2) Sprat (*Sprattus sprattus*)

ORDER GADIFORMES (ANACANTHINI)

Family Gadidae - codfishes

4. $(1)^{-1}$ Pacific cod (*Gadus macrocephalus*)

5. $(15)^{-1}$ Atlantic cod (*Gadus morhua*)

6. $(7)^{-1}$ Haddock (*Melanogrammus aeglefinus*)

7. (5) European whiting (*Merlangius merlangus*)

8. (2) Silver hake (*Merluccius bilinearis*)

9. (2) European hake (*Merluccius merluccius*)

10. $(1)^{-1}$ Pacific whiting or hake (*Merluccius productus*)

11. $(2)^{-1}$ Blue whiting (*Micromesistius poutassou*)

12. (6) Pollock or saithe (*Pollachius virens*)

13. (1) Walleye pollock (*Theragra chalcogramma*)

ORDER PERCIFORMES (PERCOMORPHI; ACANTHOPTERYGII)

Family Carangidae - jacks and pompanos

14. $(2)^{-1}$ Horse mackerel or scad (*Trachurus trachurus*)

Family Scombridae - mackerels and tunas

15. (2) Atlantic mackerel (*Scomber scombrus*)

Family Xiphiidae - swordfishes

16. (2) Swordfish (*Xiphias gladius*)

Family Scorpaenidae - scorpionfishes

17. $(1)^{-1}$ Redfish or ocean perch (*Sebastes marinus*)

18. (1) Deepwater redfish (*Sebastes mentella*)

ORDER PLEURONECTIFORMES (HETEROSOMATA)

Family Bothidae - lefteye flounders

19. (1) Megrim (*Lepidorhombus whiffiagonis*)

20. (1) Summer flounder (*Paralichthys dentatus*)

Family Pleuronectidae - righteye flounders

21. (1) Pacific halibut (*Hippoglossus stenolepis*)

22. (1) Yellowfin sole (*Limanda aspera*)

23. (2) Yellowtail flounder (*Limanda ferruginea*)

24. (5) European plaice (*Pleuronectes platessa*)

25. (2) Greenland halibut (*Reinhardtius hippoglossodes*)

Family Soleidae - soles

26. (6) Dover sole (*Solea vulgaris*)

INVERTEBRATES

27. (2) Atlantic sea scallops (*Placopecten magellanicus*)

based on the magnitude of S, summing the ranks on either side of the median survival ratio (R/S), subtracting the rank sum above the median R/S from the sum below, and dividing by the maximum possible difference ($\frac{1}{4}N^2$ for N even and $\frac{1}{4}(N^2-1)$ for N odd) to normalise it. The smallest rank sum was also used in a Wilcoxon two-sample rank test to test for the significance of deviation from the null hypothesis of no density-dependence (straight line through the origin). Values approaching the extremes of the possible range of the normalised index [-1,1] would occur if the data exhibited pronounced convexity or concavity, respectively. Statistically significant cases where the index is less than zero indicate depensation over the observed range of S and statistically significant cases where the index is greater than zero indicate compensation in the data. In the former case, the replacement survival ratio gives an overestimate of the slope at the origin of the full S-R relationship; in the latter case, it gives a conservative estimate.

Other variables that could potentially influence the replacement reference points include taxonomic affiliation and life history parameters. The life history parameters that were considered in the present analysis include natural mortality (M), the average body weight at 50% maturity, the maximum average body weight, and the maximum observed spawning stock biomass. Multiple linear regression and forward selection stepwise regression techniques were used to investigate the relative importance of these and other variables, including the index of concavity.

Table 2. Estimates of replacement %SPR and sources of information for stocks included in the survey, grouped by geographic location. Species numbers refer to Table 1.

Case No.	Stock	Species No.	Replacement %SPR	Sources of information
A. ICES stocks (Northeast Atlantic and associated areas)				
1	Irish Sea cod	5	3.9	1,15
2	Irish Sea whiting	7	11.4	1,15
3	Irish Sea plaice	24	10.1	1,15
4	Irish Sea sole	26	23.5	1,15
5	Celtic Sea cod	5	6.6	1,15
6	Celtic Sea whiting	7	6.9	1,15
7	Celtic Sea plaice	24	5.0	1,15
8	Celtic Sea sole	26	19.2	1,15
9	Blue whiting, northern stock	11	–	2,15
10	Blue whiting, southern stock	11	7.4	2,15
11	NE Arctic cod	5	5.8	3,15,*
12	NE Arctic haddock	6	24.3	3,15,*
13	NE Arctic saithe	12	9.8	3,15
14	Redfish in areas IIA & B	18	18.2	3,15
15	Greenland halibut in areas I & II	25	21.6	3,15
16	Icelandic summer herring	1	18.6	4,15,*
17	Norwegian spring herring	1	–	4,15,*
18	North Sea sole	26	12.3	5,15
19	North Sea plaice	24	11.2	5,15
20	Div. VIId sole	26	11.5	5,15
21	Div. VIIe sole	26	25.8	5,15
22	Bay of Biscay sole	26	5.6	5,15,*
23	Div. VIIe plaice	24	7.3	5,15
24	North Sea cod	5	3.4	6,15
25	Div. VIa cod	5	11.0	6,15
26	Div. VIId cod	5	5.3	6,15
27	North Sea haddock	6	15.5	6,15
28	Div. VIa haddock	6	18.2	6,15
29	North Sea whiting	7	50.1	6,15
30	Div. VIa whiting	7	37.2	6,15
31	Div. VIId whiting	7	42.7	6,15
32	North Sea saithe	12	16.7	6,15
33	Div. VI saithe	12	24.6	6,15
34	Cod in the Kattegat	5	8.2	7,15,*
35	Cod in the Skagerrak	5	6.1	7,15,*
36	Plaice in the Kattegat	24	8.7	7,15,*
37	North Sea herring	1	10.8	8,15,*
38	Celtic Sea herring	1	27.9	8,15
39	Div. VIa north herring	1	16.8	8,15
40	Clyde herring	1	23.0	8,15,*
41	Div. VIa south & VIIb,c herring	1	23.4	8,15
42	Div. VIIa herring	1	14.6	8,15
43	Baltic cod in area 22	5	2.5	9,15
44	Baltic cod in areas 22& 24	5	2.9	9,15
45	Baltic cod in areas 25–32	5	8.8	9,15
46	Herring in the Western Baltic & Kattegat	1	6.8	10,15,*
47	Herring in areas 25–29 and the Gulf of Riga	1	30.4	10,15,*
48	Herring in coastal areas 25–27	1	39.5	10,15,*
49	Herring in the Gulf of Riga	1	27.1	10,15,*
50	Herring in area 30E	1	63.5	10,15,*
51	Herring in area 31E	1	65.4	10,15,*
52	Herring in the Gulf of Finland	1	17.5	10,15,*
53	Sprat in areas 26& 28	3	45.8	10,15,*
54	Sprat in areas 22–32	3	35.7	10,15,*
55	Mackerel, western stock	15	42.8	11,15
56	Greenland halibut in areas V & XIV	25	8.5	12,15,*
57	Icelandic saithe	12	24.9	12,15
58	Faroe saithe	12	21.4	12,15,*
59	Cod in the Faroe Plateau	5	17.2	12,15,*
60	Faroe haddock	6	31.5	12,15,*
61	Hake, northern stock	9	51.5	13,15,*
62	Hake, southern stock	9	34.1	13,15,*
63	Megrim in areas VII & VIII	19	55.1	13,15,*

Table 2. Cont'd.

Case No.	Stock	Species No.	Replacement %SPR	Sources of information
64	Sardine in areas VIIIc & IXa	2	55.4	14,15
65	Horse mackerel, western stock	14	–	14,15,*
66	Horse mackerel, southern stock	14	22.3	14,15,*
B. Northwest Atlantic stocks (Canada)				
67	Pollock in NAFO areas 4VWX & 5Zc	12	23.7	16,17,18,*
68	Haddock in NAFO area 4X	6	26.0	19,*
69	Haddock in NAFO areas 5Zjm	6	–	18,20,*
70	Cod in NAFO areas 5Zjm	5	–	21,*
71	Herring in NAFO area 4T	1	9.5	22,23,24,25,*
C. Northwest Atlantic stocks (USA)				
72	Georges Bank cod	5	11.9	26,27
73	Gulf of Maine cod	5	8.4	28,29,*
74	Georges Bank haddock	6	20.6	30,31,32,33
75	Silver hake, northern stock	8	30.8	26,34,*
76	Silver hake, southern stock	8	42.4	26,34,*
77	Georges Bank yellowtail flounder	23	14.2	18,28,35,*
78	Southern New England yellowtail flounder	23	10.3	18,28,35,*
79	Summer flounder	20	3.7	26,36,*
80	NW Atlantic Redfish	17	–	30,37
81	Gulf of Maine herring	1	14.9	26,38,39,*
82	NW Atlantic mackerel	15	40.7	18,28,40,*
83	Georges Bank scallops	27	2.0	28,41,42,43,44,*
84	Mid-Atlantic scallops	27	2.9	28,41,42,43,44,*
D. Atlantic stocks (Highly migratory)				
85	North Atlantic swordfish	16	8.6	42,45,*
86	NW Atlantic swordfish	16	10.1	42,45,*
E. Pacific coast stocks (USA)				
87	Pacific whiting	10	–	46,47,48,*
88	East Bering Sea Pacific cod	4	–	49,50,51,*
89	Bering Sea walleye pollock	13	43.8	52,53,54,*
90	Pacific halibut	21	24.6	52,55,56,*
91	Bering Sea yellowfin sole	22	20.4	52,54,57,*

* S-R plots and/or YPR and SPR analyses derived by present author from VPA-type output or reanalysis of published data.

1–15 ICES (1990a)–ICES (1990o)
16 Annand et al. (1990)
17 C. Annand, Bedford Institute of Oceanography, Halifax, NS, pers. comm.
18 O'Brien et al. MS
19 O'Boyle et al. (1989)
20 S. Gavaris, St. Andrews Biological Station, St. Andrews, NB, pers. comm.
21 J.J. Hunt, St. Andrews Biological Station, St. Andrews, NB, pers. comm.
22 M. Chadwick, Department of Fisheries and Oceans, Moncton, NB, pers. comm.
23 Claytor et al. (1991)
24 R. Claytor, Department of Fisheries and Oceans, Moncton, NB, pers. comm.
25 Messieh (1976)
26 NMFS (1990)
27 Serchuk (1990)
28 NMFS (1991)
29 Serchuk et al. (1991)
30 Gabriel (1985)
31 Gabriel et al. (1989)
32 NMFS (1986)
33 Overholtz et al. (1986)
34 NEFMC (1990)
35 Conser et al. (1991)
36 Gabriel (1990)
37 Mayo et al. (1983)
38 NMFS (1989)
39 Stevenson and Lazzari (1990)
40 Overholtz (1991)
41 Conser (1991)
42 R. Conser, Northeast Fisheries Science Center, Woods Hole, MA, pers. comm.
43 Serchuk and Wigley (1991)
44 Wigley (1991)
45 ICCAT (1989)
46 M. Dorn, Alaska Fisheries Science Center, Seattle, WA, pers. comm.
47 Dorn et al. (1990)
48 Dorn and Methot (1991)
49 T. Sample, Alaska Fisheries Science Center, Seattle, WA, pers. comm.
50 Thompson (1990)
51 G.G. Thompson, Alaska Fisheries Science Center, Seattle, WA, pers. comm.
52 Megrey and Wespestad (1990)
53 Wespestad et al. (1990)
54 V.G. Wespestad, Alaska Fisheries Science Center, Seattle, WA, pers. comm.
55 W. Clark, International Pacific Halibut Commission, Seattle, WA, pers. comm.
56 A.M. Parma, International Pacific Halibut Commission, Seattle, WA, pers. comm.
57 Bakkala and Wilderbuer (1990)

Finally, relationships between the reference points of recruitment overfishing and the more widely used reference points from YPR analysis ($F_{0.1}$ and F_{max}) were examined by comparing their relative magnitudes.

Results

To date, 91 stocks distributed amongst 27 species have been included in the survey (see Tables 1 and 2). Of the 91 stocks, there were 66 from the Northeast Atlantic and associated areas (ICES stocks); five from the Canadian coast and 13 from the U.S. coast of the Northwest Atlantic; two highly-migratory swordfish stocks from the North Atlantic; and 5 stocks from the U.S. coast of the Northeast Pacific (Table 2). It was difficult to develop objective criteria, *a priori*, to determine whether particular data sets were likely to produce estimates of F_{rep} that would be "unacceptably-biased" estimates of recruitment overfishing thresholds (the latter being equated with F_τ, the threshold fishing mortality rate associated with extinction of the stock). Cases were excluded only if it seemed likely that F_{rep} would be an extremely conservative estimate of F_τ. In particular, F_{rep} values less than zero imply that the population is unable to replace itself even in the complete absence of fishing (suggesting that it is either near or beyond "virgin" levels, or has otherwise attained a size greater than that which can be

Table 3. Estimates of replacement %SPR by (A) all cases, (B) whether or not the index of concavity was statistically significant, and (C) taxonomic group. Calculated statistics were derived from arcsine-transformed percentages and then back transformed.

Cases considered	*N*	Mean	Median	Mean ± 2(s.e.)	Range
A.					
All cases	83	18.7	17.2	15.5–22.0	2.0–65.4
B.					
concavity sign.	36	19.3	18.9	15.4–23.5	2.9–51.5
concavity non-sign.	46	17.5	13.2	13.1–22.4	2.0–65.4
C.					
Atlantic cod	14	6.8	6.3	4.9–9.0	2.5–17.2
other gadoids	23	25.7	24.6	20.2–31.5	6.9–51.5
Atlantic herring+	9	17.4	16.8	13.5–21.6	9.5–27.9
other clupeids	10	37.5	37.6	25.0–51.0	6.8–65.4
flatfish	19	14.5	11.5	10.0–19.6	3.7–55.1
Perciformes	6	22.4	20.2	11.6–35.5	8.6–42.8

+ excludes Baltic stocks of Atlantic herring, which are grouped in the other clupeids category.

Table 4. Bivariate correlations between variables considered in regressions. rep%SPR is the replacement %SPR; WT_{max} is the average body weight of the oldest age group considered in YPR and SPR calculations; $WT_{50\%mat}$ is the average body weight at 50% maturity; SSB_{max} is the maximum observed SSB over the range of observations used to construct the *S-R* plot; *M* is natural mortality; **concav** is the index of concavity in the *S-R* data. Probability levels: ns, not significant; *, $p < 0.05$; **, $p < 0.01$; ***, $p < 0.001$.

	log rep %SPR	log WT_{max}	log $WT_{50\%mat}$	log SSB_{max}	*M*	log (**concav**+1)
log (rep %SPR)	1					
log (WT_{max})	-0.47***	1				
log ($WT_{50\%mat}$)	-0.41***	0.96***	1			
log (SSB_{max})	0.46***	-0.19ns	-0.21ns	1		
M	0.27*	-0.03ns	-0.09ns	0.22ns	1	
log (**concav**+1)	0.25*	-0.02ns	0.04ns	0.09ns	0.07ns	1

supported by current levels of recruitment), and are implausible as estimates of F_τ under "average" conditions. F_{rep} will also be an overly-conservative fishing threshold if most of the observations are made near virgin stock size for a strongly-compensatory *S-R* relationship, such that the slope of the line bisecting the observations considerably underestimates the slope at the origin; however such situations are difficult to identify since both virgin stock size and the underlying *S-R* relationship are generally unknown.

Eight of the 91 cases were excluded from analyses of recruitment overfishing thresholds because either: (i) $F_{rep} < 0$ (replacement %SPR > 100%) for either the first half, the last half, or all of the time series of *S-R* observations, (cases 9, 17, 65, 69, 80, 87 and 88 in Table 2), or (ii) the data were dominated by occasional years of exceptionally high recruitment that resulted in stock size moving systematically away from the origin for several successive years (cases 65, 70 and 87). The latter category, although somewhat more arbitrary, excluded only one additional stock (case 70).

For the remaining 83 cases, the replacement %SPR ranged from 2.0 to 65.4%, with a mean of 18.7% (Table 3). The untransformed distribution was highly skewed (Fig. 2). As expected, the mean replacement %SPR was higher for cases with a significant index of concavity (36 cases) compared to those with a non-significant index (46 cases, for a total of 82 cases where the index could be computed); but the differences were not significant (*t*-test, arcsine-transformed data, $p > 0.05$). Differences between the medians were more pronounced, but also not significant (median test and Kolmogorov-Smirnov two-sample test, $p > 0.05$).

In contrast, there were clear and statistically significant differences among species groups (Table 3). In earlier analysis species were grouped solely on the basis of the taxonomic category of Order, but preliminary cluster analysis suggested pronounced differences between Atlantic cod and the rest of the gadoids, and Atlantic herring outside of the Baltic compared to other clupeids. Atlantic cod had the smallest mean replacement %SPR, suggesting that it has relatively the greatest resilience to fishing. Flatfish and Atlantic herring outside of the Baltic appear to have higher than average resilience, and many of the other gadoids and small pelagics apparently have relatively low resilience to fishing. The following paired comparisons between means were significantly different (*t*-tests, arcsine-transformed data, $p < 0.05$): Atlantic cod and each of the other species-groups, flatfish with other gadoids and other clupeids, and Atlantic herring and other clupeids.

Stepwise multiple linear regression (MLR) was conducted using both untransformed and log-transformed variables. Log transformations generally seemed to improve the correlations

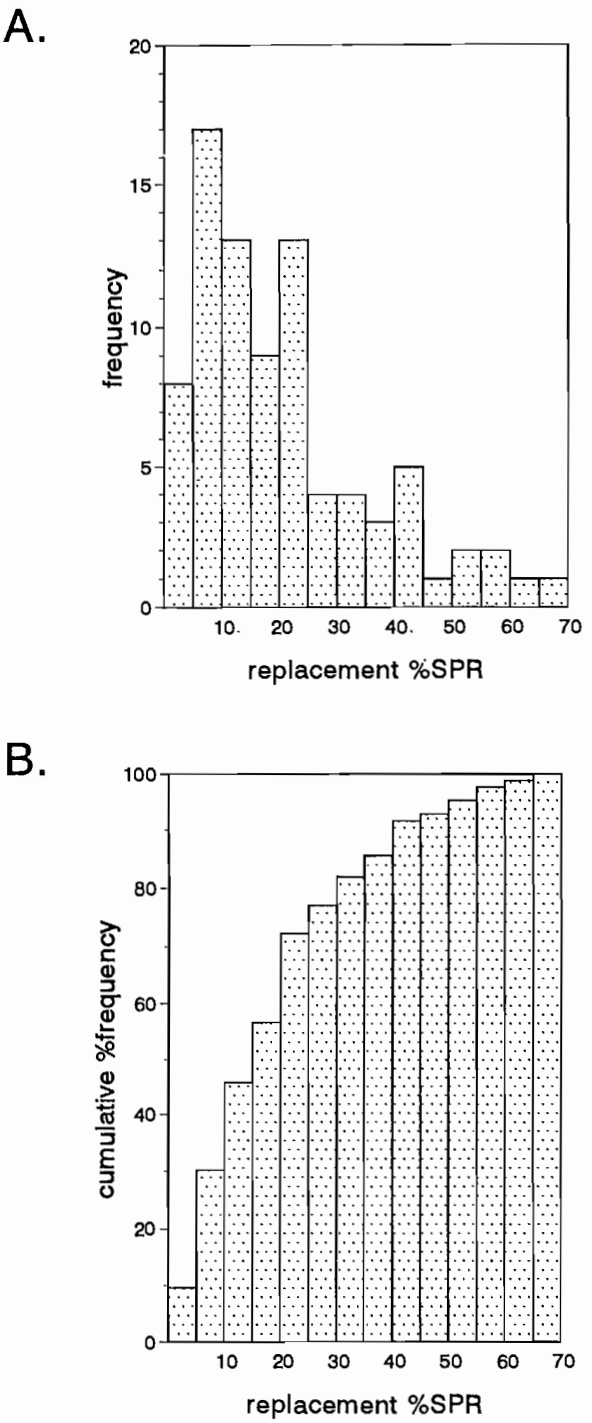


Fig. 2. Frequency histogram (A) and cumulative frequency histogram (B) of replacement %SPR ($N = 83$).

between variables. Bivariate correlations between the logarithm of replacement %SPR and life history parameters exhibiting significant correlations ranged from −0.47 to 0.46 (Table 4). Replacement %SPR was also significantly correlated with the index of concavity. The only significant correlation among the independent variables themselves was the two body weight variables. Three cases had missing values for either maximum SSB or the index of concavity.

The first regression considered all of the variables in Table 4, along with the categorical variable, species name. The second omitted maximum SSB and the index of concavity, since these are the two variables that are the least likely to be known for most stocks. Results were verified by considering all possible subsets in standard MLR analysis. Both regressions were highly significant ($p < 0.001$) and in each case the proportion of variation explained exceeded 60% (Table 5). In both examples, maximum average body weight was the most important covariate (first to be entered by the forward selection process). Maximum SSB, species name and the index of concavity also contributed significantly to the first regression. The second regression included only species and maximum body weight. Omission of maximum SSB and the index of concavity from the list of variables considered did not substantially alter the significance of the regression or the proportion of variation explained, primarily because the increased sample size happened to result in the addition of two species. One of these (NW Atlantic sea scallops, the only invertebrate species in the survey, see Table 1) was such a clear outlier in bivariate regressions of replacement %SPR on body weight (Fig. 3) that it was probably largely responsible for species name becoming the first variable to be entered by the forward selection procedure. Subsequent regressions only considered the 80 cases with complete information for all life history parameters. (Note that this reduction in sample size, together with the cases excluded due to a probable bias in the estimate of replacement %SPR (Table 1), reduces the number of species included in the regression analyses from 27 to 22.)

The problem with regressions that include species as a categorical variable is that they cannot be used for predictive purposes unless the species of interest is one of those already incorporated in the regression. However, omission of species from the list of potential independent variables resulted in a substantial reduction in the proportion of variation explained, particularly when maximum SSB was also excluded (Table 6). An alternative was to conduct analyses separately by taxonomic group. Four Orders were considered: Gadiformes (gadoids only), Clupeiformes (clupeids only), Pleuronectiformes (flatfish) and Perciformes. For each group, maximum body weight was the first variable entered into the regression and, for all groups excepting the gadoids, it was the only variable (Table 6). Relationships between replacement %SPR and maximum body weight are shown separately by taxonomic group in Fig. 4. For flatfish, the regression was non-significant when the three stocks of halibut were included. The regression for Perciformes should also be treated with caution since it was based on only six observations.

Values of %SPR corresponding to the $F_{0.1}$ and F_{max} reference points from YPR analysis were remarkably consistent (Fig. 5 and Table 7). Interquartile ranges (25th to 75th percentiles) were respectively 35.0–40.2% and 17.3–24.5% (compared to 8.6–27.1% for the replacement %SPR); ranges encompassing 90% of the estimates were respectively 31–46% and 12–29%. Some of the differences between species and species-groups (Table 7) were statistically significant, but the magnitude of the differences was never large. The mean

Table 5. Stepwise regression results for log replacement %SPR. (A) Independent variables considered for selection were species, $\log(WT_{max})$, $\log(WT_{50\%mat})$, $\log(SSB_{max})$, M and log(**concav**+1). (B) As for A, but excluding $\log(SSB_{max})$ and log(**concav**+1). See Table 4 for descriptions of the variables.

		A	B
N		80	83
Variables selected	1.	$\log(WT_{max})$	species
	2.	$\log(SSB_{max})$	$\log(WT_{max})$
	3.	species	M^{+}
	4.	log(**concav**+1)	$\log(WT_{50\%mat})^{+}$
	5.	M^{+}	
	6.	$\log(WT_{50\%mat})^{+}$	
Adjusted R^2		61.4%	64.3%
probability level		< 0.001	< 0.001

$^{+}$ subsequently removed (i.e., excluded from final regression results).

replacement %SPR of 18.7% is only slightly lower than the mean %SPR at F_{max} but considerably lower than the mean %SPR at $F_{0.1}$ (Table 7). On average, the corresponding values of F_{rep} exceeded F_{max} by a factor of 1.4, and $F_{0.1}$ by a factor of 2.4 (Table 8). Overall, F_{rep} was more than four times $F_{0.1}$ 25.0% of the time; and more than four times F_{max} 7.1% of the time (Fig. 6). Atlantic cod stood out from the other species groups in terms of the relative magnitude of F_{rep} in relation to both $F_{0.1}$ and F_{max} (Table 8).

However, there are also numerous instances where F_{rep} fell below one or both of the reference points from YPR analysis (Fig. 6 and Table 8). F_{rep} was less than $F_{0.1}$ in 12.5% of the cases where they were both known; and less than F_{max} in 37.1% of the cases where both were defined. Cases where $F_{0.1}$ exceeded F_{rep} included zero Atlantic cod stocks, 3 of a total of 22 other gadoid stocks (13.6%), 4 of 17 clupeid stocks (23.5%), 1 of 19 flatfish stocks (5.3%), and 2 of 6 Perciformes stocks (33.3%). Cases where F_{max} exceeded F_{rep} included zero Atlantic cod stocks, 13 of 20 other gadoid stocks (65%), 5 of 5 clupeid stocks (100%), 5 of 18 flatfish stocks (27.8%), and 3 of 5 Perciformes stocks (60%). Thus, if F_{rep} is an appropriate measure of the recruitment overfishing threshold, adoption of either $F_{0.1}$ or F_{max} as management targets will not necessarily guard against recruitment overfishing.

The stocks considered in this study have been fished at a wide range of fishing mortality rates relative to F_{rep}. Fishing mortality rates averaged over the most recent 10 years of observations ranged from 0.19 to 2.50 F_{rep}, with 77.1% of the stocks falling between 0.5 and 1.5 F_{rep}, and 38.6% in the range 0.8–1.2 F_{rep}. The median result was 1.17 F_{rep}, which should have resulted in an overall trend of declining spawning stock biomass.

Discussion and Conclusions

Replacement biological reference points show great promise as a basis for overfishing definitions for fishery management. They are already widely used in the U.S. They can be easily and objectively calculated from common stock assessment information, such as VPA-type outputs and results from dynamic pool models. Four additional attributes, that are new results from this study, are:

1. Replacement %SPR levels are strongly influenced by taxonomic affiliation and life history parameters.

2. While the replacement biological reference points may sometimes be conservative estimates of recruitment overfishing thresholds (because they do not take account of compensation), for the cases analysed here the degree of apparent compensation in the data had relatively little effect of the magnitude of estimates of replacement %SPR.

3. The combined effect of (1) and (2) is that the replacement %SPR relationships derived in this paper can be used to provide guidance for selecting preliminary or default estimates for recruitment overfishing thresholds when the required stock assessment results are not available.

4. For the cases considered here, F_{rep} was usually higher than the common management targets of $F_{0.1}$ and F_{max}; but this was not always so. Thus, if F_{rep} is a valid recruitment overfishing threshold, $F_{0.1}$ and F_{max} management strategies may not always guard against recruitment overfishing.

Influence of Taxonomic Affiliation and Life History Parameters on Replacement %SPR

Although this study represents only a preliminary analysis of replacement reference points (primarily replacement %SPR and F_{rep}) and associated data, the results are promising and some general conclusions have emerged. First, there appear to be consistent differences in levels of replacement %SPR between some taxonomic groups (Table 3). Flatfish and large gadoids such as Atlantic cod have relatively low values of replacement %SPR (suggesting relatively high resilience to fishing), while the smaller gadoids and many of the small pelagics have relatively high values of replacement %SPR (low resilience). These between-species and species

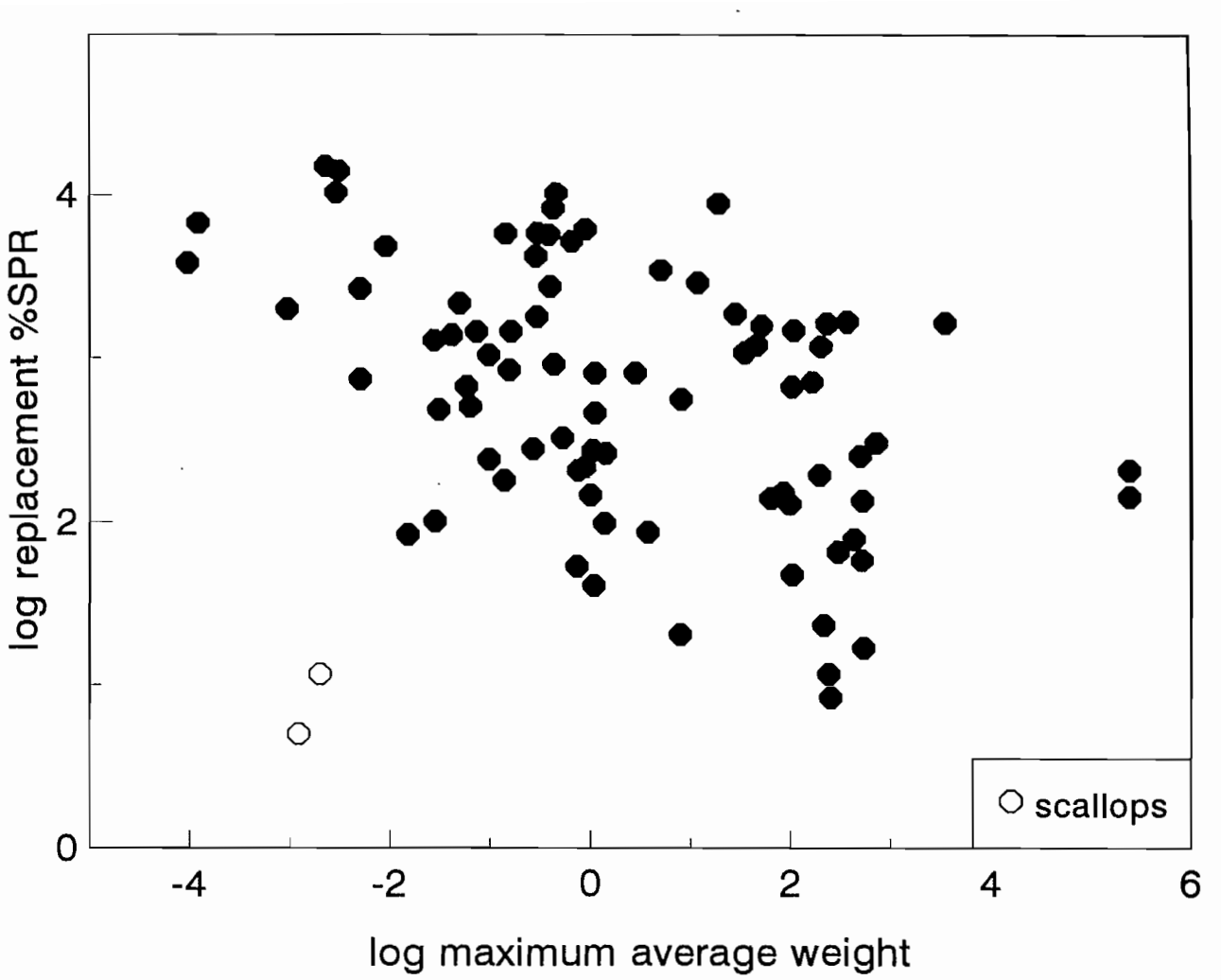


Fig. 3. Relationship between replacement %SPR and observed maximum average body weight ($N = 83$).

group differences appear to be more important than any potential biasing effect of concavity in the S-R observations (Table 5, case A).

Second, a substantial proportion of the variation in replacement %SPR can be explained by certain life history parameters. The most important parameters identified to date are the maximum body weight or the body weight at 50% maturity (Figs. 2 and 3 and Tables 4–6). In particular, gadoids and pelagics that reach relatively small maximum size and/or mature at a small size seem to have high replacement %SPR. Pope (1990; ICES 1990p) has also found a relationship between %SPR and average body weight at 50% maturity. It is worth examining the possible causes of these and other correlations in Table 4.

First, it is unlikely that the correlation with body weight can be extended across all taxa involved in commercial fisheries. For example, elasmobranchs have large body size but generally do not exhibit high resilience to fishing and would probably have moderate or high threshold values of replacement %SPR. Body size is probably a proxy for fecundity, which may be more strongly tied to resilience and therefore to replacement %SPR. Other authors have already suggested that high fecundity may be associated with high resilience (e.g., Cushing and Harris 1973). Certainly the life history parameters considered here do not capture all between-species differences. Omission of species from the overall regression resulted in a decrease in the proportion of variation explained from 61.4% to 42.6% (Tables 5 and 6).

There is also an indication that stocks with high biomass may have relatively low resilience to fishing, particularly for the gadoids (Tables 4–6). However, this result seems somewhat counter-intuitive and may be spurious. The maximum observed SSB depends both on the maximum possible stock size (virgin biomass) and the level of exploitation applied during the period of the observations. For a given stock under stable conditions, replacement %SPR will be positively correlated with stock size. Stocks that are closer to virgin levels might also be expected to exhibit greater concavity in the S-R observations. However, there was no significant correlation between the maximum observed SSB and the index of concavity (Table 4), perhaps partially due to the fact that SSB was expressed in absolute terms and here are probably wide disparities in the maximum observed levels relative to virgin levels.

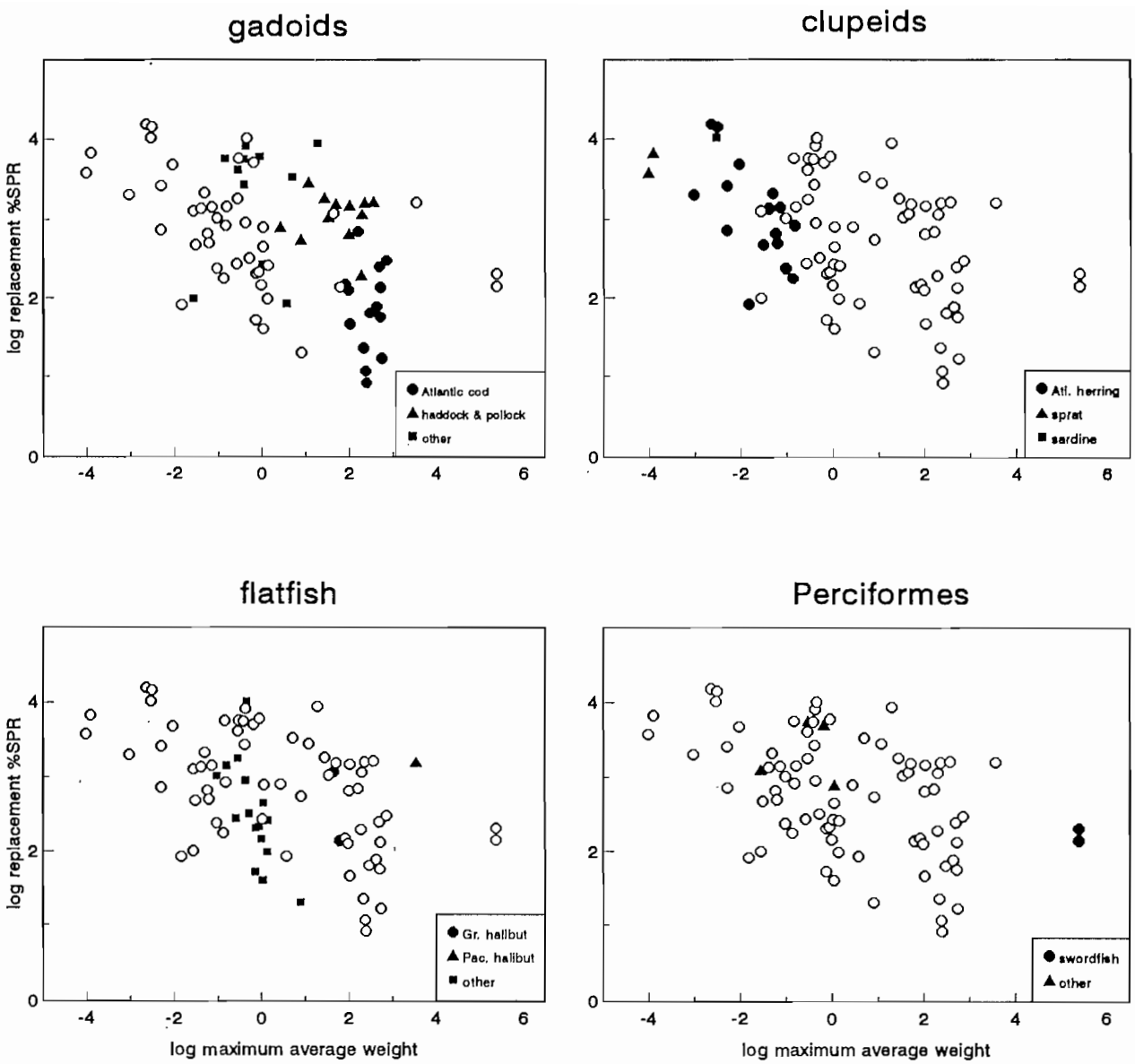


Fig. 4. Relationship between replacement %SPR and observed maximum average body weight by taxonomic group: gadoids ($N = 37$), clupeids ($N = 19$), flatfish ($N = 19$), and Perciformes ($N = 6$).

Replacement Biological Reference Points as Estimates of Recruitment Overfishing Thresholds

The validity of using the replacement reference points as reference points for recruitment overfishing thresholds depends on the validity of the assumption that F_{rep} is an estimate of F_τ, the threshold fishing mortality rate associated with extinction (or, equivalently, that the replacement survival ratio is an estimate of the slope at the origin of the full S-R relationship). There are two major types of situations where F_{rep} might overestimate F_τ: the case of a depensatory S-R relationship, and the situation where environmental conditions or other factors have led to unusually high survival rates through much of the period used to estimate the replacement reference points. Similarly, the main factors causing F_{rep} to underestimate F_τ are a compensatory S-R relationship where most of the S-R observations come from stock sizes above the point where the S-R relationship can be approximated by a straight line through the origin, and an unusually high proportion of years with poor survival rates. Biases may also result from estimation errors and the use of life history information from a fished-down stock to calculate the maximum (unfished) SPR ($SPR_{F=0}$). For example, if the SPR analysis is based on a plus group weight at age derived from a fished-down stock, $SPR_{F=0}$ will be underestimated and replacement %SPR will therefore be overestimated. Similarly, density-dependent effects that result in earlier maturity and faster growth at lower density will lead to overestimates of $SPR_{F=0}$ and underestimates of replacement %SPR. However, F_{rep} (as estimated by F_{med}) may be relatively robust to a number of other sources of error in inputs and assumptions (e.g. M) because their effects on S-R data and SPR analysis tend to cancel (Jakobsen 1993).

Table 6. Stepwise regression results for log replacement %SPR when the categorical variable, species, is omitted. All cases (2) is identical to all cases (1) except that SSB_{max} was removed from the model. Other numbers in brackets refer to the order of selection of the regression variables. Probability levels: ns, not significant; $*$, $p < 0.05$; $**$, $p < 0.01$; $***$, $p < 0.001$. See Table 4 for descriptions of the variables.

	N	intercept	log WT^1_{max}	log SSB^2_{max}	log $WT^1_{50\%mat}$	log **concav+1**	M	adj. R^2	p
All cases (1)	80	2.18	-0.52(1)	0.15(2)	0.41(3)		2.71(4)	42.6%	***
All cases (2)	80	2.69	-0.51(1)		0.38(2)		3.52(3)	31.5%	***
gadoids	37	2.42	-0.63(1)	0.21(2)	0.38(4)	0.87(3)		55.5%	***
clupeids	19	2.35	-0.42(1)					35.3%	**
flatfish[3]	15	2.21	-1.00(1)					60.7%	***
Perciformes	6	3.25	-0.18(1)					63.7%	*

[1] units=kg
[2] units=thousands of metric tons
[3] excludes halibut

Table 7. Estimates of %SPR at (A) $F_{0.1}$ and (B) F_{max}, by taxonomic group. Calculated statistics were derived from arcsine-transformed percentages and then back transformed.

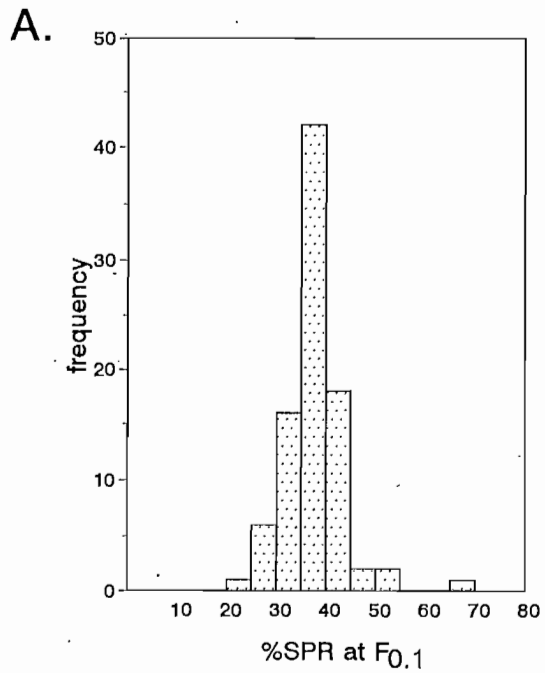
Taxonomic group	N	Mean	Median	Mean ± 2(s.e.)	Range
A. %SPR at $F_{0.1}$					
All cases	88	37.8	37.8	36.6–39.0	23.3–65.1
Atlantic cod	15	39.4	40.0	37.7–41.0	31.7–43.8
other gadoids	26	40.1	38.5	37.3–42.9	29.4–65.1
Atlantic herring[+]	10	34.6	34.4	30.8–38.5	23.3–45.9
other clupeids	8	34.1	35.8	30.9–37.4	26.5–40.0
flatfish	19	37.1	36.9	35.5–38.7	29.8–44.9
Perciformes	8	36.0	37.2	32.3–39.7	28.6–44.2
B. %SPR at F_{max}					
All cases	68	21.1	20.8	19.6–22.8	10.6–55.3
Atlantic cod	15	24.1	23.7	22.1–26.1	17.8–31.9
other gadoids	22	23.1	21.9	19.5–26.9	10.6–55.3
Atlantic herring[+]	3	18.5	20.5	11.2–27.2	11.1–24.0
other clupeids	2	17.0	17.0	–	13.5–20.8
flatfish	18	18.7	18.2	16.8–20.7	12.5–29.5
Perciformes	6	16.6	15.8	11.8–22.1	10.8–27.9

[+] excludes Baltic stocks of Atlantic herring, which are grouped in the other clupeids category.

In general, one would expect there to be more cases where F_{rep} is a conservatively-biased estimate of F_τ (i.e., it underestimates F_τ), due primarily to the fact that there is often evidence of concavity in the S-R observations. In fact, although the null hypothesis of linearity through the origin could not be rejected in most cases (56%), there was an even greater number of cases (82%) where the alternative null model of a (median) straight line with zero slope could not be rejected using a similar non-parametric test. This alternative null model has not been given serious consideration here because it is an inappropriate (risk-prone) null hypothesis for fisheries management. Rather than assuming that the maximum survival ratio is infinite, we have assumed that the replacement (median) survival ratio is a valid estimate of the maximum survival ratio under average (environmental) conditions. Since evidence of apparent compensation in the data did not lead to significantly higher estimates of replacement %SPR on average (Table 3), both sets of data can be considered to be more or less equally valid estimates of recruitment overfishing thresholds. It should be noted, however, that this conclusion cannot be generalised beyond the present analysis (and obviously does not hold for the data sets explicitly excluded from the analysis). Most of the fisheries considered here have had a long history of exploitation and are probably well beyond the fishing-down phase. But the null hypothesis of a straight line through the origin would generally be expected to be overly-conservative for developing fisheries where accumulated biomass is still being fished down.

Guidance for Selecting Estimates of Recruitment Overfishing Thresholds

The regressions derived in this paper (Tables 5 and 6) are all highly significant and generally explain a moderate to high proportion of the variation in the dependent variable (replacement %SPR). These results, together with the basic statistics for the species subgroups (Table 3) can be used to set preliminary recruitment overfishing thresholds, particu-



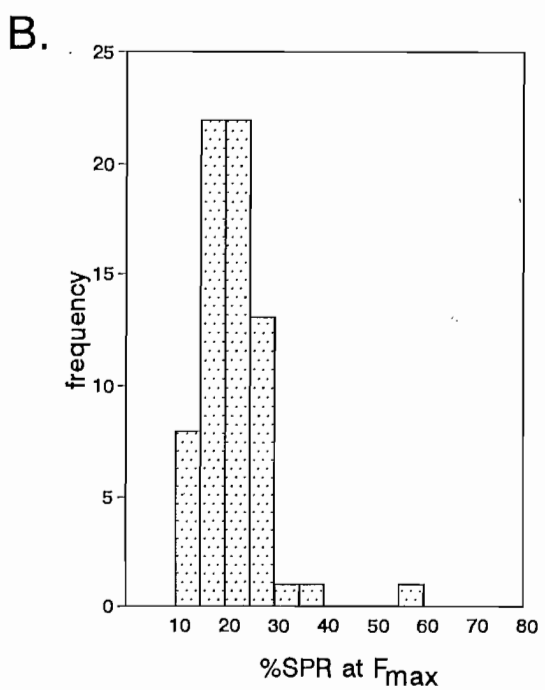
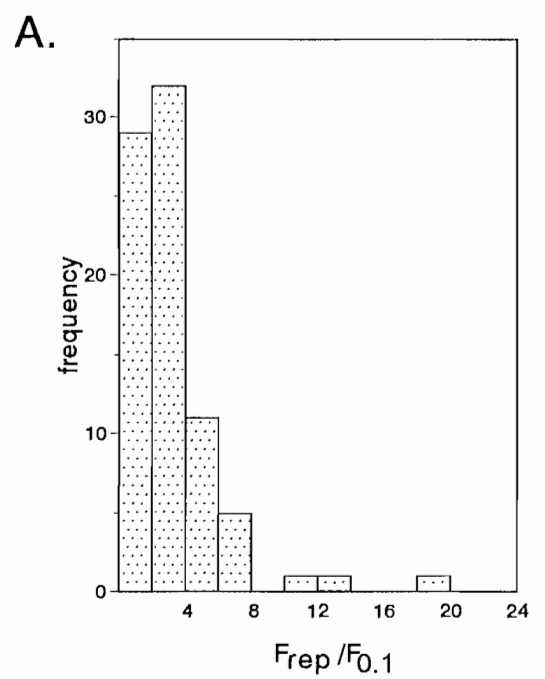


Fig. 5. Frequency histograms of %SPR at (A) $F_{0.1}$ ($N = 88$) and (B) F_{max} ($N = 68$).



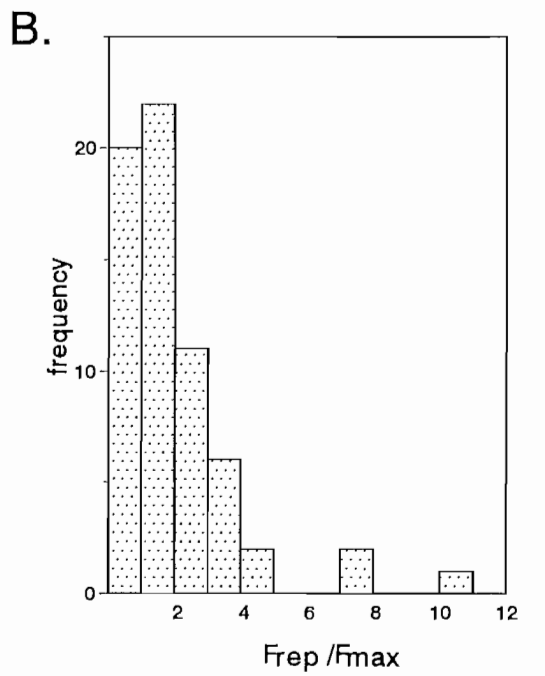


Fig. 6. Frequency histograms of the ratio of F_{rep} to (A) $F_{0.1}$ ($N = 80$) and (B) F_{max} ($N = 64$).

larly in situations where there are insufficient data to calculate the reference points explicitly. Due to the preliminary nature of the analysis, and other uncertainties, estimates based on this study need to be treated with caution.

If there is little or no basic life history information, then it will obviously be difficult to provide definitive guidelines for management. At best, it may be possible to use the results of this study to make some tentative, qualitative statements about whether the stock is likely to have low or high resilience to fishing depending on, for example, the maximum body size of individuals in the population and the taxonomic relationship, if any, to species included in this survey. Stocks of fish that do not attain a very large body size are apparently likely to have relatively low resilience. In particular, a cautious approach to management may be required for some of the smaller gadoids and pelagics. Also, stocks with large virgin biomass or high

Table 8. Estimates of the ratio of F_{rep} to (A) $F_{0.1}$ and (B) F_{max}, by taxonomic group. Calculated statistics were derived from log-transformed ratios and then back transformed. The reported mean is the geometric mean (i.e., uncorrected for bias).

Taxonomic group	N	Mean	Median	Mean ± 2s.e.	Range
A. $F_{rep}/F_{0.1}$					
All cases	80	2.4	2.3	2.0–2.8	0.3–19.1
Atlantic cod	14	4.6	4.9	3.7–5.8	2.4–11.4
other gadoids	22	1.8	1.9	1.4–2.3	0.7–7.3
Atlantic herring+	9	2.3	2.2	2.0–2.7	1.5–3.0
other clupeids	8	1.0	1.1	0.5–2.0	0.3–4.6
flatfish	19	2.9	2.9	2.2–3.8	0.7–7.6
Perciformes	6	1.6	2.0	1.1–2.3	0.8–2.4
B. F_{rep}/F_{max}					
All cases	64	1.4	1.3	1.2–1.7	0.2–10.5
cod	14	2.7	2.8	2.1–3.5	1.0–7.3
other gadoids	20	0.9	0.8	0.7–1.2	0.2–4.0
Atlantic herring+	3	1.2	1.3	1.0–1.4	1.0–1.3
other clupeids	2	1.4	1.4	–	0.8–2.5
flatfish	18	1.4	1.4	1.0–1.9	0.4–4.6
Perciformes	5	0.8	1.1	0.5–1.4	0.3–1.1

+ excludes Baltic stocks of Atlantic herring, which are grouped in the other clupeids category.

natural mortality may be relatively less resilient to fishing (may not be able to withstand large reductions in biomass).

In situations where there is sufficient information to construct YPR and SPR curves, but fewer than 10 or so S-R observations, the results of this study can be used more directly. For species that are dissimilar to any of those well-represented in this survey, it might be appropriate to use, say, the 80th percentile of the observations as a preliminary estimate of the threshold replacement %SPR, and to then find the corresponding estimate for F_{rep} on the SPR curve. Use of the 80th percentile of the cases in this study results in a default estimate for the threshold replacement %SPR of 30.8% (Fig. 2), which we will round down to 30%. It is interesting to note that Clark (1991) suggested a %SPR value of 35% to achieve at least 75% of the MSY yield when the S-R relationship is unknown; i.e., this study suggests 30% as a default 'threshold', whereas Clark recommended 35% as a management 'target'. The two levels are too similar to be used concurrently as indicators of danger zones and safe zones, respectively, within the same management plan. It is likely that Clark's suggested target will be too low (optimistic) for some stocks because it was based on theoretical models that did not take account of recruitment variability and other sources of uncertainty, and may not have covered the full range of possible degrees of compensation. On the other hand, the 80th percentile of the observations included in this survey will be an overly-conservative threshold for most stocks.

An alternative to the 80th percentile result is to use the overall average replacement %SPR of about 20% (Table 4) as the default (which is, coincidentally, the default that has been used by the New England Fishery Management Council to define overfishing thresholds for groundfish stocks, and the default suggested by Goodyear 1989, 1993 based on limited empirical information). However, we suggest that 20% should generally be considered too low for use as a default threshold since it is risky to assume that a stock is "average" when nothing is known about the spawning-recruitment relationship. Another alternative is to obtain a preliminary estimate of the threshold replacement %SPR from one of the overall regressions on life history parameters (Table 6).

Obviously, the results from this survey are most applicable to the situation where the species is closely related to one or more of those well-represented in the survey. In this case, the mean and confidence interval for that species or species-group (Table 3) or one of the regressions based on the taxonomic group of Order (Table 6) could be used directly. Alternatively, if there is sufficient information (SPR analysis and preferably 10 or more S-R observations) to calculate the replacement biological reference points explicitly, the results from this study can be used as a check on the empirical estimates.

Relative magnitudes of F_{rep}, $F_{0.1}$ and F_{max}

The U.S. 50 CFR Section 602 Guidelines (see Background) imply that growth overfishing occurs at a lower fishing mortality rate than recruitment overfishing. If F_{max} and F_{rep} are used as the biological reference points of growth and recruitment overfishing, respectively, then the present study indicates that this generalisation is true only 63% of the time. On average, F_{rep} was 1.4 times F_{max} (Table 8); however, the average %SPR associated with F_{max} was only slightly higher than the average %SPR associated with F_{rep} (Table 7).

Another widely-held belief is that $F_{0.1}$ is a conservative reference point for defining recruitment overfishing. Assuming that F_{rep} is the true recruitment overfishing threshold, the present study indicates that while this belief is generally valid ($F_{0.1}$ was only about 40% F_{rep} on average, Table 8), there was an appreciable number of cases included in the survey (12.5%) where F_{rep} was less than $F_{0.1}$. This suggests that a management target of $F_{0.1}$ may not always prevent recruitment overfishing. The average %SPR associated with $F_{0.1}$

was 37.8% (Table 7), about 8% higher than the default threshold (30%) suggested above.

Concluding Remarks

There is no single, simple answer to the question of how much spawning per recruit is enough. The results of this study do however provide some guidance, particularly for species similar to those included in the survey and when certain life history parameters are known. Based on the overall cumulative distribution of replacement %SPR (Fig. 2), a conservative strategy would be to maintain at least 30% SPR as a default value when there is no other basis for estimating the replacement level. A 30% level of SPR was enough for 80% of the fish stocks considered; however, it may be overly-conservative for an "average" stock.

It is important to remember that the replacement biological reference points calculated in this study are intended to serve as management thresholds, not targets. In general, it should be expected that threshold fishing mortality rates such as F_{rep} will be higher than targets such as $F_{0.1}$ and F_{max}. The fact that this was not always so in the present study indicates that it is essential to consider recruitment overfishing reference points explicitly. In cases where F_{rep} exceeds F_{max}, it does not make sense to apply a fishing mortality rate of F_{rep} since it will produce less yield. On the other hand, if F_{rep} is near or below F_{max} (or $F_{0.1}$), it may be prudent to sacrifice some yield and fish conservatively. For populations that are depleted, the fishing mortality rate should be less than F_{rep} to promote rebuilding.

Acknowledgements

Numerous groups and individuals provided the stock assessment information and related analyses required to estimate the replacement reference points. In particular, we wish to mention the ICES Stock Assessment Working Groups (ICES 1990a–o) and to give special thanks to the scientists acknowledged as personal communications in Table 2. Mike Fogarty and Jake Rice reviewed an earlier draft and provided helpful suggestions for revision. The first author is also grateful to the New Zealand Ministry of Agriculture and Fisheries for providing funding for the initial phases of the study, and to the National Research Council-NMFS Research Associateship program which enabled the work to be completed.

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The Behaviour of F_{low}, F_{med} and F_{high} in Response to Variation in Parameters Used for Their Estimation

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Jakobsen, T. 1993. The behaviour of F_{low}, F_{med} and F_{high} in response to variation in parameters used for their estimation. p. 119–125. *In* S. J. Smith, J. J. Hunt and D. Rivard [ed.] Risk evaluation and biological reference points for fisheries management. Can. Spec. Publ. Fish. Aquat. Sci. 120.

Part of the estimation of F_{low}, F_{med} and F_{high} is based on the SSB/R curve. The behaviour of these biological reference points in response to variation in the parameters used to calculate the SSB/R curve is investigated. Although the year-to-year variation in the estimates usually is small, it may be confusing in management considerations and it is suggested that it might be preferable to stabilize the estimates by using long-term values for the estimation parameters. F_{med} is more robust than F_{max} in response to changes in the assumed value of natural mortality rate (M) in an assessment, especially when the ratio to current F is considered. The reason for this robustness is investigated and the usefulness of F_{low}, F_{med} and F_{high} as biological reference points is discussed.

L'estimation de F_{inf}, de F_{moy} et de F_{sup} se fonde en partie sur la courbe de la biomasse du stock reproducteur par recrue (SSB/R). On étudie le comportement de ces points de référence biologiques par suite d'une variation des paramètres utilisés pour calculer cette courbe. Même si la variation d'une année à l'autre des estimations est en général faible, elle peut être déroutante au niveau des considérations en matière de gestion et on semble indiquer qu'il serait préférable de stabiliser les estimations en ayant recours à des valeurs à long terme pour l'estimation des paramètres. La valeur F_{moy} est plus robuste que la valeur F_{max}. en réponse aux modifications de la valeur supposée du taux de mortalité naturelle (M) dans une évaluation, en particulier lorsque l'on considère le rapport à la valeur F actuelle. On étudie le pourquoi de cette robustesse et on traite de l'utilité des valeurs F_{inf}, F_{moy} et F_{sup} comme points de référence biologique.

The terms F_{low}, F_{med} and F_{high} were introduced in the ICES Working Group on Methods of Fish Stock Assessment (Anon. 1984, 1985), but the idea (the concept of F_{rep}) was first put forward by Shepherd and Sissenwine at the Dahlem Conference on Exploitation of Marine Communities (Beverton et al. 1984). The procedure for estimating these reference points is illustrated in Fig. 1. In a stock and recruitment plot straight lines that leave 90% (F_{low}), 50% (F_{med}) and 10% (F_{high}) of the points above the line are drawn through the origin. The slope of these lines is R/SSB (recruits per unit of spawning stock biomass) and the corresponding fishing mortalities can then be read from a SSB/R curve.

F_{med} corresponds to the level of fishing mortality where accessions to the stock by recruitment in half of the "observed" years has been more than sufficient to balance the losses due to mortality. F_{low} and F_{high} correspond to fishing mortalities where recruitment has been more than sufficient to balance the mortality in about nine years and one year out of ten, respectively.

The stability of these reference points for North-East Arctic cod and haddock was investigated by Jakobsen (1992), who concluded that they were quite robust in response to variation in the parameters used to estimate them. One particular property was that they were much less influenced by the choice of natural mortality rate (M) than F_{max} and $F_{0.1}$ which both are very sensitive to M. The results were, however, not claimed to be generally valid.

Because the estimation of F_{low}, F_{med} and F_{high} follows the same basic procedure and responds to variations in the parameters in a similar way, reference in the following text is usually made only to F_{med}. The stock and recruitment plot and its effect on the estimation of F_{med} is not treated in this paper, except in relation to the representativeness of the historical series and the effect of natural mortality.

The estimation of F_{med} is based on two sets of parameters. The fundamental set is the historical series of SSB and recruits. In the simulations presented in this paper the stock and recruitment plot is assumed to be fixed, i.e., the estimate of F_{med} from one year to the next will not be influenced by changes in the plot.

The other set of parameters are those used to calculate the SSB/R curve. These are weight-at-age in the stock, exploitation pattern, maturity ogive, and natural mortality rate. In assessments the latter is usually assumed to be constant over time, at least for the age groups in the spawning stock. The other parameters are, however, frequently subject to some change from one year to the next and this will result in some variation in the estimate of F_{med}, even if the stock and recruitment plot is unchanged. These variations are of the same type that short-term changes in growth and exploitation pattern cause in estimates of F_{max} and, to a lesser extent, $F_{0.1}$.

This paper deals with two aspects of the F_{med} estimation:

1. The year-to-year variation caused by the parameters used to calculate the SSB/R curve.

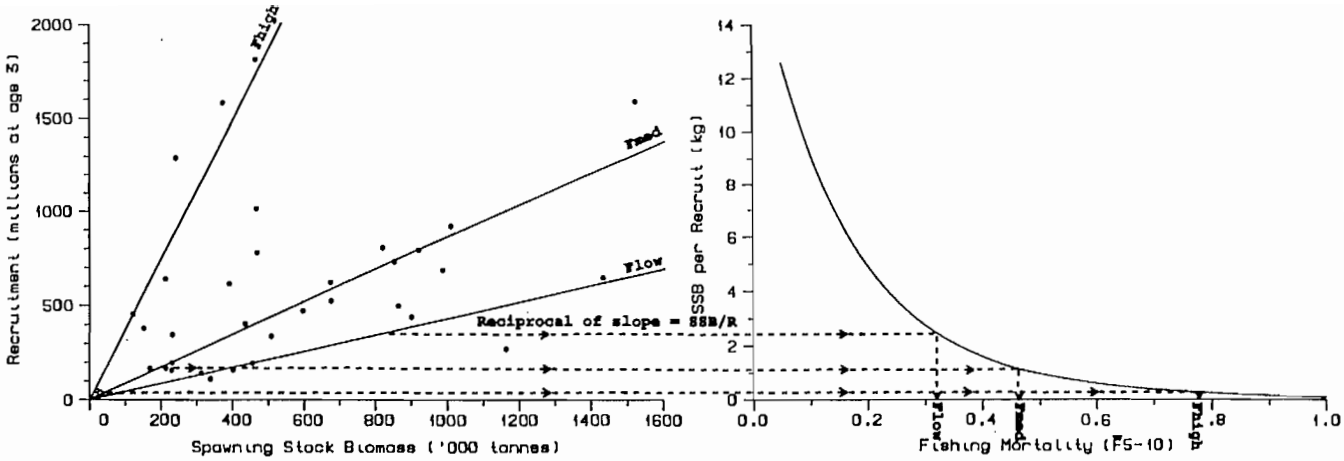


Fig. 1. Estimation of F_{low}, F_{med} and F_{high}.

2. The reasons for the robustness of F_{med} to changes in the assumption of natural mortality rate.

Materials and Methods

To test the effect of changes in growth, exploitation pattern and maturity ogives on F_{med}, three stocks were simulated. For each stock, there were three scenarios for each of the parameters weight-at-age, exploitation pattern and maturity ogive (Table 1), using the values in Scenario two for the other two parameters.

The growth data were based on the von Bertalanffy (1938) growth equation $L_t = L_\infty \left[1 - \exp(-kt)\right]$. The values used for k and L_∞ are not taken from any particular stock, but were chosen to roughly represent growth patterns seen in some commercially important stocks, e.g., of cod, haddock and herring. The values are given in Table 1. The weight was assumed to be proportional to the cube of the length. In Stock A and Stock B, k was kept constant in the three scenarios, whereas L_∞ was kept constant for Stock C. Exploitation patterns and knife-edge maturity ogives were chosen to be approximately what one might expect for stocks with the given growth rates. In the scenarios the exploitation patterns and maturity ogives were shifted by one age group up and down from the Scenario 2 values.

The effect of a change in natural mortality rate from year to year is not included in the calculations. Thus, in all the simulations $M = 0.2$. However, the reasons for the robustness of F_{med} in response to change in natural mortality rate for the whole historic period of the data series, were investigated.

Results

Growth, Exploitation Pattern and Maturity

The results of the simulations are shown in Figures 2–4. The SSB/R derived from the stock and recruitment plot is not given, but is represented by the range of values which, when applied to the SSB/R curve, give F_{med} estimates from 0.1 to 0.6 for Scenario 2. These are the "True F_{med}" values on the x-axis in the figures. The difference between the "true" Scenario 2 curve and the curves of Scenarios 1 and 3 represents the change in the F_{med} estimate resulting from the simulated shift in parameter values.

The differences in weight-at-age between the scenarios for each stock are considerable and are beyond the extremes for many stocks, especially if year-to-year variations are considered. For the North-East Arctic cod, however, larger variation than this has been experienced, although not for all age groups at the same time (Anon. 1992). The results of the simulations show that the estimate of F_{med}, as might be expected, will increase with growth (Fig. 2). The difference between the scenarios in the figure is increasing with the level of F_{med}, but is in relative terms larger for smaller values of F_{med}. For most stocks the effect of growth on the estimation of F_{med} on a year to year basis will be much smaller than in the simulations and for many probably negligible.

The effect of shifting the exploitation pattern by one age group is shown in Fig. 3. The estimate of F_{med} will increase if entry to the fishery is delayed. This effect is more dependent on the level of F_{med} than was the case for the effect of growth. The change in the exploitation pattern from year to year is normally small unless there are major changes in fishery regulations, e.g., area closures or increase of minimum legal mesh size in trawls. In such cases the effect on F_{med} can be considerable.

If maturity is reached at an earlier age than before, the estimate of F_{med} will decrease (Fig. 4). Although a shift of one year in maturity is more than experienced for many stocks, the age at maturity for the North-East Arctic cod is currently about three years younger than it was 40 years ago (Anon. 1992; Jorgensen 1990). Changes from one year to the next will be much smaller, but the effect on F_{med} could still be significant. Also, earlier maturity is likely to be accompanied

Table 1. Parameters for simulated fish stocks used in the calculations ($M = 0.2$ in all cases).

WEIGHT AT AGE (kg)

	Stock A $k = 0.05$; L_∞ variable			Stock B $k = 0.10$; L_∞ variable			Stock C k variable; $L_\infty = 100cm$		
	Scenario			Scenario			Scenario		
Age	1 $L_\infty = 180$	2 $L_\infty = 200$	3 $L_\infty = 220$	1 $L_\infty = 45$	2 $L_\infty = 50$	3 $L_\infty = 55$	1 $k = .084$	2 $k = .010$	3 $k = .119$
1	0.007	0.009	0.012	0.001	0.001	0.001	0.005	0.009	0.014
2	0.050	0.069	0.092	0.005	0.007	0.010	0.037	0.060	0.095
3	0.158	0.216	0.288	0.016	0.022	0.029	0.111	0.174	0.271
4	0.347	0.476	0.634	0.033	0.045	0.060	0.232	0.358	0.543
5	0.631	0.866	1.152	0.056	0.076	0.101	0.403	0.609	0.902
6	1.015	1.393	1.854	0.084	0.115	0.153	0.620	0.918	1.329
7	1.502	2.060	2.742	0.116	0.160	0.212	0.879	1.276	1.806
8	2.090	2.867	3.815	0.152	0.209	0.278	1.172	1.670	2.315
9	2.775	3.807	5.067	0.190	0.261	0.348	1.493	2.090	2.840
10	3.553	4.873	6.486	0.230	0.316	0.420	1.835	2.526	3.368
11	4.416	6.057	8.062	0.271	0.371	0.494	2.193	2.969	3.889
12	5.357	7.348	9.780	0.311	0.427	0.568	2.561	3.412	4.393
13	6.368	8.735	11.626	0.351	0.481	0.641	2.934	3.850	4.877
14	7.440	10.206	13.585	0.390	0.535	0.711	3.306	4.276	5.334
15	8.567	11.751	15.641	0.427	0.586	0.780	3.676	4.689	5.764

EXPLOITATION PATTERN

	Stock A			Stock B			Stock C		
	Scenario			Scenario			Scenario		
Age	1	2	3	1	2	3	1	2	3
1	0.0	0.0	0.0	1.0	0.5	0.0	0.1	0.0	0.0
2	0.1	0.0	0.0	1.0	1.0	0.5	0.6	0.1	0.0
3	0.4	0.1	0.0	1.0	1.0	1.0	1.0	0.6	0.1
4	0.7	0.4	0.1	1.0	1.0	1.0	1.0	1.0	0.6
5	1.0	0.7	0.4	1.0	1.0	1.0	1.0	1.0	1.0
6	1.0	1.0	0.7	1.0	1.0	1.0	1.0	1.0	1.0
7+	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0

AGE AT MATURITY (knife-edge)

Scenario	Stock A	Stock B	Stock C
1	5	1	3
2	6	2	4
3	7	3	5

by increased growth, which both will increase the estimate of F_{med}.

Natural Mortality

In fish stock assessments M is, with few exceptions, assumed to be constant from year to year and there is usually little evidence to support the value chosen. If M is changed in a Virtual Population Analysis (VPA), both fishing mortalities and stock size will change. If M is increased, F will decrease and stock numbers will increase. Furthermore, F at all age groups will decrease by roughly the same amount M is increased by, provided that the VPA is appropriately tuned. The explanation for this is that the total mortality rate ($Z = F + M$) in a VPA tends to remain fairly constant over a reasonably chosen range of M values, as shown by Ulltang (1977).

Jakobsen (1992) showed that F_{med} is more robust than F_{max} to changes in the choice of natural mortality rate (M) in the assessment of North-East Arctic cod and haddock and

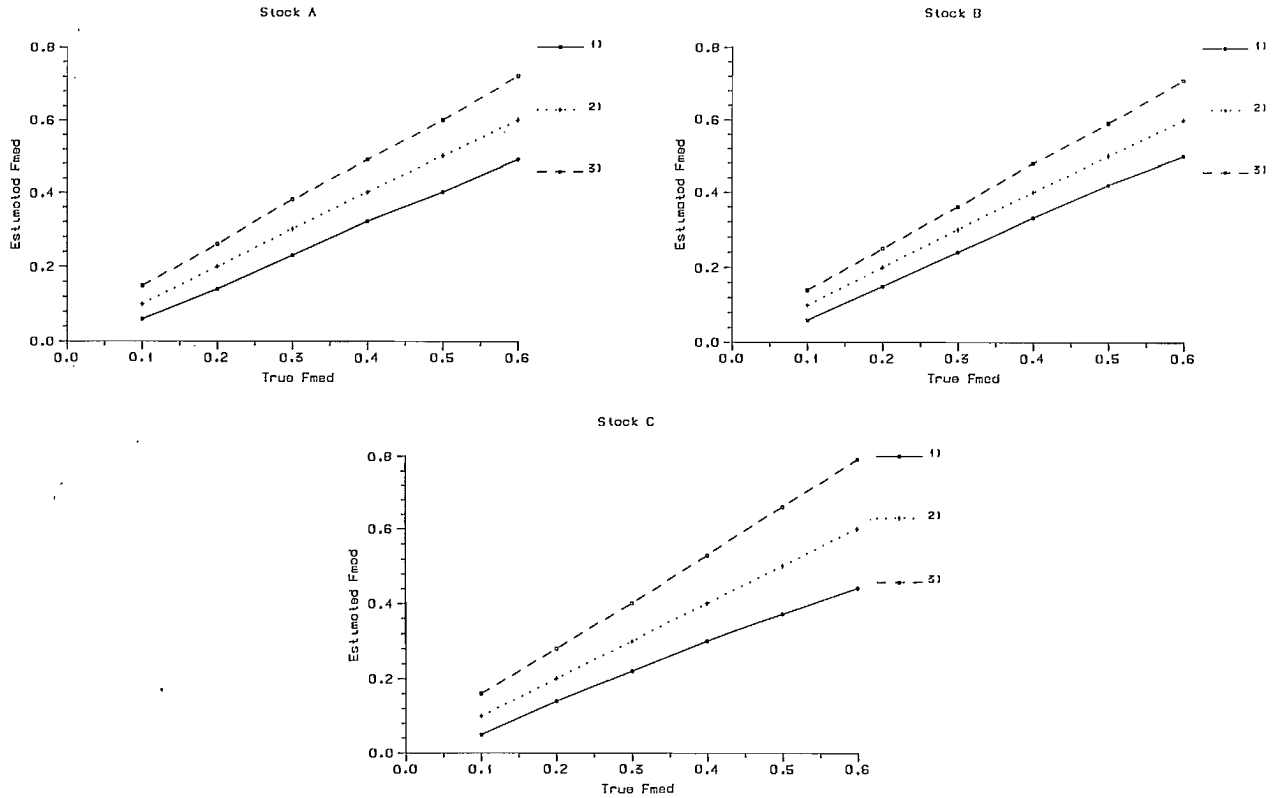


Fig. 2. Effect of weight-at-age on the estimation of F_{med}. Numbers on curves represent scenarios (Table 1).

even more robust when it is compared to the current level of fishing mortality. The results for North-East Arctic cod are reproduced in Fig. 5. It should be noted that the effect of M on the ratio between the reference points and current F is dependent on the level of the latter. For instance, if current F had been close to the F_{med} level, the ratio $F_{\text{med}}/F_{\text{current}}$ would have been close to 1 for all choices of M, whereas the ratios for $F_{0.1}$ and $F_{\max}$ would have been even more sensitive. This robustness appears to be a general property of F_{med}, but it has not been obvious why this is so.

The relative change in stock numbers in a VPA will not be the same for all age groups when natural mortality is changed. Usually the relative change is biggest for the youngest age groups. Using the catch equation of the VPA:

$$C = N\frac{F}{Z}(1 - \exp(-Z)),$$

where C is given and assuming that Z is unchanged, it is seen that the ratio between two stock numbers N_a and N_b estimated from different natural mortality rates M_a and M_b is approximated by the equation

$$\frac{N_a}{N_b} = \frac{F_b}{F_a}$$

If $M_b = M_a + \Delta M$, then $F_a = Z - M_b$, $F_b = Z - M_a - \Delta M = F_a - \Delta M$, and

$$\frac{N_a}{N_b} = \frac{F_a - \Delta M}{F_a}$$

This ratio will differ most from 1 when F_a is low. When M is changed the relative difference in stock numbers will therefore be biggest for the youngest, not fully recruited, age groups than for the older ones.

Consequently, the relative change in recruitment will be larger than the relative change in SSB, and there will be a change in the SSB/R ratio estimated from the stock and recruitment plot. However, the change in the ratio is linked to a change in the historic exploitation pattern which will be carried over into the exploitation pattern used to calculate the SSB/R curve. This will give an SSB/R curve corresponding to the new VPA, compensating for the change in SSB/R from the stock and recruitment plot. The change in F_{med} will therefore be similar to the change in F in general, thus changing M by for example 0.1 will give approximately the same change in F_{med}, but in the opposite direction.

Since F_{med} and current F are changed by approximately the same amount (ΔM), the ratio between them will change from

$$\frac{F_{\text{med}}}{F_{\text{current}}} \text{ to approximately } \frac{F_{\text{med}} - \Delta M}{F_{\text{current}} - \Delta M}$$

Thus, the change in ratio will depend firstly on the difference between F_{med} and current F. If they are at the same level, the ratio will be virtually unchanged. If they are different, the size of ΔM relative to the F values will be of importance.

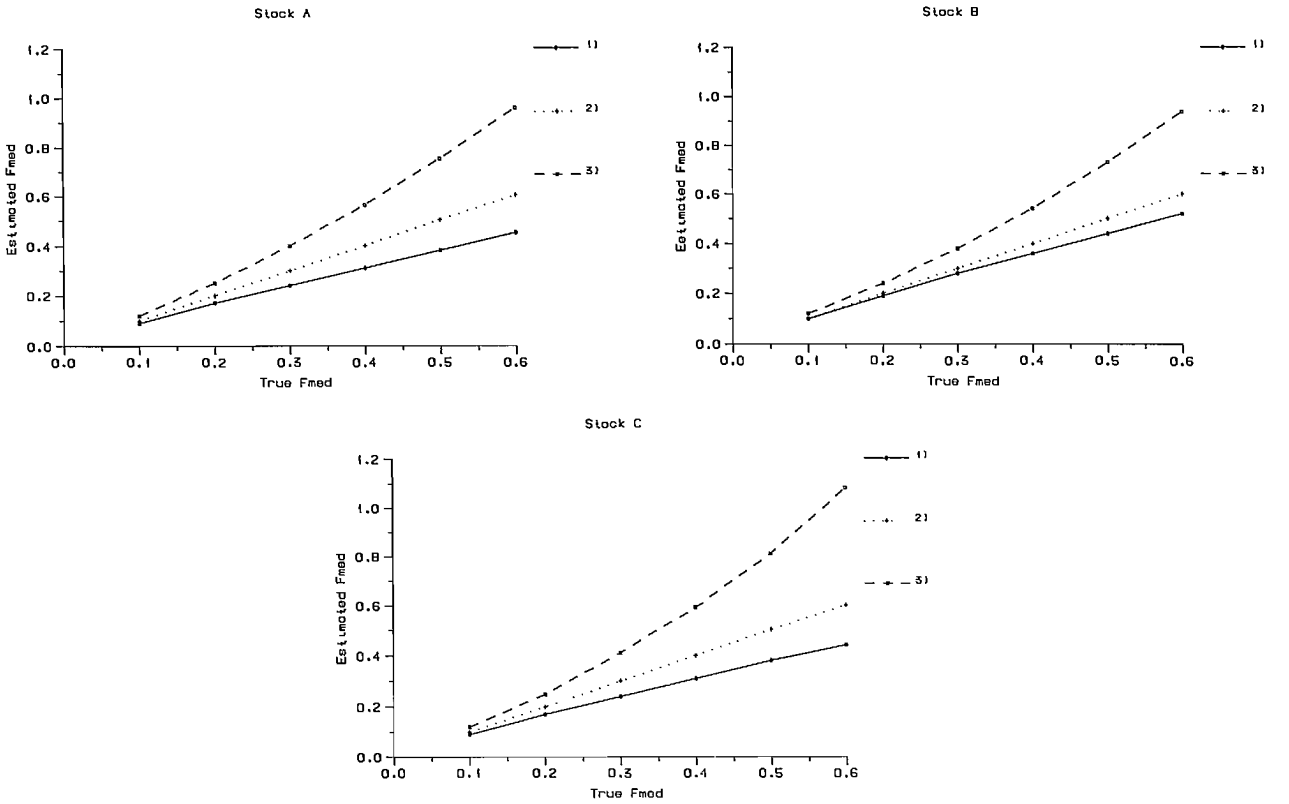


Fig. 3. Effect of exploitation pattern on the estimation of F_{med}. Numbers on curves represent scenarios (Table 1).

The main difference between F_{med} and F_{max} in their response to change in M is that F_{max} will be shifted in the same direction as M (Beverton and Holt, 1957) and accordingly in the opposite direction of the current F. The ratio of F_{max} to current F is therefore much more sensitive to errors in the choice of M than the ratio of F_{med} to current F. Although $F_{0.1}$ is less sensitive to M than F_{max} is, it behaves similarly and its ratio to current F is therefore more sensitive than for F_{med}.

Discussion

In Yield- and SSB/recruit calculations the parameters (weight-at-age, exploitation pattern, maturity ogive) predicted for the year for which management advice is given are normally used. This seems appropriate, because these parameters decide how a given fishing mortality will affect the stock in that year. However, this means that the Y/R and SSB/R curves may change from year to year and accordingly also the reference points estimated from these curves. These reference points, including F_{max} and F_{med}, are therefore not by definition fixed values. Such changes in the estimation of F_{med} will normally be small and are not likely to be very different from those experienced with F_{max}. The changes can, however, in some cases be considerable and the fact that the biological reference points are not fixed is in itself enough to cause some confusion in management considerations.

It can be argued that long-term average rather than recent values of the parameters should be used in the estimation of biological reference points. One reason is that year to year variation in estimates of F_{med} and F_{max} not always reflects real or significant changes in the parameters. The main advantage, however, would be more stability in the estimates. Considering the degree of accuracy in estimation of biological reference points in general, loss of precision in using long-term values is hardly an argument against it. However, one exception is the case where there is reason to assume permanent changes in the parameter values. This would for example be the case when regulations affect the exploitation pattern.

Since the choice of M is somewhat arbitrary, this can cause errors in the estimates of the biological reference points. In management considerations, it is the level of the biological reference points relative to current fishing mortality that is usually looked at. In this respect, F_{med} is clearly much more robust than F_{max}.

The usefulness of F_{med} as a biological reference point is not generally accepted. One reason for this may be that biological reference points frequently have been used as target levels of fishing mortality in management advice. This understanding still persists to some extent, in spite of repeated statements that they are to be treated just as markers. If seen as a target level, it is easy to point to the fact that F_{med} for many stocks are estimated on the basis of a historical period where the stock has been at a low level (Corten 1990), and that fishing at F_{med} therefore may prevent a rebuilding of the stock.

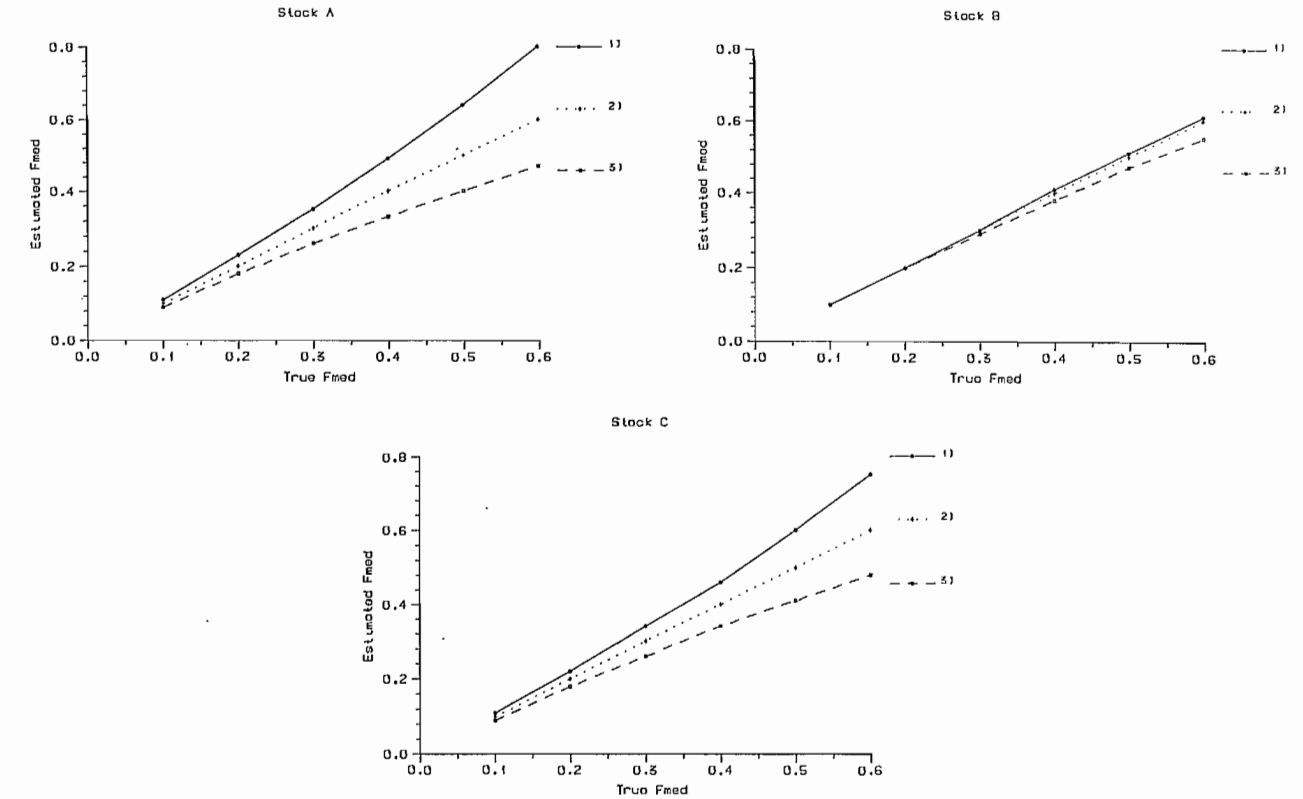


Fig. 4. Effect of maturity ogive on the estimation of F_{med}. Numbers on curves represent scenarios (Table 1).

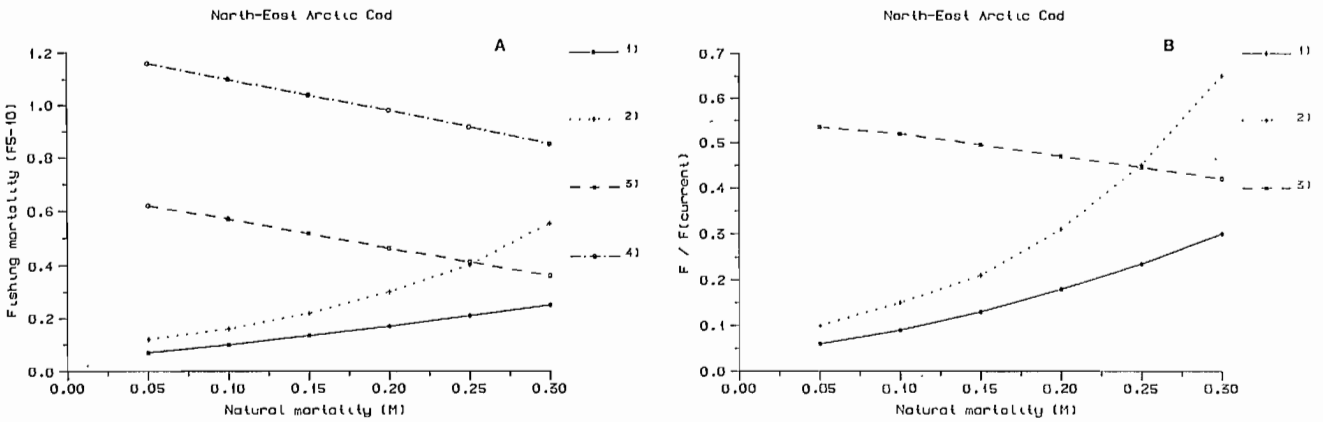


Fig. 5. The effect of natural mortality rate on the estimation of biological reference points. A) Absolute change in estimates. 1) $F_{0.1}$, 2) F_{max}, 3) F_{med}, 4) $F_{current}$. B) Change in estimates relative to estimated current F. 1) $F_{0.1}$, 2) F_{max}, 3) F_{med}.

While it is important to keep in mind that F_{med} reflects only the stock history during the period from which it is estimated, aiming at F_{low} will be a fairly safe strategy to start rebuilding a stock, even if the historic time series is short. However, fishing at F_{low} will normally mean severe restrictions in the fishery. On the other hand, if current F is close to F_{high}, there are good chances that the stock soon will decline to levels of SSB not previously experienced.

All biological reference points are subject to some sort of error and should not be given more weight in the management advice than they are worth. One reference point may be useful for one stock but clearly unreliable for others and a sound judgment is needed in all cases. F_{med} has its limitations, but also the strength of being based on historical evidence, and seems to be no less reliable than other reference points, e.g., $F_{0.1}$ and F_{max}, that are often used without reservations.

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The Barents Sea capelin stock collapse: A lesson to learn

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The Barents Sea capelin stock nearly collapsed in the period 1983–1986, and the fishery was banned in the period 1987–1990. The most likely factors causing the collapse are discussed in connection with the model that was used for management. The biological reference point applied was an optimal spawning stock for MSY, and the paper discusses if there presently is adequate knowledge to apply this reference point. A new model is presented that utilizes results from the ongoing multispecies research in the Barents Sea.

Le stock de capelans de la mer de Barents s'est presque effondré entre 1983 et 1986, et la pêche a été interdite entre 1987 et 1990. Les facteurs les plus probables de cette situation sont traités en rapport avec le modèle de gestion qui était utilisé. Le point de référence biologique appliqué était un stock reproducteur optimal pour un rendement maximum soutenu, et les auteurs de l'article se demandent si les connaissances actuelles sont suffisantes pour l'application de ce point de référence. On présente un nouveau modèle dans lequel on utilise les résultats de recherches plurispécifiques en cours dans la mer de Barents.

The Barents Sea capelin stock declined severely during the period 1983 to 1986, and did not start recovering until 1990. It is believed that the decline was initiated by an abrupt change of oceanographic conditions during the winter 1982–1983. An increase in the inflow of Atlantic water caused very good recruitment conditions for the Norwegian spring spawning herring and the Northeast Arctic cod stock. During the fall of 1983 it is likely that the 0-group herring caused a decline in capelin recruitment by predation (Moxness and Øiestad 1979) and that the negative influence of herring on the capelin recruitment accelerated as the herring grew older. As the 1983 year class of cod grew older, the natural mortality of adult capelin also increased rapidly. Most likely as a consequence of the vanishing of the capelin as source of food, the individual growth of the cod decreased dramatically. The model used for management prior to the stock decline — CAPELIN — (Tjelmeland 1985; Hamre and Tjelmeland 1982) did not take into account these effects.

The management for the years 1991 and 1992 introduced a new element: the role of the cod as predator on capelin. However, the management objective from the pre-decline period, i.e. to aim at a spawning stock level of capelin of about 0.5 million tonnes, was unchanged. This paper analyses some of the problems arising when the cod-capelin-herring interaction is considered in capelin management.

The capelin model presented here uses submodels for only three processes: maturation of capelin, predation from cod on mature capelin and recruitment of capelin. The other processes important for management relevant modeling, i.e., natural mortality on immature capelin and growth, are incorporated without using models relating the respective processes to modeled or exogenous variables.

For an appropriate analysis of the significance of various error sources of an exploited stock, it is how the errors affect the final "product", i.e., the quota, that is of interest. Therefore, both a mathematical model for the species considered and a management decision rule based on the model should be constructed. This paper is a first attempt at providing a basis for such a comprehensive analysis for the Barents Sea, where the uncertainty in the underlying data and submodels play a vital role.

We have omitted many details concerning the estimation of parameters, and we have not carried the analysis far towards applications. The main goal of the paper is to show that management decision rules where the uncertainties and species interactions play an integrated role may be found for the Barents Sea capelin. More details concerning the estimation of the parameters as well as more comments on the uncertainty of the individual processes may be found in Tjelmeland (1992).

Biology

The biological system is described by Hamre (1990a). The features of the system that are considered in this paper can be described on the basis of Fig. 1.

Life History of Capelin

In the autumn the capelin stock feeds in the northern parts of the Barents Sea, mainly north of 74°N in the western regions. In the eastern regions the capelin may be found far south also in the autumn but not in great abundance and consisting of mostly young fish. The capelin stock has been surveyed each year during September since 1972 by a joint

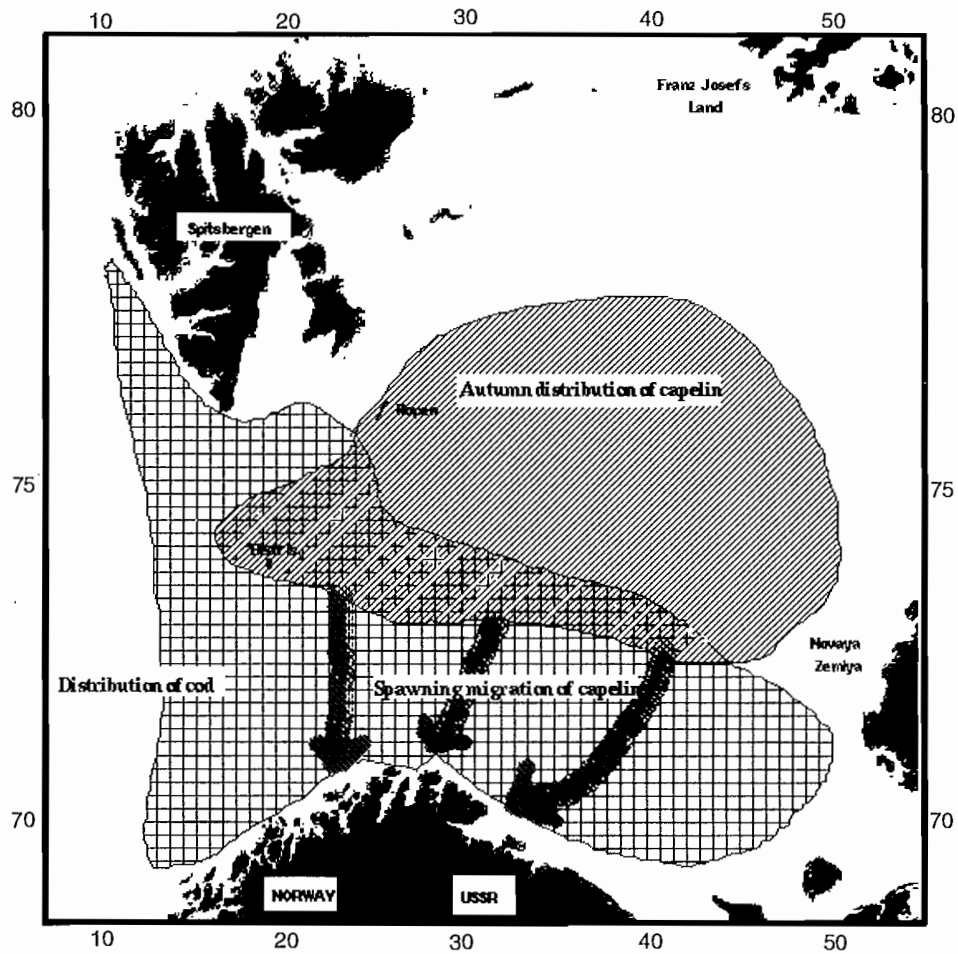


Fig. 1. Distribution of cod and capelin in the Barents Sea with the spawning migration of capelin indicated.

Soviet-Norwegian cruise using acoustic information combined with trawl data. These survey data are the foundation of the capelin modeling and management.

The geographical distribution in the autumn may vary strongly from year to year, in some periods being more southerly and westerly and in other periods being more to the north and east, see distribution maps in Dommasnes and Rottingen (1985).

At the end of the year and the following winter the mature part of the population moves southwards and will eventually spawn along the coast of Norway and Russia in April. In some years the capelin will have a westerly spawning migration, in other years an easterly spawning migration. The mechanisms governing the geographical variation of the spawning migration are poorly understood, although there have been some attempts to make conceptual models (Ozhigin and Luka 1985; Tjelmeland 1987a).

In the Barents Sea it has not yet been possible to obtain a direct measurement of the spawning stock. Therefore, one is totally dependent on mathematical models that cannot be directly verified against data to assess both a) the amount of capelin that will approach the coast to spawn and thereby be predated upon by cod and fished by humans and b) the effect of the predation on the capelin stock and hence the amount of capelin that is left to spawn. Recent multispecies research has shown that the consumption and, worse, the variation in consumption from year to year, from cod on capelin may be large, and exceed the winter fishery (Mehl 1989, Bogstad and Mehl 1992). The present paper is a first attempt to pave the ground for a logically consistent management of capelin where the predation from cod is taken into consideration, and where also the capelin as source of food for cod may be taken into account.

It is believed that most of the mature capelin die after spawning. However, in recent years this assumption has been challenged, both in Norwegian public discussions and by requests from ACFM to the ICES Atlanto-Scandian Herring and Capelin Working Group.

The immature capelin may also migrate far southwards, but at a later time than the mature component. The data for

assessing the migration of the immature capelin are poor, but an overall impression is that the extent of the southwards migration may vary from year to year and that the most southern distribution may be in the months May and/or June. The northwards migration will take place in July and August.

Predation from Cod

During the spawning migration the capelin will overlap fully with the cod stock, and be subject to heavy predation. The magnitude of the predation will depend on the migration route, because there is an east-west gradient in both the abundance and age distribution of cod.

It is likely that the cod stock's predation on immature capelin is more variable than the predation on mature capelin, since the migration, and thereby the overlap, may be more variable. But there is no doubt that in some years the predation from cod also on immature capelin may be highly significant. According to Mehl (1989) and Bogstad and Mehl (1992), capelin may comprise up to 50% of the diet of cod on a yearly basis.

It has been observed on 0-group cruises in the Barents Sea that the 0-group cod may eat 0-group capelin. In years of high cod recruitment, the capelin recruitment may therefore be hampered.

Predation from Herring

In years of good recruitment herring juveniles may enter the southern Barents Sea in great quantities. The herring will grow up in the southeastern part and gradually move westwards as it grows older. It will leave the Barents Sea at age 3 to 4 (Hamre 1990b).

During the 0-group stage there may be considerable overlap between herring and capelin. Due to the size difference, the herring may prey on capelin, thus having an influence on capelin recruitment (Moxness and Øiestad 1979).

In connection with an oceanographic event in 1982–1984 where the inflow of Atlantic water into the Barents Sea abruptly increased, the recruitment to the herring stock was very good, and the 1983 year class of herring had a significant impact on the Barents Sea ecosystem during its presence. In particular, the herring may be considered one of the major causes for the Barents Sea capelin stock decline in 1983–1986 (Hamre 1988, 1990a). Events of very rich year classes of herring may occur once or twice every 10-year period, see a discussion in Hamre (1990b).

The Fishery

The capelin fishery is conducted in two seasons. In the autumn season from August to December both immature and maturing capelin are caught. In the winter season from January to April the catch consists mainly of mature capelin.

Development of the Capelin and Cod Stocks

Population estimates of the capelin stock from acoustic surveys are presented in Fig. 2, while VPA estimates for cod are given in Fig. 3.

The present model is based on a conceptual model for the Barents Sea ecosystem developed by Hamre (1990a). In accordance with this model, the period 1972–1991 may be divided into three parts:

1. 1972–1983. The capelin stock peaked in 1975, because of small predation pressure from herring and because slower than usual growth led to late maturation. The decline from 1975 was mostly caused by large portions of the stock maturing, but also by high fishing pressure. The development of the capelin stock can in this period be understood using a one-species model where the maturation process is length-dependent.
   The cod stock had a steady decline because of high fishing pressure and low recruitment.

2. 1983–1989. The capelin stock nearly collapsed after the strong 1983 year class of herring entered the sea and because the cod stock started to recover with strong year classes from 1982 onwards. The driving forces are stock interactions and the development can no longer be understood in a single-species context.
   The strength of the 1982–1986 year classes of cod at the 0-group and juvenile stage created an optimism that was not justified. The 1984–1986 year classes turned out to be poor when they entered the fishery. The subsequent decline of the cod stock may partly be caused by cannibalism and discards. An important factor might also be a higher fishing pressure than anticipated when the quotas were set, because the individual growth turned out to be much lower than anticipated, probably connected to the decline of the capelin stock.

3. 1990. The capelin stock is back in full strength, and the cod stock is recovering due to good individual growth and improved recruitment.

Management

The management of the Barents Sea capelin up to 1984 is described by Hamre (1985). Prior to 1982 the stock was managed by using a spawning stock of 0.5 million tonnes as a



Fig. 2. Barents Sea capelin 1972–1991. No. 1+, billion. September acoustic survey.



Fig. 3. North-East Arctic cod 1972–1991. No. 3+, million (January 1). 1991 VPA estimate.

lower safeguarding limit. This number was based on a comparison of historical calculations of the spawning stock and observed recruitment. In the autumn 1982 results from Hamre and Tjelmeland (1982) and the CAPELIN model (Tjelmeland 1985) was used to introduce MSY management. The optimal spawning stock of 0.4 million tonnes calculated by the CAPELIN model did not differ much from the previous used safeguarding limit. It is worth noting that already in 1982 multispecies considerations were taken into account in the management of the capelin stock, since the value of the stock as source of food for predating organisms was explicitly considered (Anon. 1983).

In 1984 the first signs of capelin recruitment failure were manifest (Anon. 1985) as the abundance of one year old capelin was extremely low. The implications of this observation were not fully understood, to a large extent because the acoustic abundance estimate of the one-group had proven not to be reliable on earlier occasions.

At its 1985 meeting (Anon. 1986) the Atlanto-Scandian Herring and Capelin Working Group noted that the capelin stock had decreased far more than could be accounted for by the fishery. The conclusion was that environmental changes had taken place that invalidated the model. The Working Group did not recommend any fishing in 1986. The Mixed Soviet-Norwegian Fishery Commission, however, set a catch quota of 0.12 million tonnes for the 1986 winter fishery. The fishery was closed in 1987–1990.

In 1990 the stock was estimated at 5.8 million tonnes and in 1991 at 7.1 million tonnes by the acoustic survey in September. The fishery was reopened in 1991 with a catch quota of 0.85 and 0.25 million tonnes for the winter and autumn fishery, respectively.

The quotas prior to 1986 were given in a two-step procedure, based on the September survey:

1. A winter quota was suggested by simulating the stock until spawning the following winter.

2. An autumn quota by simulating the stock until the winter one and a half year ahead.

There has been a suggestion from the scientists involved to reduce the autumn quota because of the higher uncertainty involved in another year of simulation (Anon. 1985).

In 1990, the consumption of capelin by cod was, for the first time, taken explicitly into account when setting the capelin quota for the coming year.

Model formulation and parameter estimation

Old Formulation — CAPELIN

The assessment model CAPELIN used up to the stock decline had the following characteristics:

1. A knife-edge maturation function with length.

2. Yearly constant, but estimated, natural mortality on immature capelin.

3. Yearly constant, but estimated, growth.

4. Yearly constant and not estimated natural mortality on mature capelin.

5. Estimated Beverton-Holt spawning stock-recruitment relationship.

6. One month time step.

Besides splitting the capelin stock into males and females, the present paper elaborates on the submodels 1 and 5 and treats the mortality and growth of immature capelin not as yearly constants but as randomly fluctuating processes.

In the CAPELIN model the natural mortality was modeled using a constant and rather unsubstantiated natural mortality coefficient, while in the present paper the actual size of the cod stock together with estimates of the predating potential per cod is used. Therefore, no comparison between old and new spawning stock biomasses for capelin can be made.

New Formulation — CAPSEX

In the following, a revised model, CAPSEX, will be described and the parameters will be estimated process by process. The natural starting point is the maturation, because in the present version of the model the estimated values of the maturation parameters will not be dependent on the values of other parameters. This follows from the conceptual model of the life history of capelin presented earlier and from the fact that the stock data are from September, prior to the separation of the stock into a mature and an immature component. The converse is not true. The estimated values of the parameters governing growth and natural mortality will be strongly dependent on the estimated values of the maturation parameters.

The maturation of capelin is believed to be above all a length-dependent process, although for a given length there might be a higher probability for maturation with higher age (Forberg and Tjelmeland 1985).

Formulation

The maturing fraction in each length group, applied on the September cruise data, is given by:

$$m(l) = \frac{1}{1 + \exp\left\{4P_1^{\text{sex}}\left(P_2^{\text{sex, age}} - l\right)\right\}}$$

where l is the fish length in cm.

The estimation of maturation parameters poses severe difficulties. A study of gonads reveals that maturation also may depend on age. With some support from Forberg and Tjelmeland (1985) we fix P_1^{sex} to 0.6 for both sexes. In Tjelmeland (1987b) a relation between maturation parameters by sex and age was assessed from gonadal data and an overall scaling was estimated by using the model. In Tjelmeland (1987a) the maturation was taken from gonadal data alone. Maturation studies should be done in the future using all collected gonadal data to lessen the burden on model estimations.

The natural mortality on immature capelin is supposed to be the same for all age groups and both sexes.

Estimation Method

The data input to the model is the acoustic estimate obtained during September each year, the catch in number and the weight in the catches by age and month, assuming all catch in the period January–April to be on mature capelin. The analysis will for the time being be restricted to utilizing only these data, although several other sources of data exist that may improve the basis for parameter estimation and that may be a foundation for refined models. For the moment disregarding recruitment, the biological processes affecting the population from one year to the next are the maturation and mortality. The parameters in the maturation and mortality models will be estimated by comparing the number of fish at age 3, 4 and 5 years simulated by starting the model in the previous autumn, to corresponding measured values.

Table 1. Bootstrap estimates of capelin maturity parameters and covariance matrix.

	Length at maturity P_2	
	females	males
Means	13.65 cm	14.04 cm
Covariance matrix		
	0.078	0.438
		0.125

In choosing the estimation method, assumptions on the stability of the parameters have been made:

1. The probability of a fish of given age, sex and length to mature is constant from year to year.
2. The mortality is regarded as fluctuating, i.e., no pre-estimation assumptions on the mortality is made.

Thus, the following estimation scheme is adopted:

1. The maturation parameters are fixed throughout the whole time period of estimation.
2. The mortality is estimated year by year.
3. The estimation is performed by varying the maturation parameters until the least-squares goal function attains its minimum value.

The estimation is thus performed in a double iteration.

Uncertainty Estimates

There is not enough knowledge of the uncertainties connected to the acoustic estimate to construct a goal function that gives the probability of obtaining the actual measurements given that the model is true, i.e., a maximum likelihood method for constructing the parameter confidence intervals cannot be used. Therefore, resampling is used. From the period of available data, 18 one-year periods are picked at random with replacement and the estimation of maturation and mortality parameters was performed (naive bootstrap). Eighty-five estimations are performed and the parameter confidence intervals and correlation are calculated from these estimates (Table 1).

Goal Functions

Two different goal functions have been tried, both using unweighted least squares:

1. Comparing absolute simulated number of fish in each age group and for each sex to absolute measured number of fish in each age group and for each sex.
2. Comparing the ratio of simulated number of fish to measured number of fish to 1.0 in each age group and for each sex.

Table 2. Summary of mortality, growth and recruitment estimates for capelin in the Barents Sea.

	Mean value	Minimum value	Maximum value	Lower 75% quantile	Upper 75% Quantile	Data points
Mortality of immature capelin (month^{-1})	0.115	-0.046	0.249	0.047	0.140	18
Growth in length of immature females (cm year^{-1})	2.40	1.43	3.15	2.18	2.44	17
Growth in weight of immature females (g year^{-1})	7.28	2.51	10.52	6.09	7.81	13
Growth in length of immature males (cm year^{-1})	3.20	2.41	3.95	2.83	3.12	16
Growth in weight of immature males (g year^{-1})	11.50	7.25	16.02	8.61	12.19	13
Overlap variable	1.23	0.91	1.81	0.94	1.70	6
Increase factor weight mature, 2–3 years	1.75	0.88	2.00*	1.20	2.00*	12
Increase factor weight mature, 3–4 years	1.29	0.72	1.72	1.11	1.34	12
Increase factor weight mature, 4–5 years	1.12	0.53	1.76	1.12	1.32	12
Recruitment at age 2, deviation from model in billions. (Mean measured recruitment = 168 billion.)	–	-129	78	-44	60	15

* Estimated at the limit

Given that the maturation and mortality models both are correct, the two goal functions should give the same estimated values for the parameters. However, both the maturation model and the mortality model are highly idealized. Thus, differences in the estimated values tell us something about how the model assumptions deviate from the realities. For instance, goal function 1 put more weight on the more numerous 3 year old fish than on older fish. If there is something wrong with the assumptions on age dependence, we will get different estimates.

It is not straightforward to determine how simple the models of maturation, spawning survival and mortality can be. We have made 24 series of estimations using the two different goal functions; ages 3 and 4 or ages 3, 4 and 5; increased length at maturity for 2 year old fish for females, males and both sexes; and no spawning survival or survival of females.

None of the 24 estimations performed significantly better than the simple model which used goal function 2, ages 3 and 4, no spawning survival and no age dependence of maturation parameters; therefore this model was chosen.

Consistency with Weight at Age in the Catches of Mature Capelin

Once the maturation parameters are determined, the mean weight at age of the maturing population is determined. During the estimation of the maturation parameters, the simulated individual weight of the maturing capelin from October 1 to spawning at April 1 is not changed. The mean weight at age is compared to the mean weight at age in the catches of mature capelin in the period January–April. If the simulated weight is higher than the measured weight there is an inconsistency if no real weight decrease has taken place. In this case, the goal function has been increased with a (somewhat arbitrary) penalty function, which is the quadratic deviation of the ratio between simulated and measured weight from unity.

The weight at age data from the catches in the mature population may be difficult to interpret. Some of the catch may have been taken on immature capelin. Also, it is difficult to construct reliable weight at age because the geographical



Fig. 4. Estimates of natural mortality of immature capelin (month^{-1}) in the Barents Sea.

distribution of the catch may differ from the geographical distribution of the mature stock. Hence, a gradient in the weight at age may cause errors in the estimate of mortality of immature capelin given in Fig. 4. The data used are weighted averages of Soviet and Norwegian catches.

We consider the use of weight at age data from the catch statistics to be the weakest part of the analysis presented in this paper.

Estimation Results

The length at maturity estimate is 13.68 cm for female and 14.00 cm for males when fixing the maturation over the whole time period. These estimates are reasonably close to the bootstrap estimates in Table 1 when the bootstrap variances are taken into account.

Natural Mortality of Immature Capelin

The natural mortality of the immature part of the stock can be estimated once the maturation parameters are estimated. A summary of the mortality estimates is shown in Table 2. The overall impression is that the mortalities were considerably higher and more variable in the 80's than in the 70's. Later, mortalities will be drawn at random using these values, with no account for autocorrelation. A model that incorporates long-time trends in mortality might considerably reduce variance in the method suggested in this paper.

Growth of Immature Capelin

A model relating the growth of the immature fish from September one year to September the next year to model variables has not been used. It was assumed that the growth is independent of length and independent of age. When the maturation parameters are fixed, the yearly growth is given. A summary is shown in Table 2.

Growth of Mature Capelin

Growth of mature capelin is simulated by estimating the difference in weight in the April catch to the mean weight at age in the modeled mature population in September the previous year, where the latter is dependent on the estimated values of the maturation parameters. A summary is shown in Table 2. The total growth of the mature capelin is evenly allocated over October to March.

Predation from Cod on Mature Capelin

The effect of predation on mature capelin is evaluated by using parameters estimated by the area-distributed model MULTSPEC (Bogstad and Tjelmeland 1990, 1992). The MULTSPEC predation equations are formulated in an area-integrated form as follows (details concerning different units in MULTSPEC and CAPSEX are omitted):

The capelin abundance is transformed to capelin density:

$$\text{Capelin concentration} = \frac{\text{Capelin abundance}}{\text{Area size}}$$

Total food concentration is given by:

$$\begin{aligned}\text{Total food} &= \text{Overlap} \times \text{Capelin Concentration} \\ &\quad + \text{Other food concentration}\end{aligned}$$

The individual cod's predation ability is made dependent on the age. The mature cod at this time of the year migrating towards the spawning grounds is supposed not to prey on capelin. The maximum consumption by cod on capelin then becomes:

$$\begin{aligned}\text{Cod} &= P_4 \times \text{Temp} \times \sum_{\text{age}=3}^{\text{Max age}} N^{\text{Cod}}_{\text{Year}}(\text{age}) \\ &\quad \times \left(1 - \text{Ogive}^{\text{Cod}}_{\text{Year}}(\text{age})\right) \times W^{\text{Cod}}_{\text{Year}}(\text{age})^{0.802}\end{aligned}$$

The temperature effect is given by:

$$\text{Temp} = \exp\left\{0.104 \times T_{\text{Year}} - 0.000112 \times T^3_{\text{Year}} - 1.5\right\}$$

The feeding level is given by:

$$f = \frac{\text{Total food}}{\text{Total food} + P_3}$$

The consumption on capelin then becomes:

$$\text{Cons} = \text{Cod} \times f \times \frac{\text{Capelin concentration}}{\text{Total food concentration}}$$

and the natural mortality is calculated as:

$$M = -\ln\left(1.0 - \frac{\text{Cons}}{\text{Capelin concentration}}\right)$$

The parameters P_3, P_4 and the other food concentration are taken from estimation work with MULTSPEC (Bogstad and Tjelmeland 1992). The size of the area used in converting abundance to concentration was set to the sum of MULTSPEC areas in which the capelin occurs during the spawning migration.

In this paper we assume that the spawning takes place on April 1, and that the predation from cod is the only natural mortality on mature capelin in the period January 1 to April 1.

Estimation of the Overlap Variable

The overlap variable has been estimated by specifying that the capelin spawning biomass should be the same using MULTSPEC and CAPSEX for the years 1984–1989, provided the predation parameters and capelin maturation parameters are the same. A summary is shown in Table 2. In using CAPSEX to estimate the maturation parameters and the parameters estimated by MULTSPEC to evaluate predation on mature capelin it is assumed that the amount of late maturing capelin (i.e., capelin spawning in June–July) is negligible.

Cod Data and Historic Runs

The cod data used are the VPA-estimate made by the Arctic Fisheries Working Group (Anon. 1992). We make the assumption that only immature cod eats mature capelin in the period January–March and the immature part of the stock is calculated using the maturity ogive given by the Working Group. This assumption is made because we know that the mature part of the cod stock migrates westwards to the Lofoten/Vesterålen area in this period.

The weight at age data are the same as those used by the Working Group. However, it is evident from the Working Group report that there is a larger discrepancy in weight at age between the Soviet late autumn cod survey and the Norwegian young cod survey the following winter than can be accounted for by growth. The working group uses an arithmetic average of the two surveys. Recent investigations indicate that differences in age reading may explain much of the discrepancy.

Errors in the maturity ogive and weight at age may have a substantial impact on the management of capelin, because the calculated consumption from cod on capelin, and hence the calculated capelin spawning stock biomass, may be seriously affected.

Recruitment

A recruitment model incorporating the influence from herring was used:

$$R = P_5 \frac{B}{B_{\frac{1}{2}} + P_6 \times H_0 + P_7 \times H_{1+} + B}$$

where,

B = Spawning stock biomass

R = Number of recruits (2 year old)

$B_{\frac{1}{2}}$ = Spawning stock half-value neglecting influence from herring

H_0 = Index of 0-group herring

H_{1+} = index of older herring

A summary of the results of the estimation is included in Table 2. This model accounts for 67% of the variance in the data.

The above recruitment model has the property that even if there may be great variation in recruitment due to fluctuation in the amount of herring, an additional amount of spawning capelin will always give an additional amount of recruits.

The index of 0-group herring is drawn at random from the 0-group indices calculated from the yearly 0-group survey in the period 1972–1991 (Anon. 1991).

The index of older herring is calculated by drawing a 1983-type herring event with a probability of two instances in an 18-year period.

The recruitment is calculated by using the above model and adding a stochastic term drawn from a uniform probability density function with a range given by the standard deviation of the parameter values.

Some important processes have been neglected:

1. Influence from 0-group cod on recruitment.
2. Influence of larval drift on capelin recruitment.
3. Influence on recruitment of killing of 1-group capelin during the autumn fishery.
4. Misreporting of the catch of mature capelin by fishermen. Some capelin may have been discarded in connection with roe production and fishing for roe capelin for the consumption market.

Also, the stock-recruitment results are strongly dependent on the stock data for cod being correct. Several tuned VPA's for cod with different values of M should be run to check the sensitivity towards M.

It might be possible to use abundance estimates of the age 1 capelin for recruitment if the coverage of the survey area by Norway and Russia in the Barents Sea is improved in the future. However, we feel that at present, the time series of reliable data is somewhat short.

Fishing

The catch data used are numbers caught by age and month. In the period January to April all catch is assumed to occur on the mature capelin, and the catch each month is assumed to take place in the middle of the month. The autumn catch is converted to a fishing mortality, which is assumed to be the same for mature and immature capelin.

Summary of Estimations

The results of the various estimations are summarized in Table 2.

Runs into the Future

When the model is used for investigating management options, repeated runs into the future have to be made. Runs using different management strategies must be compared. Assumptions about the development of the cod stock must be made, and it is likely that the management strategy for capelin will depend on these assumptions. In this paper only one cod stock scenario is used, a natural mortality of 0.2, weight at age as in 1990 and a fishing mortality (F_{5-10}) of 0.3. The dependence of F on age and the maturity ogive are taken to be the same as calculated by the Working Group in 1990. The recruitment is stochastic and uniformly distributed between 200 and 600 million individuals at age 3. These assumptions lead to a buildup of the stock to a biomass of immature cod of about 1.3 million tonnes. An average recruitment of 400 million at age 3 is the same as used by the Working Group in the long-term prognosis made in 1991.

The year to year variation of growth, overlap and recruitment are assumed stochastic and modeled from the observed deviations from the model as used on historic data (see the summary in Table 2).

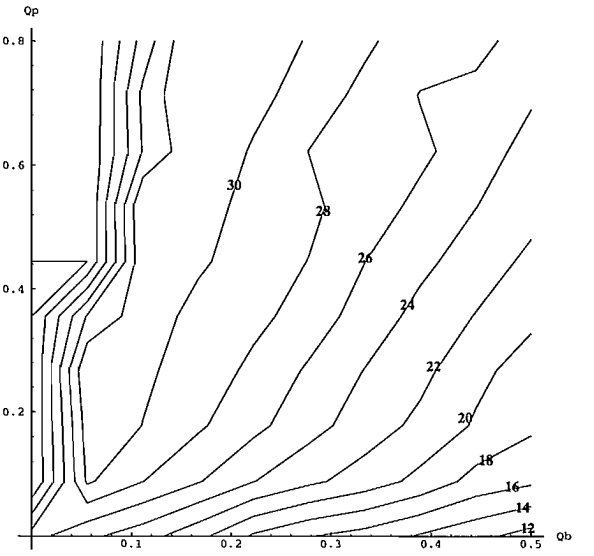


Fig. 5. Total catch over 50 years as a function of Q_p and Q_b for an experiment where 4 herring events occurred. The units for the isolines are millions of tonnes.

Management objectives

In this paper we limited ourselves to look at management options for regulating the catch of mature capelin in the winter only. In the future management of cod and capelin in the Barents Sea one might try to optimize the combined yield of capelin and cod. The present standing of the multispecies research does not allow for this, for instance a reliable relationship between growth of cod and capelin abundance does not yet exist. In this paper we address the problem in a straightforward and simplistic manner by trying to lay some of the theoretical foundation for a management that seeks to optimize the long-term yield of capelin, subject to the constraint that enough food for the cod stock is provided.

The uncertainties in the various processes modeled are rather large, and a way to deal with the uncertainties in a management context must be found. When the autumn acoustic cruise has been conducted, it is possible to calculate the probability density distribution of spawning stock biomass that corresponds to a given catch. It is possible that deviations on either side of the centre of this distribution will not have exactly the opposite effect, i.e., it may be that the negative long-term effect of a spawning stock of 50,000 tonnes (say) below the centre will be greater than the positive long-term positive effect on the stock of a spawning stock of 50,000 tonnes above the centre. To allow for such asymmetric effects, we propose the following decision rule: "The winter catch quota will give a probability of Q_p for the spawning stock to be smaller than Q_b million tonnes".

In order to estimate the effect of different values of Q_p and Q_b, the following series of simulations have been performed:

1. Values for Q_p and Q_b are set.

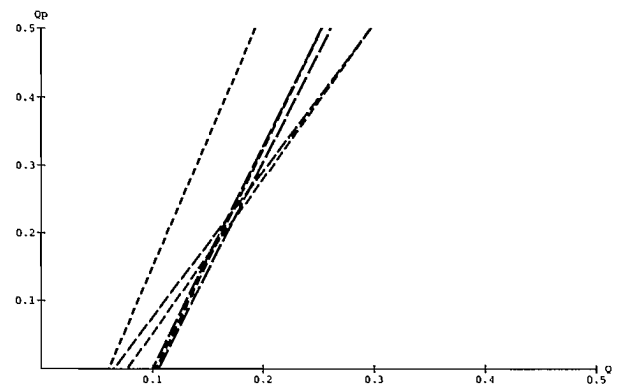


Fig. 6. Q_p — Q_b line for experiments with different sequence of herring events, other processes having the same sequence of events.

2. Starting the autumn 1990, the distribution of the spawning stock for various catch levels has been calculated, drawing all uncertain processes at random from the data summarized in Table 2. The catch corresponding to the values of Q_p and Q_b is calculated. We assume that this catch is known with certainty and this assumption has the effect of merely shifting the probability density distribution of the spawning stock towards lower spawning stocks)

3. Applying this catch, the stock has been calculated forward to the next autumn, drawing all uncertain processes at random from the data summarized in Table 2.

4. Step 2–3 are repeated starting with the simulated stock, until the year 2040.

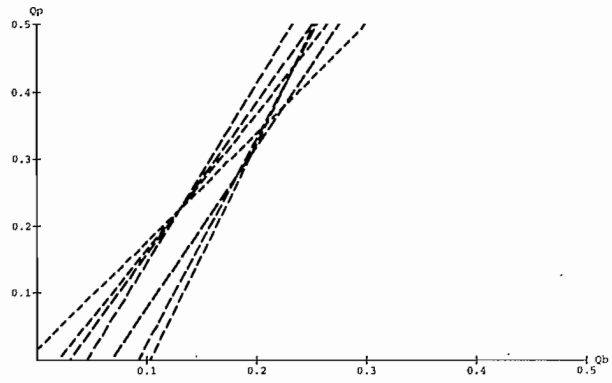


Fig. 7. Q_p — Q_b line for experiments with unchanged sequence of herring events, all other processes having different sequence of events.

Table 3. Total yield and average target spawning stock biomass of capelin for different herring event sequences.

Number of herring events	Total yield at $Q_p = 0.5$ million tonnes	Average Q_b at $Q_p = 0.5$
0	54.77	0.261
2	56.20	0.253
3	37.37	0.297
4	26.62	0.253
5	24.09	0.297
6	16.50	0.193

5. The whole procedure is repeated for various values of Q_p and Q_b.

The same sequence of random events have been used for all Q_p — Q_b combinations.

The total yield for each combination will be dependent on the sequence of random events. Especially, the number of herring events drawn will be critical for the total yield. Figure 5 shows how the yield varies with Q_p and Q_b for a sequence of events in which 4 herring events occurred during the 50 year period used.

It seems that the optimal value of Q_p as a function of Q_b can be approximated by a straight line. A detailed study of the output (not shown) shows that there is virtually no variation in the maximum total catch along this line (hereafter referred to as the Q_p — Q_b line). Hence, either Q_b or Q_p may be chosen freely and the other calculated by optimization.

To investigate the stability of the Q_p — Q_b line two groups of runs have been made. Figure 6 shows how the Q_p — Q_b line varies when different sequences of herring events are drawn, keeping all other processes at the same sequence of events. The lines corresponding to larger total yield have been displayed using a dashing with longer strokes.

Figure 7 shows how the Q_p — Q_b line varies when the herring events have the same sequence (4 events), varying all other processes.

Table 4. Total yield and average spawning stock biomass for simulation experiments with and without regard of capelin as source of food for cod (Four herring events).

	Without Capelin as food for cod		With Capelin as food for cod	
Stochastic Sequence No.	Total yield at $Q_p = 0.5$ 10^6 tonnes	Average Q_b at $Q_p = 0.5$	Total yield at $Q_p = 0.5$ 10^6 tonnes	Average Q_b at $Q_p = 0.5$
1	26.62	0.253	25.01	0.290
2	31.25	0.252	30.51	0.225
3	35.83	0.275	35.30	0.266
4	29.88	0.233	30.59	0.256
5	17.73	0.300	Simulations	
6	30.26	0.255	not performed	

The two sets of experiments exhibit no clear difference with regard to the slope or position of the lines. Also, in neither case does there seem to be any trend in slope or position of the lines with total yield over the period.

Table 3 shows a summary of the runs where the herring event sequence has been varied, the other processes being kept constant. The runs where the herring event sequence has been kept constant and the other processes being varied, are summarized in Table 4.

Capelin as Food for the Cod Stock

During the capelin stock decline, the growth of the cod decreased dramatically. Figure 8 shows the weight increment for various age groups of cod (calculated from the weight in the stock at January 1, as used by the Arctic Fisheries Working Group). The 1984 value refers to growth during 1984. In the same period, the CAPSEX model yields the ratio of consumption of capelin to immature cod biomass (q/b ratio) for immature cod and is displayed in Fig. 9.

The growth of cod decreased notably in the time period 1985–1988. In the same period the modeled q/b-ratio remained below 0.1. It then seems reasonable that this value might be a critical level used with management. It must be noted that we deal with the winter feeding by the cod only, and that in investigating the significance of the capelin stock for the growth of cod the summer/autumn feeding also should be taken into account. This feeding might be the reason why the growth starts to increase already in 1988 where we have a minimum in winter feeding. The growth of the immature capelin during 1988 might have contributed substantially to this growth.

In order to implement consideration for the cod feeding into the decision rule for capelin management, we have experimented with the same decision rule as before, allowing the constraint that only 10% of the q/b-ratio distribution to be below 0.1. This implies a trade-off between capelin catch and cod feeding that in actual management should be established by taking into account the relative values of cod and capelin.

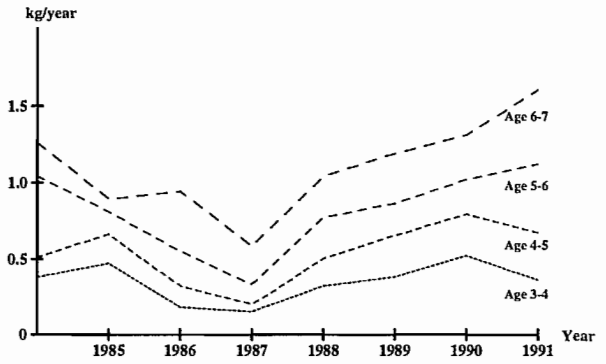


Fig. 8. Growth rate of North-east Arctic cod by age groups. Time axis denotes start of growth period.

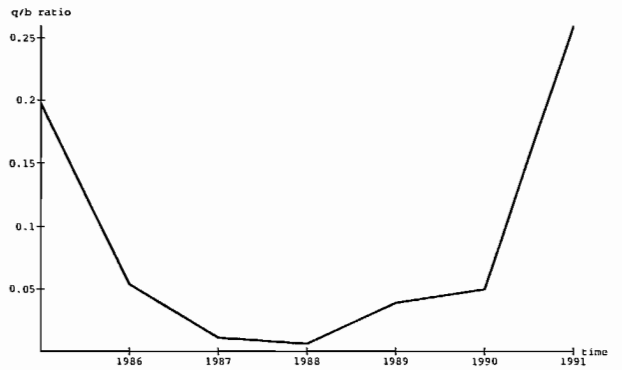


Fig. 9. q/b ratio for immature cod predating on mature caplin.

This trade-off should be based upon a proven relationship between cod feeding and cod growth. The present investigation just serves as an illustration.

The same series of random events as used for the runs underlying Fig. 5 are used. Figure 10 shows the result. Also in this case, the optimal value of Q_p as a function of Q_b can be approximated with a straight line, and also in this case there is little variation of the total yield along the line. Figure 11 shows this line for several experiments where the sequence of herring events has been kept constant and all other processes varied.

Table 4 shows a summary of the runs with and without taking into account capelin as source of food for cod. Experiments with the same sequence of events are shown on the same line.

The difference between the Q_b at $Q_p = 0.5$ for experiments with and without provision for cod feeding is by far smaller than the variation in Q_b at $Q_p = 0.5$ within each experiment. The main conclusion is that with the present model a management decision rule based on a maximum yield principle only also provides enough food for the cod stock. However, a cod stock scenario with a bigger predating cod stock might change

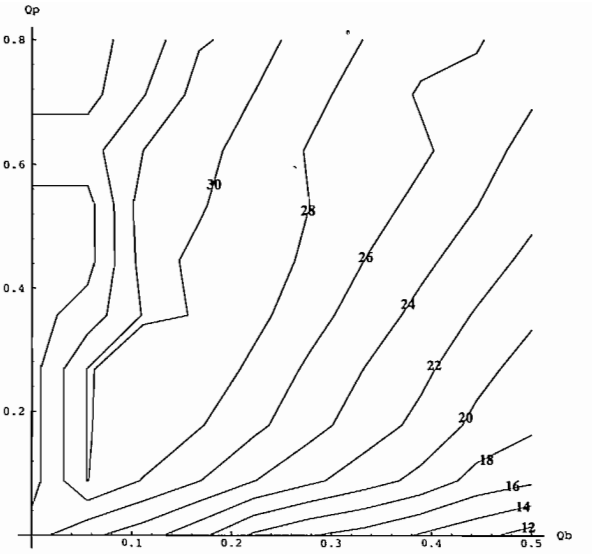


Fig. 10. Total catch over 50 years as a function of Q_p and Q_b for experiments where 4 herring events occurred. The units for the isolines are millions of tonnes.

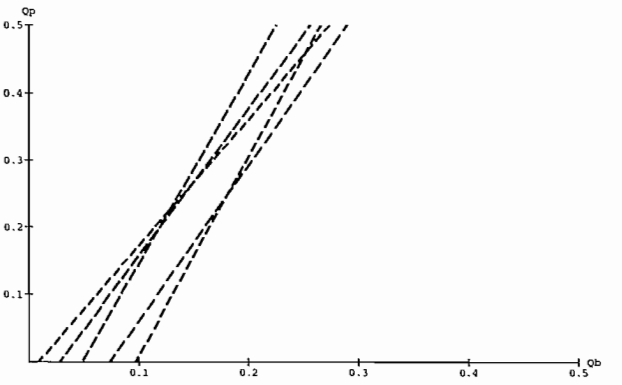


Fig. 11. Q_p — Q_b line for experiments with unchanged sequence of herring events, all other processes having different sequence of events.

this, as would a stricter criterion for enough food for the cod stock than the one we have used of keeping the percentage of sub-0.1 q/b-ratios lower than 10.

By comparing Tables 4 and 5 (left columns) we see that the expectation and variance of Q_b at $Q_p = 0.5$ does not seem to be very different in the two cases. This means that the uncertainty in herring recruitment and the combined effect of uncertainty in all other processes contribute roughly equally to the uncertainty in the management decision rule. However, the variance of the total yield is greater in the case of stochastic herring recruitment, all other processes at the same sequence of events than in the case of herring recruitment at the same sequence of events, all other processes being stochastic. The apparent lack of correlation between Q_b at $Q_p = 0.5$ and total yield in Table 3 indicates that a management decision rule that

also takes account of herring recruitment, which is observed at the time of the management decision, might not improve the precision very much.

Discussion

Prior to the collapse, the capelin stock was managed by an optimal spawning stock criterion, where the objective was to maximize the long-term yield. Some weaknesses of the procedure were:

1. The model was parameterized for a certain time period, in which growth and natural mortality was kept constant.
2. The natural mortality per month was kept the same for immature and mature capelin, although the overlap between capelin and the main predators is widely different for mature and immature capelin.
3. The significance of 0-group cod and 0-group and older herring for hampering capelin recruitment was not taken into account.
4. The natural mortality on mature capelin was not connected to the actual size of the cod stock, which is the main predator.

Even if these effects are known qualitatively, we are just in the beginning of quantification. Before we can do significantly better on the quantitative aspects, we have to rely on getting a feeling of the magnitude of the uncertainty involved in not knowing these effects very well. If we can use this uncertainty in management in such a way as to avoid situations where the probability of doing damage is great, then the lesson has been learned. We hope to have paved the way to dealing with a more complex management situation, where both the uncertainties and the effects from 0-group and older herring and 0-group cod must be taken into account.

The present paper presents a minimal model. Future improvements might be:

1. The analysis should be extended to comprise a combined winter-autumn fishery.
2. The natural mortality on immature capelin should be modeled, with one component due to the cod stock and one residual component. Also, it should be tested whether separate components for predation from harp seal and minke whales reduce the total variance significantly.
3. The growth should, if possible, be modeled by seeking relations between growth (estimated on a yearly basis), temperature and capelin abundance. In the CAPSEX software there is a provision for relating growth to the observed plankton abundance.
4. It should be explored whether the overlap variable could be related to temperature or trends in temperature.
5. The cod and capelin stock estimates are treated as indices in the single-species management of these species conducted so far. A future development of the method presented in this paper should include tests of how much we must scale the cod and capelin stocks before the quotas that would be suggested in an actual situation would be significantly affected.
6. The sensitivity towards uncertainties in the parameter estimates should be tested, including uncertainty in the parameters estimated in MULTSPEC.

Applications could be:

1. Giving catch quota advice on capelin. However, this must be done in close contact with the work in the Arctic Fisheries Working Group of ICES, which must provide development options for the cod stock.
2. Retrospective analysis:
   (a) What consequences did the 1986 catch on capelin have?
   (b) How could we have reacted in 1983 when the strong year class of herring became evident?

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Session II
Identifying and Quantifying Uncertainties

Assessing the Impact of Sampling Error on Model-Based Management Advice: Comparison of Equilibrium Yield per Recruit Variance Estimators *

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The assessment of fish stocks often relies on data estimated from complex sampling schemes. For instance, catch and weight-at-age estimates result from sampling commercial landings. This paper studies the propagation of sampling error in catches in an equilibrium yield per recruit model. The covariance matrix of catch estimators is calculated for a given design using sampling theory. The impact of this uncertainty on estimated yield per recruit is assessed by three techniques: the delta method, trials from a multinormal distribution of catches (Gaussian approximation), and bootstrap. The comparison of results leads to the following conclusions. (i) For the model studied, the delta method probably overestimates the variance of the response. (ii) Gaussian approximation and bootstrap give similar values. As the latter is free of approximation and of restrictive assumptions, this suggests that the yield model chosen is to some extent insensitive to the distributional form of catch estimators. (iii) Bootstrap is the method of choice, provided that resampling accurately mimics the whole complexity of the original sampling design. If resampling is not properly achieved, variances could be severely underestimated.

Les données utilisées pour l'évaluation des stocks de poissons sont souvent estimées à partir de procédures d'échantillonnage complexes. Ainsi, les estimations de captures et de poids aux âges proviennent de l'échantillonnage des captures commerciales. Cet article s'intéresse à la propagation de l'erreur d'échantillonnage des captures dans un modèle de rendement par recrue à l'équilibre. La matrice de covariance des estimateurs des captures est calculée pour un protocole donné par application de la théorie de l'échantillonnage. L'impact de cette incertitude sur l'estimation du rendement par recrue est évalué grâce à trois techniques: la delta-méthode, des tirages selon une distribution multinormale pour les captures (approximation Gaussienne) et le bootstrap. La comparaison des résultats amène aux conclusions suivantes: i) Dans le cas du modèle étudié, la delta-méthode sur-estime probablement la variance de la réponse. ii) L'approximation Gaussienne et le bootstrap donnent des valeurs semblables. Ce dernier ne requérant pas d'approximation ni d'hypothèse restrictive, cela suggère que le modèle de production est dans une certaine mesure insensible à la forme de la distribution des estimateurs des captures. iii) Le bootstrap est la méthode à choisir, pourvu que le rééchantillonnage reproduise précisément toute la complexité de la procédure d'échantillonnage; les variances pourraient sinon être gravement sous-estimées.

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Variance Estimates for Fisheries Assessment: Their Importance and How Best to Evaluate Them

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Management of many of the world's fisheries is based on estimates provided by quantitative fisheries assessment methods. The consequences of the uncertainties in this process are that these estimates usually have large variances, and may also be subject to bias. The precision (variances) of the estimates of a number of management-related quantities provided by some of the standard stock assessment methods for certain demersal fish species off southern Africa, are evaluated using the conditioned parametric bootstrap technique. The results of this evaluation demonstrate that many of the TAC estimates corresponding to standard harvesting strategies such as $F_{0.1}$ and $f_{0.1}$ are extremely poorly determined. In contrast, greater precision is evident for $F_{\text{status-quo}}$ harvesting strategy TAC's and for estimates of current replacement yield. For this reason, current management of a number of the fish species off southern Africa has had to be based on these latter harvesting strategies, despite the fact that they are suboptimal in the sense that they make no attempt to move the resource towards a more productive level. The conditioned parametric bootstrap technique is very intense computationally, whereas other variance-estimation procedures available are less so. Monte-Carlo simulation is used to investigate the performance of a number of these procedures for estimating the SE's and CV's of four management-related quantities for the Cape hake stock off northern Namibia. Results indicate that the unconditioned bootstrap approach should be avoided, and that the computationally rapid asymptotic method performs at least as well as the jackknife and some other bootstrap procedures in this case. Estimates of variance for management-related quantities estimated by observation-error production-model assessment procedures can be severely negatively biased in the presence of process error, even though the estimates of the quantities themselves may have small bias. Thus the risk associated with a particular harvesting strategy is likely to be underestimated if perceptions about uncertainty in resource assessment are based on such variance estimates.

La gestion de bon nombre de pêches dans le monde repose sur les estimations découlant de méthodes quantitatives d'évaluation des pêches. Les incertitudes en cause dans ce processus font en sorte que ces estimations varient souvent et peuvent être biaisées. La précision (variance) des estimations des valeurs reliées à la gestion, obtenues par quelques-unes des techniques courantes d'évaluation de stock pour certaines espèces de poissons démersaux d'Afrique australe, est évaluée à l'aide de la technique bootstrap paramétrique conditionnée. Cet exercice démontre que beaucoup d'estimations de TPA qui correspondent aux stratégies de capture connues telles que $F_{0.1}$ et $f_{0.1}$ sont extrêmement mal déterminées. Par opposition, une précision accrue est évidente pour les TPA de stratégie de capture $F_{\text{status-quo}}$ et pour les estimations de rendement de remplacement actuelles. Ceci explique pourquoi la gestion actuelle de plusieurs espèces de poissons sur les côtes d'Afrique australe a dû reposer sur ces dernières stratégies de capture, malgré que celles-ci soient sous-optimales du fait qu'elles ne tentent aucunement d'augmenter la productivité de la ressource. La technique de boostrap paramétrique conditionnée demande beaucoup de calcul alors que d'autres techniques d'estimation de variance en exigent moins. La simulation de Monte-Carlo est utilisée pour examiner le rendement de plusieurs de ces techniques pour l'estimation des ET et des VC de quatre valeurs reliées à la gestion dans le cas du stock de merlu du Cap, sur les côtes de la Namibie septentrionale. Les résultats révèlent que la méthode bootstrap non conditionnée devrait être évitée et que la méthode asymptotique à calcul rapide est au moins aussi efficace que la technique du canif et, dans ce cas, que certaines autres techniques bootstrap. Les estimations de variance pour les valeurs reliées à la gestion estimées par les techniques d'évaluation de modèle de production d'observation et d'erreur peuvent être considérablement biaisées par la présence d'erreurs de méthode, bien que les estimations même des quantités puissent ne comporter qu'un faible biais. Ainsi, le risque associé à chaque stratégie de capture risque fort d'être sous-estimé si les perceptions sur l'incertitude en évaluation de ressource reposent sur de telles estimations de variance.

The estimates of management-related quantities such as MSY's and TAC's obtained from quantitative fisheries assessment methods, such as production models (e.g., Schaefer 1954, 1957; Schnute 1977) and age-structured methods (e.g., Deriso et al. 1985; Pope and Shepherd 1985), are subject to error for a number of reasons. These include the

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Table 1. Estimates, with CPB estimated percentage CV's in parenthesis, of four management-related quantities for five fish stocks off southern Africa. B^* is the current exploitable biomass, RY* the current replacement yield, and $TAC_{0.1}$ the total allowable catch under an $f_{0.1}$ harvesting strategy. The estimates in this table were obtained by means of the production-model of Appendix A. Units are '000 t.

Stock/Quantity	MSY	B^*	RY*	$TAC_{0.1}$
Cape hake				
SA south coast[1]	52 (3)	146 (12)	52 (3)	47 (9)
SA west coast[1]	138 (2)	626 (8)	137 (3)	114 (8)
Southern Namibia[2]	165 (3)	644 (12)	164 (4)	141 (9)
Northern Namibia[2]	256 (3)	1576 (8)	253 (3)	256 (8)
Horse mackerel				
SA south coast[3]	38 (24)	206 (9)	37 (14)	40 (36)

Sources : [1] - Punt (1991b)
[2] - Punt (1989b)
[3] - Punt (1991c)

effects of model mis-specification, which arises because the true underlying dynamics of the resource are not well known, and also the poor quality or small quantity of the data. This paper examines why variance-estimation is important as well as how to choose a variance-estimator.

The extent of the biases associated with these assessment methods for a particular stock and management quantity can be estimated by Monte-Carlo simulation (see, for example, Punt 1988), and some adjustment for such a bias can be made. For example, Punt (op. cit.) assessed the estimates of the $f_{0.1}$ harvesting strategy TAC's provided by production models for the Cape hake stocks off South Africa to be positively biased. This was a contributory factor to the decision to manage these stocks on the basis of an $f_{0.2}$ rather than an $f_{0.1}$ harvesting strategy. At a more quantitative level, Hall et al. (1988) take explicit account of estimates of bias for stock-recruitment relationship parameters in evaluating the performances of alternative management procedures for Pacific herring.

However, even if point estimates are corrected for bias, the imprecision of such estimates means that the results provided by assessment methods may differ substantially from the corresponding true values. An overestimate of the appropriate TAC could lead to serious unintended depletion of a resource, particularly for stocks with a relatively high sustainable yield/biomass ratio. This consequence can be exacerbated by the time-series nature of many fisheries assessment methods, which usually leads to positive autocorrelation between TAC estimates for successive years.

In South Africa, the approach which has been followed to take account of imprecision is to obtain estimates of the variances of the TAC's associated with the various alternative harvesting strategies under consideration for each species. A single harvesting strategy is then selected, based on the trade-off between anticipated good performance for some management objectives and the achievement of reasonably precise TAC estimates. For example, the $f_{0.1}$ strategy [see Appendix A, Equation (A5)] attempts to drive the resource to a biomass level which is judged "optimal" in the sense of high utilization of the resource and enhanced profitability of the associated fishery in the longer term. In contrast, a strategy of setting TAC's equal to the current replacement yield is likely to be suboptimal in these respects.

This paper provides conditioned parametric bootstrap (CPB) coefficients of variation (CV's) for a number of management-related quantities obtained from production-model and VPA assessments of three demersal fish species off southern Africa [Cape hake (*Merluccius sp.*), Cape horse mackerel (*Trachurus trachurus capensis*), and sole (*Austroglossus pectoralis*)], and discusses their implications for the management of these resources. It then deals with the question of the selection of the method to be used to provide such estimates of precision. The CPB technique (Appendix B) is very intense computationally, whereas other variance-estimation procedures available are less so and would therefore be preferred unless they are less reliable. Monte-Carlo simulation is used to investigate the performance of five alternative variance-estimation procedures: the CPB, the conditioned nonparametric bootstrap (CNB), the unconditioned nonparametric bootstrap (UNB), the jackknife and an asymptotic method, for a particular case. These techniques are applied to estimate the standard errors (SE's) and coefficients of variation (CV's) of some management-related quantities for simulated data generated from a model of the Cape hake stock off northern Namibia, to address this question of reliability.

Estimation of Precision

Tables 1 and 2 contain estimates of four management-related quantities, together with their estimated CPB CV's, for three of South Africa's commercially most important demersal fish species (Cape hake, Cape horse mackerel and sole). Table 1 provides the estimates obtained by means of a production-model estimation approach, while Table 2 provides those from ad hoc tuned VPA. The production-model approach is documented in Appendix A; the Schaefer form of the surplus production function was used for the calculations reported. The ad hoc tuned VPA is described in Butterworth et al. (1990), and is applied to fleet-disaggregated data using Laurec-Shepherd tuning. The management-related quantities

Table 2. Estimates, with CPB estimated percentage CV's in parenthesis, of four management-related quantities for six fish stocks off southern Africa. The notation is as in Table 1, except that TAC_{sq} refers to a $F_{status\text{-}quo}$ TAC (maintaining fishing mortality at its current level), and $\widetilde{TAC}_{0.1}$ corresponds to a $F_{0.1}$ harvesting strategy. The estimates in this Table were obtained by means of an ad hoc tuned VPA. Units are '000 t.

Stock/Quantity	MSY	B^*	RY*	$\widetilde{TAC}_{0.1}$
Cape hake				
SA south coast[1]	–	131 (18)	52 (76)	36 (75)
SA west coast[1]	–	163 (10)	70 (13)	42 (16)
Southern Namibia[2]	164 (33)	433 (17)	219 (18)	131 (25)
Northern Namibia[2]	212 (54)	352 (31)	149 (36)	113 (48)
Sole				
SA east coast[3]	0.8 (8)	2.4 (15)	0.6 (15)	0.8 (23)
Horse mackerel				
Namibia[4]	621 (53)	4457 (46)	477 (57)	823 (60)

Sources : [1] - Punt (1991b)
[2] - Punt and Butterworth (1989)
[3] - Borchers (1991)
[4] - Butterworth et al. (1989)

estimated by the production-model assessments are the maximum sustainable yield (MSY), the current biomass (B^*), the current replacement yield (RY*), and the $f_{0.1}$ strategy TAC for the forthcoming year ($TAC_{0.1}$). Those provided by the VPA are MSY, B^*, and the $F_{status\text{-}quo}$ and $F_{0.1}$ strategy TAC's for the forthcoming year (TAC_{sq} and $\widetilde{TAC}_{0.1}$, respectively). The $f_{0.1}$ and $F_{0.1}$ harvesting strategies are defined in Appendix A (Equation A5) and at the end of Appendix C, respectively.

The estimates of replacement yield all have smaller CV's than do the estimates of $TAC_{0.1}$ (Table 1). Similarly, except for the South African south coast hake, the estimates of TAC_{sq} are more precisely determined than those of $\widetilde{TAC}_{0.1}$ (Table 2). This has led to management advice for two of these stocks (South African south coast horse mackerel and east coast sole) being based on the replacement yield or $F_{status\text{-}quo}$, rather than on the $f_{0.1}$ or $F_{0.1}$ harvesting strategies respectively, because of the considerable associated improvement in precision for the TAC estimates. A further consideration in the case of the sole stock was the lesser sensitivity of TAC_{sq} than $\widetilde{TAC}_{0.1}$ to the poorly known value for natural mortality M. These choices have been made even though neither of the selected strategies attempts to drive the resource to a target level which can be argued to be optimal in some sense (see also Butterworth et al. 1992).

For hake, the precision of the production model based $TAC_{0.1}$ (and $TAC_{0.2}$) estimates has been considered sufficiently high for management to be based on these estimates rather than those of replacement yield. The production model TAC estimates are preferred to those obtained from VPA for a number of reasons, which include the following. First, they are more precisely determined. Secondly, they do not depend upon the value assumed for M, whereas the VPA estimates can be very sensitive ($\approx$ 20–30% modification for a $0.1yr^{-1}$ change in M) to this assumption, for which there is little basis in data (Punt and Butterworth 1989). Thirdly, simulation studies (Punt 1991a) have suggested that the $F_{0.1}$ harvesting strategy coupled with a VPA estimator leads to unacceptably high inter-annual catch limit variability ($\approx$ 20–50% change in TAC from one year to the next on average). Finally, the VPA estimates for B^* and $\widetilde{TAC}_{0.1}$ in Table 2 are now thought likely to be negatively biased because the VPA estimator assumes constant rather than declining selectivity-at-age for older hake (Punt in press); there is no obvious independent basis for specifying the extent of this decline, to which these estimates are particularly sensitive.

The fact that the production model estimates of MSY, RY* and $TAC_{0.1}$ for hake are generally more precise than those of the current exploitable biomass B^* (see Table 1), indicates that there is substantial covariance between the estimates of some of the model parameters. This is the underlying reason why the TAC's estimated for some harvesting strategies are more precisely determined than those for other strategies. From the above, it is clear that variance estimation has been considered important in the choice of harvesting strategies for South Africa's marine resources. This is because a TAC which cannot be estimated with reasonable precision is, together with its associated harvesting strategy and estimation procedure, not regarded as a reliable basis for scientific recommendations for management. The presentation of point estimates alone, without associated estimates of precision, may serve to hide how little is really known, to the detriment of sound management advice. However, according this importance to variance estimation immediately raises two further questions: what is the best variance estimation method and how reliable are the estimates which it provides?

Comparison of Different Variance Estimation Procedures

The CPB procedure applied to estimate CV's in Tables 1 and 2 is very intensive computationally, because it involves

carrying out a large number of minimizations of an expression which is usually a non-linear function of its parameters. There are other variance-estimation procedures, such as the jackknife and asymptotic methods (Appendix B), which are much quicker computationally. It is therefore important to assess not only how reliable (accurate and precise) estimates from the CPB and related bootstrap methods are, but also whether any of the these other quicker approaches are sufficiently reliable to be used instead. To this end, the ability of various variance-estimation procedures to provide estimates of the SE's and the CV's of B^*, B^*/K, $TAC_{0.1}$ and MSY for the stock of Cape hake off northern Namibia is examined for the case where these estimates are obtained by means of the Butterworth-Andrew production model observation-error estimator (Appendix A).

The ratio B^*/K of the current exploitable biomass to its average pristine level is included in the group of management-related quantities considered, as estimates of this ratio allow an assessment of how far resource status is removed from the target under a particular harvesting strategy. For example, the $f_{0.1}$ harvesting strategy target level for B^*/K is 0.55 when the Schaefer model for the surplus production function is assumed.

Methods

In order to assess the performance of the five alternative variance-estimation procedures considered here, a Monte-Carlo simulation method has been used. This process is illustrated in Fig. 1, and involves the construction of an "operating model" (Linhart and Zucchini 1986) of the resource under consideration (see Appendix C). This model is then used to generate a large number of artificial data sets ($U = 1, 2, \ldots, U_{\max}$), where $U_{\max}$ was chosen to be 500 for the results reported in this paper. Application of the model-estimation procedure to each of these data sets yields a series of estimates for a management quantity Q of interest: $\widehat{Q}^1, \widehat{Q}^2, \ldots, \widehat{Q}^{U_{\max}}$. For the moment, we consider the case where Q is MSY, as illustrated in Fig. 1. The actual value of MSY (MSY^{True}) is known from the operating model. The bias and variance of the model-estimation procedure can then be computed from the $\widehat{Q}^U$'s:

$$\text{(1)} \qquad \text{Bias}\,\widehat{Q} = E_U\left(\widehat{Q}^U\right) - Q^{\text{True}}$$

$$\text{(2)} \qquad \text{Var}^{\text{True}}\widehat{Q} = E_U\left[\widehat{Q}^U - E_{U'}\left(\widehat{Q}^{U'}\right)\right]^2$$

where the expectations are obtained by averaging over U, i.e.,:

$$\text{(3)} \qquad E_U\left(\widehat{Q}^U\right) = \frac{1}{U_{\max}}\sum_{U=1}^{U_{\max}}\widehat{Q}^U$$

and the superscript "True" is indicated on the variance to distinguish this from the estimates of variance to be discussed below.

For ease of interpretation, the measures of estimator performance used in this paper will be expressed in relative terms, and are shown as percentages in the tables. Thus, relative bias, coefficient of variation, and relative efficiency are respectively defined as:

Table 3. Properties of the estimates of various management quantities provided by the Butterworth-Andrew production model observation-error estimator (Appendix A) with the Schaefer form for the surplus production function, for the "base case" choice of error variances in the operating model. The statistics reported are defined in Equations (4)–(6) and (D2)–(D5), and are all quoted as percentages.

Quantity	Relative Bias (β)	CV^{True} (σ')	Relative Efficiency	Relative bias for constant CV regression model (γ)
MSY	2.4	11.5	11.8	2.4
B^*/K	14.0	11.8	18.3	14.3
B^*	38.8	41.5	56.8	38.2
$TAC_{0.1}$	1.2	18.9	18.9	1.1

$$\text{(4)} \qquad \text{Rel Bias}\,\widehat{Q} = \text{Bias}\,\widehat{Q}\,Q^{\text{True}}$$

$$\text{(5)} \qquad \text{CV}^{\text{True}}\widehat{Q} = \sqrt{\text{Var}^{\text{True}}\widehat{Q}}\,/\,Q^{\text{True}}$$

$$\text{(6)} \qquad \text{Rel Effic}\,\widehat{Q} = \sqrt{\left(\text{Rel Bias}\,\widehat{Q}\right)^2 + \left(\text{CV}^{\text{True}}\widehat{Q}\right)^2}$$

For each artificial data set $\left\{C_y, (C/E)_y\right\}^U$, where C_y and $(C/E)_y$ represent catch and catch-per-unit effort, respectively, in year y, a variance-estimation procedure m can be applied to yield an estimate of the variance of $\widehat{Q}^U$: $\widehat{\text{Var}}_m Q^U$, where m indicates any one of the different procedures detailed in Appendix B. This variance is usually expressed either as a standard error (SE) or as a coefficient of variation (CV):

$$\text{(7)} \qquad \widehat{\text{SE}}_m Q^U = \sqrt{\widehat{\text{Var}}_m Q^U}$$

$$\text{(8)} \qquad \widehat{\text{CV}}_m Q^U = \widehat{\text{SE}}_m Q^U / \widehat{Q}^U$$

The measures of performance of the variance estimator are then computed by:

$$\text{(9)} \qquad \text{Rel Bias}\left[\widehat{\text{SE}}_m Q\right] = \frac{\left[E_U\left(\widehat{\text{SE}}_m Q^U\right) - \text{SE}^{\text{True}}\widehat{Q}\right]}{\text{SE}^{\text{True}}\widehat{Q}}$$

$$\text{(10)} \qquad \text{CV}\left[\widehat{\text{SE}}_m Q\right] = \frac{\sqrt{E_U\left[\widehat{\text{SE}}_m Q^U - E_{U'}\left(\widehat{\text{SE}}_m Q^{U'}\right)\right]^2}}{\text{SE}^{\text{True}}\widehat{Q}}$$

where $\text{SE}^{\text{True}}\widehat{Q} = \left(\text{Var}^{\text{True}}\widehat{Q}\right)^{0.5}$, and similarly for coefficients of variation:

$$\text{(11)} \qquad \text{Rel Bias}\left[\widehat{\text{CV}}_m Q\right] = \frac{\left[E_U\left(\widehat{\text{CV}}_m Q^U\right) - \text{CV}^{\text{True}}\widehat{Q}\right]}{\text{CV}^{\text{True}}\widehat{Q}}$$

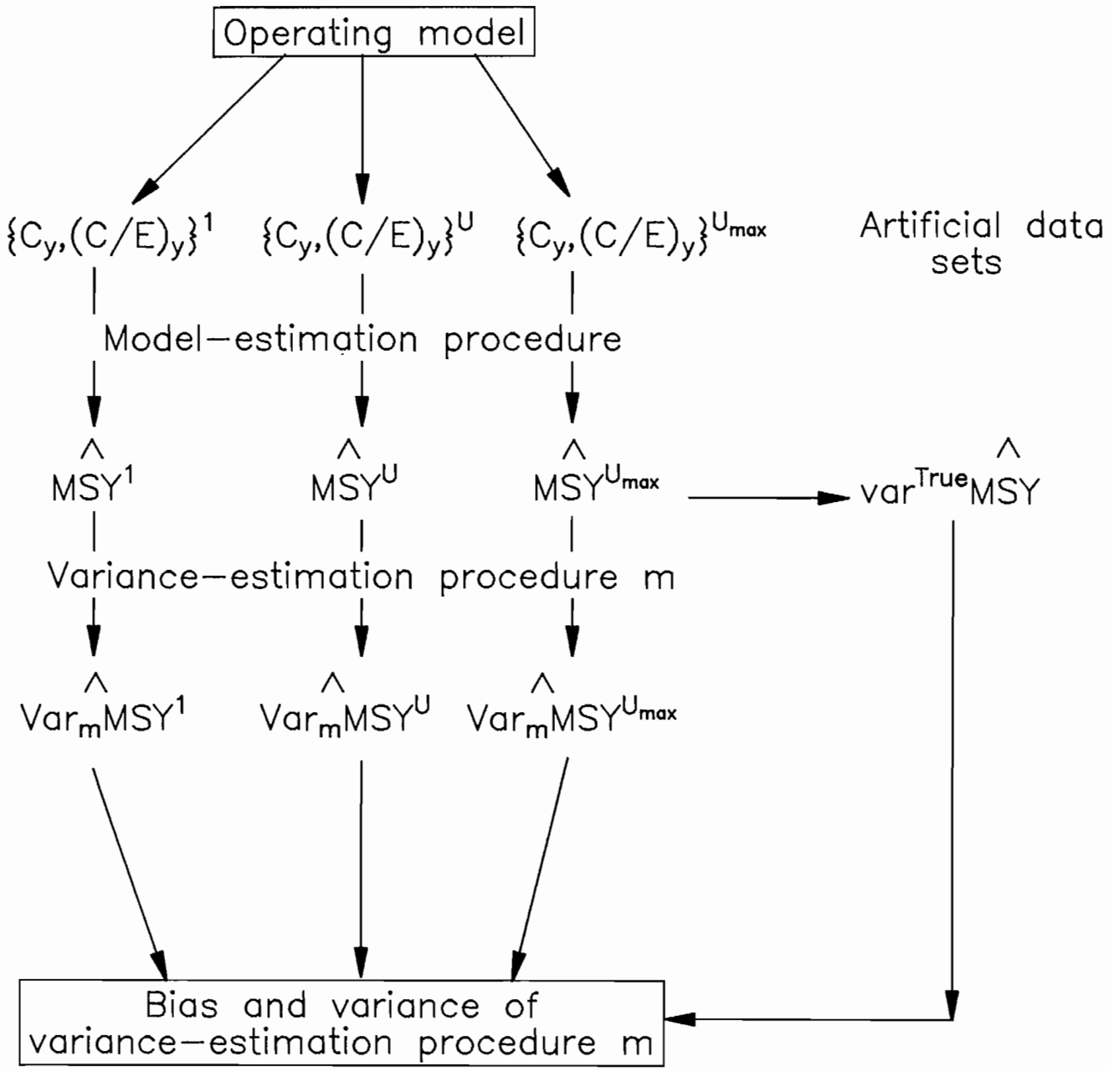


Fig. 1. Flowchart of the process used to calculate the bias and variance of the estimates provided by the various variance-estimation procedures (m) considered in this paper. The flowchart is shown for model-estimates of the management quantity MSY.

Table 4. Estimates of the relative bias, coefficient of variation, and relative efficiency (all expressed as percentages) of estimates of the SE of four management- related quantities for the "base case" choice of error variances in the operating model (i.e., $\sigma_s = 0.2$; $\sigma_r = 0.2$; $\sigma_q = 0.1$). The two CPB procedures differ in the number of replicates. In the acronyms for the bootstrap procedures, C/U indicates "conditioned" / "unconditioned" and P/N "parametric" / "nonparametric".

	Management Quantity					
	MSY			B^*		
Variance-Estimation Procedure	Relative Bias	CV	Relative Efficiency	Relative Bias	CV	Relative Efficiency
Bootstraps						
CPB[1]	−47.0	29.8	55.7	−49.0	46.3	67.4
CPB[2]	−46.4	29.5	55.0	−48.0	48.8	68.4
CNB	−46.7	30.4	55.7	−48.0	50.0	69.9
UNB	−8.1	59.4	59.9	> 1000	> 1000	> 1000
Jackknife	−42.2	33.3	53.8	−54.2	43.3	69.4
Asymptotic method	−49.0	25.0	55.0	−56.2	35.7	66.6

	Management Quantity					
	B^*/K			$TAC_{0.1}$		
Variance-Estimation Procedure	Relative Bias	CV	Relative Efficiency	Relative Bias	CV	Relative Efficiency
Bootstraps						
CPB[1]	−35.6	13.5	38.0	−41.0	22.1	46.5
CPB[2]	−35.2	12.8	37.5	−40.4	21.8	45.9
CNB	−35.4	13.6	37.9	−40.7	22.6	46.5
UNB	1.7	23.5	23.5	−0.8	43.9	43.9
Jackknife	−25.6	21.7	33.6	−32.4	28.7	43.3
Asymptotic method	−32.6	14.0	35.5	−40.0	20.1	44.8

[1] - 100 replicates
[2] - 400 replicates

$$(12)\,\mathrm{CV}\left[\widehat{\mathrm{CV}}_m Q\right] = \frac{\sqrt{E_U\left[\widehat{\mathrm{CV}}_m Q^U - E_{U'}\left(\widehat{\mathrm{CV}}_m Q^{U'}\right)\right]^2}}{\mathrm{CV}^{\mathrm{True}}\widehat{Q}}$$

Measures of relative efficiency for the various estimators m, for both SE and CV estimates, can then be obtained from the appropriate analog of Equation (6). Some technical problems of definition, related to the matter of "conditioning" of variance estimates, arise when implementing the equations above for some of the management-related quantities of interest. These are addressed in Appendix D.

Results

Values of relative bias, CV and relative efficiency for the estimates of the four management quantities of interest for the stock considered are listed in Table 3. Table 4 provides the values of the same three summary statistics of the distributions of standard error estimates provided by the five variance estimation procedures for these four quantities.

All these results are for the "base case" choice of error variances in the operating model, viz.: selectivity-at-age variability $\sigma_s = 0.2$; recruitment variability $\sigma_r = 0.2$; and fluctuations in catchability $\sigma_q = 0.1$ (see Appendix C for definitions and the rationale for the numerical choices). They are shown for only one form of the population dynamics model-estimator considered — the Schaefer form for the surplus production function. This is because results obtained for the Fox form show no marked qualitative differences. In order to assess the effects on performance of a particular choice for the number of bootstrap replicates, results are presented for two variants of the CPB procedure, viz.: 400 replicates as well as the 100 used for the other bootstrap procedures.

Figures 2 and 3 provide plots of some of the distributions of these estimates for SE. Fig. 2 shows results for the CPB approach with 100 replicates for B^*/K, B^* and $TAC_{0.1}$, while Fig. 3 compares the estimates for the SE of $TAC_{0.1}$ provided by the various variance-estimation procedures considered.

Table 5 provides the values of the summary statistics for the distributions of CV estimates. Table 6 provides results for two alternative choices of error variances in the operating model. The first ($\sigma_s = 0$; $\sigma_r = 0$; $\sigma_q = 0.15$) corresponds to observation error only, while the second ($\sigma_s = 0.2$; $\sigma_r = 0.3$;

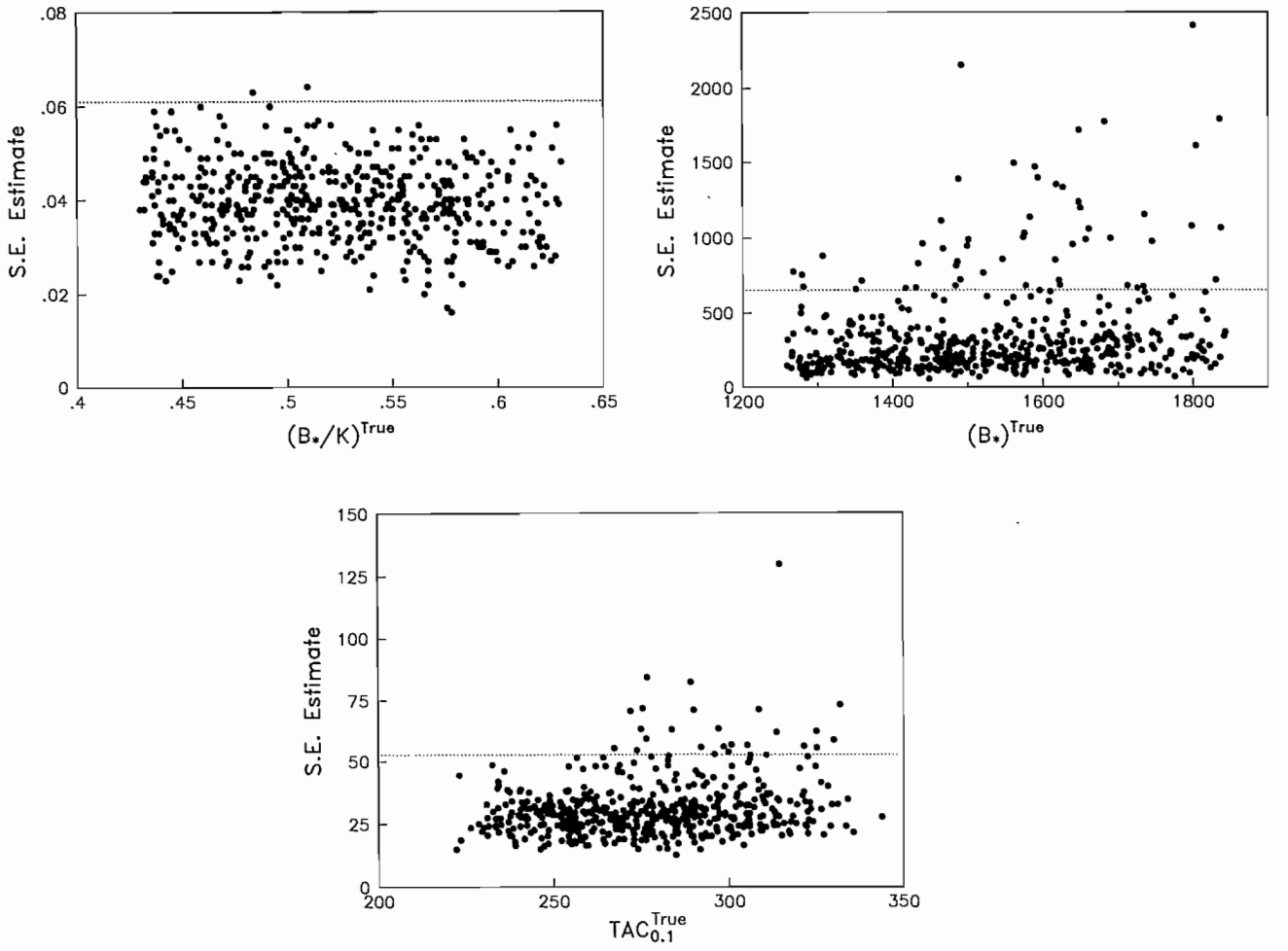


Fig. 2. Distributions of the estimates of standard error provided by the CPB procedure with 100 replicates for a) B^*/K; b) B^* and c) $TAC_{0.1}$. The dotted line in each plot is the true value of the standard error (SE^{True}). Units for b) and c) are '000 t.

$\sigma_q = 0.2$) increases the magnitude of two of the three sources of error.

Discussion

The "Base Case" Operating Model

For the "base case" choice of error variances in the operating model ($\sigma_s = 0.2$; $\sigma_r = 0.2$; $\sigma_q = 0.1$), all of the variance-estimation procedures provide estimates of SE which are negatively biased (Table 4; Figures 2 and 3). Except in the case of the UNB procedure, these negative biases are all substantial (26–56%). This means that if the SE's provided by these variance-estimation procedures are used when performing a risk analysis for some future harvesting option, risk would usually be considerably underestimated. The sizes of these biases are different for the four different management-related quantities, being generally rather larger for B^*, and smaller for B^*/K.

Although the UNB procedure provides SE estimates with much less bias than the other approaches for all four management-related quantities considered (Table 4), these estimates are far more variable [see Fig. 3(d)]. In consequence, the relative efficiencies achieved by the UNB procedure are of similar or larger magnitude to those for the other procedures. This procedure is completely unreliable for the management quantity B^* (Table 4). The poor performance of this approach is not surprising, as it was not designed to handle regression/time-series-type situations. One reason why the approach performs poorly is that in some of the replicates, influential points near the ends of the time-series are omitted. Given this particular problem, there is no reason to suspect that the performance of the approach would be improved by increasing the number of bootstrap replicates. Overall, these results indicate that the UNB procedure should be avoided for regression or time series models.

In general, because lower bias tends to accompany higher variance and vice versa, there is seldom much to choose between the variance-estimation procedures in terms of the relative efficiency measure, although a notable exception to this is the substantially inferior performance of the UNB procedure for B^* and B^*/K. The jackknife (to a limited extent) and

Table 5. Estimates of the relative bias, coefficient of variation, and relative efficiency (all expressed as percentages) of estimates of the CV of four management-related quantities for the "base case" choice of error variances in the operating model (i.e., $\sigma_s = 0.2$; $\sigma_r = 0.2$; $\sigma_q = 0.1$). Notation is as in Table 4.

	Management Quantity					
	MSY			B^*		
Variance-Estimation Procedure	Relative Bias	CV	Relative Efficiency	Relative Bias	CV	Relative Efficiency
Bootstraps						
CPB	−43.8	69.9	82.5	−66.2	16.5	68.2
CNB	−44.0	58.6	73.2	−66.2	16.4	68.2
UNB	−1.0	156.9	156.9	> 1000	> 1000	> 1000
Jackknife	−41.7	38.5	56.7	−69.5	15.5	71.2
Asymptotic method	−42.6	145.3	151.4	−70.5	11.9	71.5

	Management Quantity					
	B^*/K			$TAC_{0.1}$		
Variance-Estimation Procedure	Relative Bias	CV	Relative Efficiency	Relative Bias	CV	Relative Efficiency
Bootstraps						
CPB	−42.9	13.7	45.0	−37.5	45.1	58.7
CNB	−42.7	13.8	44.9	−37.5	38.7	53.9
UNB	−9.9	23.1	25.1	7.2	127.8	128.0
Jackknife	−34.3	20.0	39.7	−30.7	31.1	43.7
Asymptotic method	−40.4	13.4	42.6	−34.9	87.4	94.1

UNB approaches tend to be less biased but more variable than the other variance-estimation procedures. In terms of the relative efficiency measure, the asymptotic method marginally out performs the CPB method, with improvements in relative efficiency of between 1 and 7%. This suggests that this method is quite adequate for variance-estimation purposes for situations similar to that considered here.

The performances of the two 100-replicates conditioned bootstrap procedures (CPB and CNB) are very similar; differences in relative efficiency of less than 3%. However, the calculations of this paper have limited ability to discriminate between these two methods, as the same distribution (log-normal) used to generate errors in the operating model was assumed for the parametric bootstrap procedures. With regard to the choice of the number of replicates, it is possible to improve the performance of the CPB method, in terms of the relative efficiency measure, by increasing this number from 100 to 400, the extent of the improvement is minimal (1–2% at most). This suggests that 100 replicates are probably adequate for variance estimation.

CV Estimation

The estimates of the CV's of MSY and $TAC_{0.1}$ (Table 5) are slightly (3%) less biased than the estimates of their SE's (Table 4). However, the estimates of the CV's of B^* and B^*/K are notably more biased ($\approx$ 12%). The reason for this may be related to the relatively large biases of the model-estimates of B^* and B^*/K (see Table 3). Except for B^*, the CV's of the estimated coefficients of variation in Table 5 are larger than the CV's of the estimated standard errors in Table 4. This results in the relative efficiencies of the estimates of the CV's of MSY, B^*/K and $TAC_{0.1}$ being approximately 15% larger than the corresponding values in Table 4.

Overall then, estimation of SE's was achieved slightly more successfully than that of CV's.

Other Choices for Error Variances in the Operating Model

The estimated SE's are generally much closer to their true values when there is no process error in the operating model, so that the only noise is that in the fishing effort-fishing mortality relationship (see Table 6). In this case, biases are seldom larger than 10%, except for the UNB procedure. This is not an unexpected result, because in this case the assumption of pure observation error made by the model-estimator corresponds to the actual situation. The CV's, apart from those for the UNB method, hardly differ from those for the "base case", so that enhanced relative efficiencies are a reflection purely of the lesser bias.

Increasing the CV's of the variability associated with catchability (σ_q) to 0.2, and with recruitment (σ_r) to 0.3, leads to greater variability in the estimates of SE for MSY, B^* and

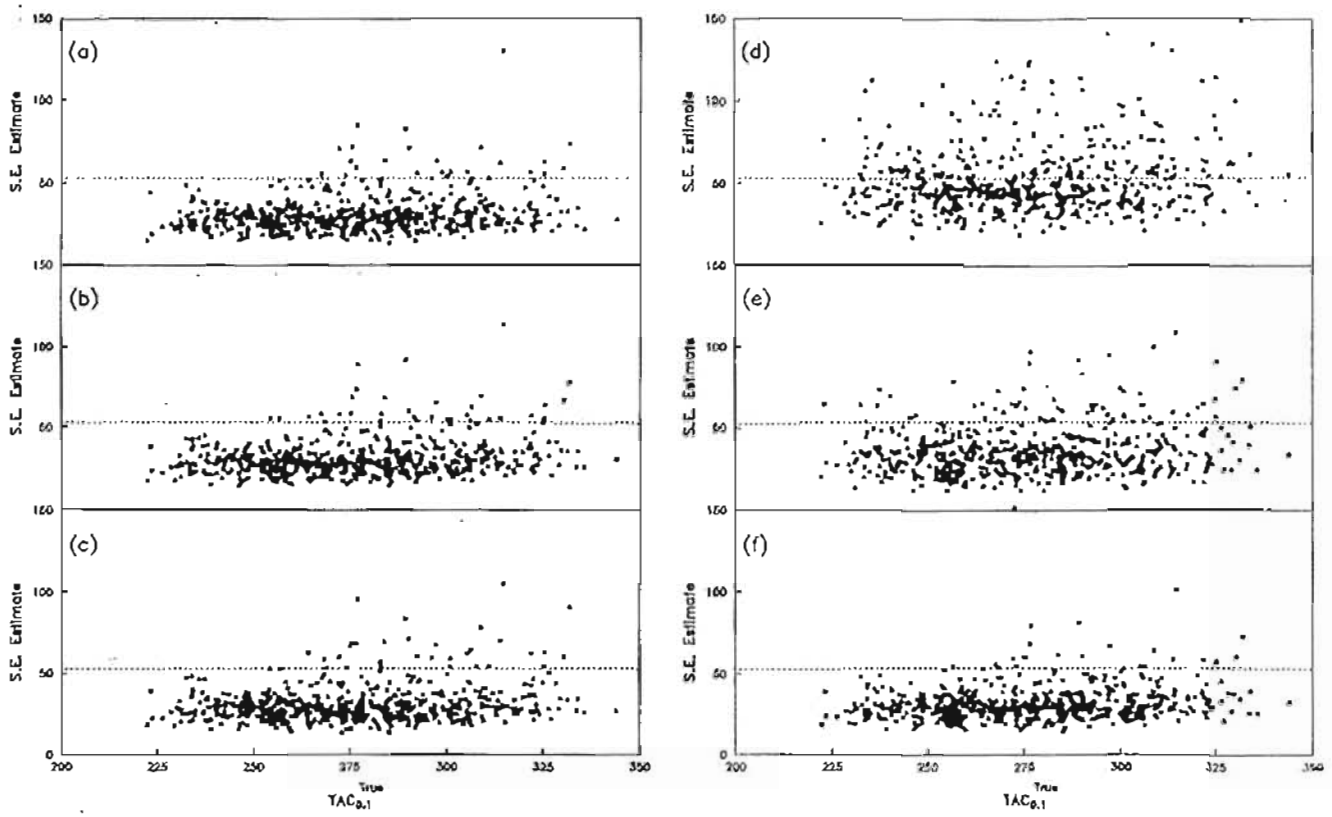


Fig. 3. Distributions of the estimates of the standard error of $TAC_{0.1}$ provided by the various variance-estimation procedures considered: a) CPB (100 replicates); b) CNB (400 replicates); c) CNB; d) UNB; e) jackknife and f) asymptotic method. The dotted line is the true value of the standard error (SE^{True}). Units are '000t.

$TAC_{0.1}$, although the CV's of the SE's of B^*/K remain very similar to their "base case" values. The relative efficiency of SE's for B^*, B^*/K and $TAC_{0.1}$ provided by the jackknife and asymptotic methods deteriorate by 25% compared to their "base case" values. However, the performance of the bootstrap methods shows much greater degradation. Thus, in circumstances where variability is considerable, it would seem that the asymptotic and jackknife methods are to be preferred.

The results of Table 6 indicate that obtaining unbiased estimates of SE's or CV's — as might be desirable for risk analysis, for example — is problematic. This is because the extent of any bias, and hence the magnitude of any bias-correction factor, depends not only on the biology, exploitation pattern and catch history of the resource, but also on the form and extent of process and observation error. While obtaining estimates for the former set of factors is usually feasible, reliable quantification of the error magnitudes can be much more difficult.

Does Insensitivity to Method Indicate Reliability?

It is clear from the above that the estimates of variance provided by all of the approaches considered have limited reliability — they are themselves quite variable and can be subject to substantial negative bias. In a real situation, a common approach to assessing reliability is to check the extent to which the estimates of interest are sensitive to the estimation method used. Table 7 lists the correlations between the estimates of SE provided by the procedures, and shows that these estimates are highly positively correlated ($\rho > 0.7$). [This is not so for the UNB procedure, but this is in any case seen as the least reliable method.] This means that, in a real situation, the similarity of variance estimates provided by a number of different methods is not a necessary indication of a greater reliability of such estimates.

Conclusions

The most important of the results above are the following.

1. The unconditioned nonparametric bootstrap (UNB) procedure performs very poorly in some circumstances, and therefore should be avoided.
2. There is little to choose between the other variance-estimation procedures for "intermediate" levels of error in the operating model. However, as error variance is increased, the performances of the asymptotic and jackknife methods do not degrade as much as those of the bootstrap procedures.
3. There is little to gain in respect of variance estimation from carrying out a very large number of Monte-Carlo replicates when applying bootstrap methods — 100 replicates are probably sufficient.

Table 6. Estimates of the relative bias, coefficient of variation, and relative efficiency (all expressed as percentages) of estimates of the SE of four management-related quantities for two alternative choices of error variances in the operating model. Notation is as in Table 5.

a) MSY

	Operating Model					
	observation error only $\sigma_s = 0; \sigma_r = 0; \sigma_a = 0.15$			Larger variances $\sigma_s = 0.2; \sigma_r = 0.3; \sigma_a = 0.2$		
Variance-Estimation Procedure	Relative Bias	CV	Relative Efficiency	Relative Bias	CV	Relative Efficiency
Bootstraps						
CPB	−4.6	27.9	28.3	−31.1	284.8	286.5
CNB	−4.9	27.3	27.7	−36.5	191.8	195.3
UNB	61.7	64.5	89.3	20.5	475.4	475.9
Jackknife	0.7	33.3	33.3	−39.8	71.4	81.7
Asymptotic method	−8.2	22.7	24.2	−41.6	81.9	91.9

b) B^*

	Operating Model					
	observation error only $\sigma_s = 0; \sigma_r = 0; \sigma_a = 0.15$			Larger variances $\sigma_s = 0.2; \sigma_r = 0.3; \sigma_a = 0.2$		
Variance-Estimation Procedure	Relative Bias	CV	Relative Efficiency	Relative Bias	CV	Relative Efficiency
Bootstraps						
CPB	8.6	50.3	51.0	> 1000	> 1000	> 1000
CNB	7.7	45.6	46.2	> 1000	> 1000	> 1000
UNB	88.6	308.8	321.2	> 1000	> 1000	> 1000
Jackknife	−4.0	42.1	42.3	−50.3	100.2	112.1
Asymptotic method	−6.8	32.7	33.4	−56.7	70.5	90.5

c) B^*/K

	Operating Model					
	observation error only $\sigma_s = 0; \sigma_r = 0; \sigma_a = 0.15$			Larger variances $\sigma_s = 0.2; \sigma_r = 0.3; \sigma_a = 0.2$		
Variance-Estimation Procedure	Relative Bias	CV	Relative Efficiency	Relative Bias	CV	Relative Efficiency
Bootstraps						
CPB	−14.1	15.4	20.9	−33.9	17.0	37.9
CNB	−14.2	15.0	20.6	−33.8	16.9	37.8
UNB	33.5	26.1	42.5	1.5	24.7	24.7
Jackknife	−2.8	24.4	24.6	−19.2	28.6	34.4
Asymptotic method	−10.6	15.4	18.7	−29.3	20.5	35.8

d) $TAC_{0.1}$

	Operating Model					
	observation error only $\sigma_s = 0; \sigma_r = 0; \sigma_a = 0.15$			Larger variances $\sigma_s = 0.2; \sigma_r = 0.3; \sigma_a = 0.2$		
Variance-Estimation Procedure	Relative Bias	CV	Relative Efficiency	Relative Bias	CV	Relative Efficiency
Bootstraps						
CPB	−11.1	20.0	22.9	−26.4	284.9	286.1
CNB	−11.4	19.5	22.5	−32.5	181.4	184.3
UNB	45.3	40.8	61.0	25.5	485.6	486.3
Jackknife	−0.8	29.0	29.0	−30.2	67.3	73.8
Asymptotic method	−9.7	18.7	21.0	−35.4	67.4	76.1

Table 7. Correlation coefficients between estimates of SE's provided by different variance-estimation procedures.

a) MSY

	CPB	CNB	Jackknife	Asymptotic method	UNB
CPB	1.00				
CNB	0.97	1.00			
Jackknife	0.91	0.91	1.00		
Asymptotic method	0.96	0.96	0.91	1.00	
UNB	0.93	0.92	0.90	0.94	1.00

b) B^*

	CPB	CNB	Jackknife	Asymptotic method	UNB
CPB	1.00				
CNB	0.93	1.00			
Jackknife	0.92	0.91	1.00		
Asymptotic method	0.96	0.96	0.95	1.00	
UNB	0.30	0.42	0.31	0.36	1.00

c) B^*/K

	CPB	CNB	Jackknife	Asymptotic method	UNB
CPB	1.00				
CNB	0.89	1.00			
Jackknife	0.73	0.73	1.00		
Asymptotic method	0.92	0.91	0.83	1.00	
UNB	0.80	0.79	0.85	0.89	1.00

d) $TAC_{0.1}$

	CPB	CNB	Jackknife	Asymptotic method	UNB
CPB	1.00				
CNB	0.95	1.00			
Jackknife	0.85	0.85	1.00		
Asymptotic method	0.96	0.95	0.89	1.00	
UNB	0.91	0.90	0.87	0.94	1.00

4. The presence of process error in a situation assessed by means of an observation-error estimator, can lead to substantial negative bias in estimates of variance.

Key conclusions are therefore as follows.

- Although observation-error estimators generally seem to provide the more reliable estimates of management-related quantities in production model assessments (Ludwig et al. 1988; Punt 1989a), the associated estimates of variance can be severely negatively biased — an aspect that needs to be considered if such variance estimates are to provide the basis for a risk analysis of various harvesting options.
- The asymptotic method of variance estimation performs at least as well as the computationally more intensive bootstrap procedures in the case considered.

A potential advantage of the bootstrap over asymptotic methods lies in the provision of confidence limit estimates. An approach based on likelihood ratios, which avoids the approximation of the delta method, can be used to provide these if the asymptotic method is used; however, the calculations have to be repeated separately for each quantity of interest, whereas the bootstrap procedure provides estimates for all quantities simultaneously. Reliable evaluations of the performance of alternative confidence interval estimators would require an exercise even more computationally intensive than has been attempted in this paper.

As with all numerical simulation studies conditioned on a specific situation — the northern Namibian hake fishery in this instance — the question of the generality of the results arises. We suspect that the first of the two key conclusions would apply generally, though the second may be a consequence of the fairly long time-series of CPUE data available for this fishery and the contrast which it shows (a marked decrease in catch rate followed by a partial recovery). There is such a variety of factors which may differ from case to case,

and upon which the reliability of variance estimators could depend, that it seems best to carry out specific simulations conditioned upon the fishery of interest, if a secure quantitative estimate of the bias of an variance estimator is required for a particular situation.

Acknowledgements

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Appendix A

The Butterworth-Andrew Observation-Error Estimator and Associated Harvesting Strategy (Butterworth and Andrew 1984; Punt 1988, 1991a)

The fishery is modelled as follows:

$$B_{y+1} = B_y + g(B_y) - C_y \quad \text{(A1)}$$

$$(C/E)_y = q\frac{B_y + B_{y+1}}{2}e^{\lambda_y} \quad \text{(A2)}$$

where,

B_y is the biomass at the start of year y and $B_1 = K$,

$g(B)$ is surplus production as a function of biomass, for example, either:

$g(B) = rB(1 - B/K)$ - the Schaefer form

$g(B) = rB(1 - \ln B/\ln K)$ - the Fox form

r is the intrinsic growth rate parameter,

K is the average biomass level prior to exploitation,

q is the catchability coefficient,

$(C/E)_y$ is the CPUE for year y,

C_y is the actual catch during year y, and

λ_y is the observation error in year y.

Note that Equation (A1) is deterministic (zero process error). Error is assumed to occur only in Equation (A2), i.e., this is a pure observation-error estimator. For the purposes of this paper, CPUE data are assumed to be available from year $y = 1$ (when exploitation commenced) to the current year $y = n$.

In order to obtain estimates of the parameters of this model, the likelihood function (L) is maximized. Ignoring constants, this corresponds to minimizing:

$$-\ln L = n\ln\sigma + \frac{1}{2\sigma^2}\sum_{i=1}^{n}\left(\ln(C/E)_y - \ln(\widehat{C/E})_y\right)^2 \quad \text{(A3)}$$

where σ is the standard deviation of the residuals, and is estimated by:

$$\widehat{\sigma}^2 = \frac{1}{n}\sum_{i=1}^{n}\left(\ln(C/E)_y - \ln(\widehat{C/E})_y\right)^2 \quad \text{(A4)}$$

$$(\widehat{C/E})_y = \widehat{q}\left(\widehat{B}_y + \widehat{B}_{y+1}\right)/2$$

The current replacement yield RY* is estimated by the surplus production during year $n + 1$, i.e., $g(B_n + 1)$. In order to obtain TAC's corresponding to the $f_{0.1}$ harvesting strategy, i.e., the harvesting strategy which involves fixing future fishing effort at that level at which the slope of the equilibrium catch versus effort curve is 1/10 that at the origin ($E_{0.1}$), for the forthcoming year $y = n + 1$, the formula:

$$\widehat{\text{TAC}}_{0.1}(n+1) = (\widehat{C/E})_{n+1}\widehat{E}_{0.1} \quad \text{(A5)}$$

is applied. Under the assumption that the $f_{0.1}$ strategy TAC estimated for year $n + 1$ will in fact be taken, an estimate of $(C/E)_{n+1}$ can be obtained from the formula:

$$(\widehat{C/E})_{n+1} = \frac{2\widehat{B}_{n+1} + g\left(\widehat{B}_{n+1}\right)}{2/\widehat{q} + \widehat{E}_{0.1}} \quad \text{(A6)}$$

Appendix B

Variance-Estimation Procedures

In order to illustrate how the variance-estimation procedures are used in practice, a description of the function which is minimized to estimate the model parameters, is given for each procedure. In general, this function may be written:

$$SS = \sum_{y=1}^{n} W_y^V\left[\ln(C/E)_y^V - \ln(\widehat{C/E})_y^V\right]^2 \quad \text{(B1)}$$

where,

$(C/E)_y^V$ is the artificially generated CPUE for year y and data set V,

$(\widehat{C/E})_y^V$ is the model-predicted CPUE for year y and data set V,

n is the number of data points, and

W_y^V are a series of weights ($y = 1, 2, \ldots, n$) constrained so that $\sum_y^n W_y^V = n$.

For estimation from the original data, the superscript V in equation (B1) is no longer necessary and each $W_y = 1$.

1. Bootstrap-Based Procedures

The bootstrap (Efron 1981, 1982, 1985; Punt 1988) estimates the standard error of $\widehat{Q}$ as follows: a large number ($V_{\max}$, where $V = 1, \ldots, V_{\max}$) of random bootstrap samples $\{X^V : V = 1, \ldots, V_{\max}\}$, where $X^V = \{W_y^V, (C/E)_y^V : y = 1, 2, \ldots, n\}$ are generated, and the set of estimates of Q for each of these: $\{\widehat{Q}^1, \widehat{Q}^2, \ldots, \widehat{Q}^{V_{\max}}\}$ is computed. Three forms of the bootstrap are considered here: the conditioned parametric (CPB), conditioned nonparametric (CNB), and unconditioned nonparametric (UNB) bootstraps. The differences between them lie in the approaches used to generate the replicate bootstrap data sets. A method will be said to be "conditioned" if it takes the regression nature of the estimation into account in some way, while it is "parametric" if it makes explicit use of the form of the likelihood

function (equation A3) — i.e., assumes a specific parametric form for the distribution function for the residuals. In all of the approaches considered here, the catch series is fixed, i.e., $C_y^V = C_y$.

The variance of $\widehat{Q}$ is then estimated by:

$$\text{(B2)} \qquad \widehat{\text{Var}(Q)} = \frac{1}{V_{\max}} \sum_{V=1}^{V_{\max}} \left[\widehat{Q}^V - Q_{(\cdot)}\right]^2$$

where $Q_{(\cdot)}$ is the mean of the $\widehat{Q}^V$'s.

a) The Conditioned Parametric Bootstrap (CPB)

This approach generates random effort series using the predicted CPUE estimates obtained by fitting the model to the actual data, and the assumption that the noise is pure observation error, i.e.:

$$\text{(B3)} \qquad (C/E)_y^V = (\widehat{C/E})_y \exp\left(\nu_y^V\right),$$

where,

$(\widehat{C/E})_y$ is the estimate of CPUE in year y obtained by fitting the model to the actual data,

ν_y^V is random variate from a normal distribution with zero mean and variance σ^2.

σ^2 is estimated from the residuals of the model fit to the actual data:

$$\text{(B4)} \qquad \widehat{\sigma}^2 = \frac{1}{n} \sum_{y=1}^{n} \lambda_y^2$$

$\lambda_y = \ln(C/E)_y - \ln\left(\widehat{C/E}\right)_y$, and n is the number of data points.

For this procedure, $W_y^V = 1$, i.e., ("artificial") data points are included for every year in the original CPUE series, and each is given an equal weight.

b) The Conditioned Nonparametric Bootstrap (CNB)

For the bootstrap samples, ν_y^V is selected at random with replacement from the set $\{\lambda_y : y = 1, \ldots, n\}$. $(C/E)_y^V$ is given, as above, by

$$\text{(B5)} \qquad (C/E)_y^V = (\widehat{C/E})_y \exp\left(\nu_y^V\right)$$

c) The Unconditioned Nonparametric Bootstrap (UNB)

This procedure involves defining the vector W^V by equating each element W_y^V to the number of times the value y is selected during sampling n times with replacement from the set $\{1, 2, \ldots, n\}$. Thus for each V:

$$\text{(B6)} \qquad \sum_{y=1}^{n} W_y^V = n$$

where W_y^V is zero or a positive integer less than or equal to n (in reality $n - 2$, otherwise estimation of the three parameters of the population dynamics model is not possible, but the probability of an instance of less than three non-zero W_y^V values for the case $n = 23$ considered here is negligible). This approach is described as "unconditioned" as a bootstrap realization need not include every year in the original CPUE data series because of zero weights. For this method, $(C/E)_y^V = (C/E)_y$.

2. The Jackknife

The jackknife estimate of the variance of $\widehat{Q}$ (Efron 1982) is:

$$\text{(B7)} \qquad \widehat{\text{Var}(Q)} = \frac{n-1}{n} \sum_{y=1}^{n} \left[\widehat{Q}_{-y} - Q_{(\cdot)}\right]^2$$

where,

$\widehat{Q}_{-y}$ is the estimate of Q computed by omitting the data point for year y, and assigning a weight $W_y^V = n/(n-1)$ to the remaining observations, and

$Q_{(\cdot)}$ is the mean of the $\widehat{Q}_{-y}$'s.

3. The Asymptotic Method

The asymptotic variance-covariance matrix for the parameters of a model is the inverse of the information matrix (Draper and Smith 1966; Dobson 1991), which is:

$$\text{(B8)} \qquad I_{ij} = \frac{\partial^2 \ln L}{\partial p_i \partial p_j}$$

where $\boldsymbol{p} = (p_1, p_2, \ldots, p_m)$ is the vector of model parameters, and L is the likelihood function (see equation A3).

The variance of a quantity which is a function of these parameters: $Q = Q(\boldsymbol{p})$ can be approximated by the delta method (Seber 1982; Ratkowsky 1983):

$$\text{(B9)} \qquad \widehat{\text{VAR}(Q)} = \sum_{i=1}^{m} \sum_{j=1}^{m} \frac{\partial Q}{\partial p_i} \frac{\partial Q}{\partial p_j} \text{COV}\left(p_i, p_j\right)$$

Note that for the Butterworth-Andrew model of Appendix A, there are four (rather than three) model parameters because σ is a model parameter in addition to r, q, and K.

Appendix C

The Operating Model

The operating model used in this study is based on the stock of Cape hake (*Merluccius capensis* and *M. paradoxus* — treated as a single species for assessment purposes) off northern Namibia (15°S–25°S). It was originally developed by ICSEAF as a basis to test different stock assessment methodologies proposed for this resource (ICSEAF 1990). It incorporates density dependent recruitment, which is functionally

related to the spawner biomass, and includes a number of error terms.

The resource dynamics are described and the symbols defined below.

Basic Dynamics

$$N_{y+1,a+1} = N_{y,a} \exp\left(-Z_{y,a}\right) \tag{C1}$$

where,

$Z_{y,a} = M + S_{y,a} F_y$,

$S_{y,a} = S_a \exp\left(\tau_{y,a} - \sigma_s^2/2\right)$,

$\tau_{y,a}$ is random variate from a normal distribution with zero mean and variance σ_2^s,

$S_a = 1/(1 + \exp[-(a - a_r)/\delta])$,

$a = 0, 1, \ldots, a_{\max}$ and

$y = -19, -18, \ldots, n$.

Thus, S_a is the average selectivity-at-age, and $\tau_{y,a}$ (process error) describes the variability in this quantity.

Stock-Recruitment Relationship (Beverton-Holt Form)

$$N_{y,0} = \frac{\alpha B_y^s}{\beta + B_y^s} \exp\left(\varepsilon_y - \sigma_r^2/2\right) \tag{C2}$$

where $B_y^s = \sum_{a=a_m}^{a_{\max}} w_a N_{y,a}$, and ε_y is a normal random variate with zero mean and variance σ_r^2, i.e., ε_y (process error) accounts for residual fluctuations in annual recruitment about a deterministic stock-recruitment relationship.

Catch-by-Mass

$$C_y = \sum_{a=0}^{a_{\max}} w_{a+1/2} N_{y,a} S_{y,a} F_y \frac{1 - exp\left(-Z_{y,a}\right)}{Z_{y,a}} \tag{C3}$$

Effort/fishing mortality relationship:

$$F_y = q E_y \exp\left(\eta_y - \sigma_q^2/2\right), \tag{C4}$$

where η_y is a normal random variate with zero mean and variance σ_q^2. Thus q is the average catchability, and η_y describes fluctuations about this average. This is observation error because it occurs in the relationship between a population dynamics quantity (annual fishing mortality F_y) and an observed quantity (annual effort E_y).

Annual Average Exploitable Biomass (taken as a mid-year value)

$$B_y^e = \sum_{a=0}^{a_{\max}} w_{a+1/2} S_a N_{y,a} \exp\left(-Z_{y,a}/2\right) \tag{C5}$$

Definition of Symbols

$N_{y,a}$ - number of fish of age a at the start of year y

$N_{y,0}$ - 0 year-class strength ("recruits") in year y

$Z_{y,a}$ - total mortality on age-class a during year y

M - natural mortality (assumed independent of age and year)

S_a - (average) age-specific selectivity

F_y - (asymptotic) fishing mortality in year y

a_r - age-at-50%-selectivity

δ - parameter which determines the width of the selectivity ogive

B_y^s - spawner biomass at the start of year y

α, β - parameters of the Beverton-Holt stock recruitment relationship

a_m - age-at-maturity

w_a - mass of a fish of age a at the start of the year

$w_{a+1/2}$ - mass of a fish of age a in the middle of the year

C_y - catch (by mass) in year y

q - catchability coefficient

E_y - fishing effort applied in year y

B_y^e - average exploitable biomass during year y.

Initial Conditions

The initial conditions ($y = -19$) correspond to the deterministic unexploited equilibrium level for the stock ($\varepsilon_y = 0$, $F_y = 0$ for $y \leq -19$):

$$N_{-19,a} = N_{-20,a}\, e^{-aM} \tag{C6}$$

where,

$$N_{-20,a} = \frac{\alpha \sum_{a=a_m}^{a_{\max}} w_a e^{-aM} - \beta}{\sum_{a=a_m}^{a_{\max}} w_a e^{-aM}} \tag{C7}$$

The catches in years -19 to 0 are all zero, but the resource is simulated from year -19 to allow recruitment fluctuations over this period, so that the resource starts from unexploited equilibrium only in some average sense.

Parameter Values

The following "base case" parameter values are taken from Butterworth (1988) and ICSEAF (1990), and were based upon production model and VPA assessments of the resource available at that time:

$M = 0.3\text{yr}^{-1}$

$a_{\max} = 9$ yr

$a_m = 3$ yr

$a_r = 3$ yr

$\delta = 0.5$ yr

$w_{a+1/2}$, see Table C1

$w_a = 1/2\left(w_{a-1/2} + w_{a+1/2}\right)$ (kg)

$\alpha = 6300$ (million)

$\beta = 1171.8$ ('000 t)

$q = 0.0004$ (Spanish OTB-7 t $\text{h}^{-1})^{-1}$

$\sigma_r = 0.2$

$\sigma_q = 0.1$

$\sigma_s = 0.2$

$n = 23$

C_y, see Table C2

TAC Evaluation

The "true" $\text{TAC}_{0.1}$ is the "$F_{0.1}$" strategy TAC for year $n+1$, and is computed by means of the formula:

Table C1. Values used for mass- and selectivity-at-age in the operating model.

AGE	Mass at mid-year[1,2]	Age-specific selectivity[3]
0	0.020	0.002
1	0.056	0.018
2	0.087	0.119
3	0.177	0.500
4	0.386	0.881
5	0.622	0.982
6	0.834	0.998
7	1.043	1.000
8	1.207	1.000
9	1.333	1.000

[1] Data source: table A.V.1 of Butterworth et al. (1986) for the hake stock off northern Namibia.

[2] $w_{a+1/2}$

[3] $S_a = \left[1 + \exp\left(-(a-3)/0.5\right)\right]^{-1}$

Table C2. Values used for annual catches in the operating model.

Year number	Catch (tons)*
1	93,510
2	212,444
3	195,032
4	382,712
5	320,430
6	402,467
7	365,557
8	606,084
9	377,642
10	318,836
11	309,374
12	389,020
13	276,901
14	254,251
15	170,006
16	97,181
17	90,523
18	176,532
19	216,181
20	228,172
21	212,172
22	231,179
23	136,962

* Data source - ICSEAF statistical bulletins for the hake stock off northern Namibia

$$\text{(C8)}\quad \text{TAC}_{0.1} = \sum_{a=0}^{a_{\max}} w_{a+1/2} S_a F_{0.1} N_{n+1,a} \frac{1 - \exp\left(-Z_{0.1,a}\right)}{Z_{0.1,a}}$$

where $Z_{0.1,a} = M + S_a F_{0.1}$, and $F_{0.1}$ is obtained by solving for $F_{0.1}$ the equation:

$$\text{(C9)}\qquad \left.\frac{\partial Y(F)}{\partial F}\right|_{F=F_{0.1}} = 0.1 \left.\frac{\partial Y(F)}{\partial F}\right|_{F=0}$$

where $Y(F)$ is the sustainable yield [given by equation C8 when the numbers-at-age correspond to a steady state situation] as a function of F.

The terminology "$F_{0.1}$" is used to differentiate this strategy from the conventional $F_{0.1}$. For age-structured assessments, such as those based on VPA, the stock-recruit relationship is effectively ignored when an estimate of $F_{0.1}$ based on yield-per-recruit calculations is used to provide the basis to calculate $\widetilde{\text{TAC}}_{0.1}$. The production model assessment method linked to an $f_{0.1}$ strategy calculates $\text{TAC}_{0.1}$ ignoring age-structure effects. Equations (C8) and (C9) take both stock-recruit and age-structure effects into account in calculating the "true" $\text{TAC}_{0.1}$ under this "$F_{0.1}$" strategy.

Appendix D

The Conditioning of Variance Measures in the Simulations

Variance measures depend on "conditioning", i.e., what is kept fixed and what is allowed to vary in the series of replicates considered. The operating model of Appendix C which generates the replicates (artificial data sets) considered in the calculations of this paper, is conditioned on a fixed annual historic total catch series and constant values for various biological parameters. However, process errors in the selectivity-at-age ($\tau_{y,a}$) and recruitment (ε_y), and observation errors related to catchability fluctuations (η_y), differ between replicates U.

This choice of conditioning does not create any difficulties, at least as far as definitions are concerned, when the management quantity Q of interest is MSY. This is because MSY^{True} is the same for each replicate, as it is a function of the values of the biological parameters of the operating model only, and does not depend on the values of the process or observation errors for a particular replicate. [Note that MSY is defined in terms of the average selectivity-at-age S_a, and is thus independent of the $\tau_{y,a}$ — see equations (C8) and (C9).] The same is not true, however, for the other quantities under consideration: B^*, B^*/K and $\text{TAC}_{0.1}$. This is because their true values depend on the series of process errors that applied for a particular replicate U, i.e., (D.1)

$$Q^{\text{True}} = Q^{\text{True},U}. \tag{D1}$$

This is illustrated in Fig. D1, which plots the estimates $\left(\widehat{B^*/K}\right)^U$ against $(B^*/K)^{\text{True},U}$ for a number of the data sets generated by the operating model of Appendix C, and shows that these two quantities are correlated, as one would hope.

The appropriateness of the definition of equation (2) of the true value of the variance now becomes questionable, because such a measure would include the effect of variation in $(B^*/K)^{\text{True}}$, as well as that related to the estimation variation. In a real application of a variance-estimation procedure, the actual value of $(B^*/K)^{\text{True}}$ would be fixed, so that a case could be made for extending the conditioning to that subset of the generated series of process errors ($\tau_{y,a}$ and ε_y) which correspond almost identically to a specific value of $(B^*/K)^{\text{True}}$. However, such a subset would be computationally intensive to generate. Further, since $(B^*/K)^{\text{True}}$ is not known exactly for the real stock of northern Namibian hake to which the calculations of this paper pertain, it can be argued that estimator performance should be considered over a range of such values, and statistics reported which are aggregates over that range.

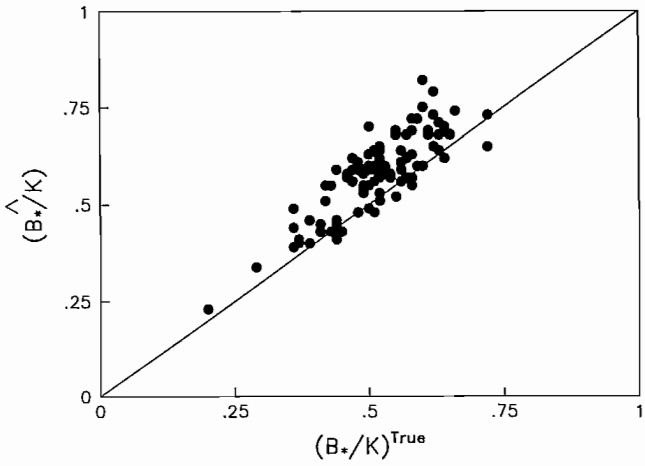


Fig. D1. A plot of model estimates against true values of B^*/K for a number of data sets generated by the "base case" operating model (before conditioning on $(B^*/K)^{\text{True}}$). The solid line indicate zero bias: $B^*/K = (B^*/K)^{\text{True}}$.

The approach adopted in this paper has therefore been to condition the artificial data sets generated to process error series which correspond to values of $(B^*/K)^{\text{True}}$ within 0.1 of its median value for data sets not so conditioned. To remove the effect of the correlation between true and estimated values from the calculation of the true variance, a regression model is fitted to the $(Q^{\text{True},U}, \widehat{Q}^U)$ data:

$$\widehat{Q}^U = (1+\beta)\,Q^{\text{True},U} + \varepsilon^U \tag{D2}$$

i.e., where the residuals, ε^U, are assumed to be normally distributed with variance σ^2 which is independent of Q^{True}. It then follows that:

$$\text{Rel Bias } \widehat{Q} = \widehat{\beta} \tag{D3}$$

$$\text{Var}^{\text{True}}\,\widehat{Q} = \widehat{\sigma}^2 \tag{D4}$$

where $\widehat{\beta}$ and $\widehat{\sigma}^2$ are the estimates obtained from the regression. The results of this process are illustrated in Fig. D2 for (B^*/K), B^* and $\text{TAC}_{0.1}$, and provide no indication that the regression model assumed is inappropriate.

Given the estimate of equation (D4) for $\text{Var}^{\text{True}}\widehat{Q}$, equations (9) and (10) of the main text can be applied as they were for the case $Q = \text{MSY}$, to provide measures of relative bias and CV for estimates of standard error provided by the different variance-estimation procedures considered.

However, this approach cannot be extended immediately to equations (11) and (12) relating to the estimates of CV provided by these procedure. If $\text{Var}^{\text{True}}\widehat{Q}$ is independent of Q^{True}, then $\text{CV}^{\text{True}}\widehat{Q}$ will depend on Q^{True}, raising a problem of definition for the denominator terms in these two equations. To avoid this difficulty, a different regression model has also been fitted to the $(Q^{\text{True},U}, \widehat{Q}^U)$ data:

$$Q^{\text{True},U} = \left(1+\gamma+\mu^U\right) Q^{\text{True},U} \tag{D5}$$

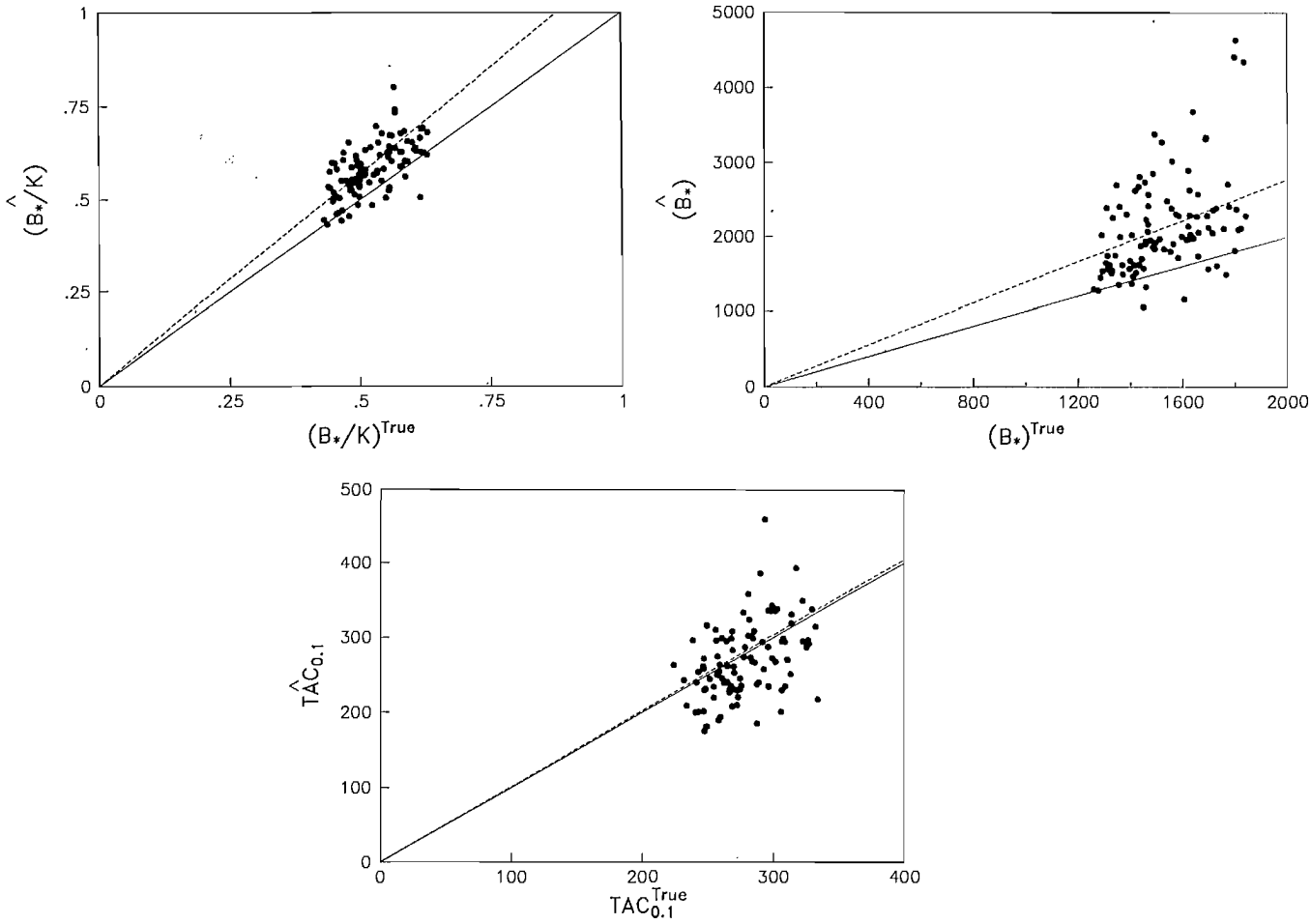


Fig. D2. Plots of model estimates against true values for data sets generated by the "base case" operating model, and conditioned on $(B^*/K)^{\text{True}}$ as described in Appendix D. The solid line indicates zero bias, and the dashed line the actual bias as determined from the regression model of Equation (D2). Plots are shown for a) B^*/K; b) B^* and c) $\text{TAC}_{0.1}$. Units for b) and c) are '000 t.

i.e., a constant coefficient of variation rather than constant variance model for the residuals μ'' which are assumed to be normally distributed with zero mean and variance σ'^2. From this it follows that:

$$\text{CV}^{\text{True}}\widehat{Q} = \widehat{\sigma}' \tag{6}$$

where $\widehat{\sigma}'$ is the estimate obtained from the regression, so that Equations (11) and (12) can then be applied straightforwardly. Naturally, the two regression models assumed [equations (D2) and (D5)] cannot both be valid. However, given the limited range of the independent variable for the conditioning chosen, this inconsistency is of no quantitative consequence for the results reported. Table 3 shows that the estimated slopes of the regression models $[(1+\beta)$ and $(1+\gamma)]$ are scarcely sensitive to the error model assumed for the residuals.

Evaluating the Accuracy of Projected Catch Estimates From Sequential Population Analysis and Trawl Survey Abundance Estimates

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Bottom trawl survey abundance estimates are the key indices used to calibrate sequential population analyses (SPA) for many of the groundfish resources in the Northwest Atlantic. The uncertainty associated with survey estimates is therefore an important element in the evaluation of population estimates from SPA. In this paper, we compare three methods of assessing the accuracy of estimates of projected catch from SPA where survey abundance estimates alone were used in a nonlinear least-squares calibration (ADAPT). The first two methods are forms of 'bootstrap' resampling where either the actual survey data is resampled or random numbers from a statistical distribution are 'sampled' to form new estimates of abundance from the survey. The third is a linear approximation method for nonlinear least-squares. All of the methods show to the same degree that the estimates of projected catch can be biased. This bias has to be taken into account when calculating the other measures of accuracy such as confidence intervals.

Le relevé au chalut de fond et les estimations d'abondance sont les outils essentiels d'étalonnage des analyses de population séquentielles (APS) pour bon nombre des ressources démersales de l'Atlantique nord-ouest. L'incertitude associée aux estimations de relevé est donc un élément important de l'évaluation des estimations de population à partir d'APS. Dans le présent article, on compare trois méthodes d'évaluation de l'exactitude des estimations des prises prévues à partir de l'APS dans des zones où seules les estimations d'abondance ont été utilisées pour l'étalonnage des moindres carrés curvilinéaires (ADAPT). Les deux premières méthodes sont des formes de rééchantillonnage «bootstrap» dans le cadre desquelles les données de relevé sont rééchantillonnées ou les nombres aléatoires d'une distribution statistique sont «échantillonnés» afin d'obtenir des nouvelles estimations d'abondance à partir du relevé. La troisième méthode est une méthode d'approximation linéaire pour les moindres carrés curvilinéaires. Toutes les méthodes révèlent, à un degré identique, que les estimations de prises prévues peuvent être biaisées. Ce biais doit être pris en considération lorsqu'on effectue d'autres mesures d'exactitude, telles que celles des intervalles de confiance.

Scientific advice on catch levels for groundfish stocks under the jurisdiction of the Canadian Atlantic Fisheries Scientific Advisory Committee (CAFSAC) has generally required the provision of a projected catch estimate at some reference target level (e.g., $F_{0.1}$) for the following fishing year. For many stocks, estimates of projected catch are obtained by projecting from the results of a sequential population analysis (SPA) of the catch-at-age of the commercial fishery calibrated to indices of abundance-at-age from research trawl surveys.

In the CAFSAC forum at present, calibration of SPA with abundance indices are generally carried out with the ADAPT framework (Gavaris 1988) which uses nonlinear least-squares to estimate the parameters. Within this framework, statistical error models are assumed for the survey abundance indices and the statistical properties of the estimation method are used to provide estimates of the variances and covariances of the model parameters. The accuracy of the parameter estimates (e.g., projected catch) within the model can be assessed by calculating estimated bias, standard errors and/or frequency percentiles for the construction of confidence intervals. The validity of the latter two statistics will be a function of the appropriateness of the linear approximation to the model used near the solution. In addition, confidence intervals require the assumption of some statistical distribution for their construction. Gavaris (1993) discusses the estimation of the bias of parameter estimates from ADAPT using Box's (1971) method. Note that for brevity, we use the term "accuracy" to generally refer to measures of statistical error such as bias, standard error and confidence intervals of estimates.

As an alternative to the analytical approach, computer-intensive methods for determination of accuracy have been gaining popularity since the "bootstrap" method was introduced by Efron (1979, 1982). The approach uses the observed data to structure simulations to derive statistical properties of estimates. Bootstrap methodology has developed into a wide range of techniques for an equally wide range of problems

(Hinkley 1988) bolstered primarily by the appealing feature that no statistical model need be assumed for the data in deriving the accuracy of estimates. In a fisheries application, Pelletier and Gros (1991) have used bootstrap resampling of the actual data sampled from the commercial landings for the estimation of catch and weight-at-age, to estimate the variance of equilibrium yield per recruit.

In many practical situations, the low sampling intensity or the complexity of the sampling design introduce complications for resampling which are difficult or impossible to reconcile. Restrepo et al. (1990) have circumvented these difficulties by proposing a method where the uncertainties in the basic data and in the assumptions (e.g., constant natural mortality) used in SPA are modelled by generating pseudo-random numbers from various statistical distributions (e.g., lognormal for survey abundance indices). Simulated data for surveys, commercial catch-at-age, commercial catch rate, etc., generated in this manner are used to calibrate a SPA with ADAPT. This process is replicated a large number of times and frequency distributions are constructed for the quantities estimated by ADAPT. This is similar to the approach used by Restrepo and Fox (1988) to incorporate uncertainties into yield-per-recruit models. Hoenig et al. (1990) have suggested that these frequency distributions can be used to make probabilistic statements, similar to those from confidence intervals, about population characteristics (e.g., fishing mortality rate, biomass).

The pseudo-random numbers used in the above simulation approaches were generated to mimic statistical distributions whose parameter values were based on observed estimates (e.g., means, variances) or on possible likely parameter values, such as assuming that instantaneous natural mortality is a uniform random variable which varies between 0.1 and 0.4. Measures of accuracy from these approaches can be highly dependant upon the statistical model and the parameter values that were chosen. The pseudo-random number approach has been referred to as a 'parametric' bootstrap (e.g., Punt and Butterworth 1993) because the data are 'sampled' from an assumed parametric statistical distribution using a random number generator. However, this nomenclature differs somewhat from that used by Efron (1982, 1987a) where parametric bootstrap estimates of accuracy are constructed based on properties of the 'assumed' distribution. In principle, application of the parametric bootstrap does not require Monte-Carlo simulation. However, except for the simple case of a normal error distribution and linear behaviour near the solution, the complexity of deriving the distribution of estimators obtained from ADAPT, and indeed many other models commonly used for fisheries assessment, from assumptions about the statistical models for the observed data used in these models is such that an analytical solution is unlikely. Monte-Carlo simulation therefore seems to be an expedient means of assessing accuracy for the general problem. We will refer to situations where the data have been simulated from a known distribution as the parametric bootstrap here and refer to the nonparametric bootstrap where the observed data has been resampled.

In this paper, we compare the estimates of accuracy from the three approaches, the analytical method, the nonparametric bootstrap and the parametric bootstrap. We confine our attention to estimating the accuracy of the projected catch and the proportion at age for the case where the trawl survey abundance indices are assumed to be the only source of uncertainty. This may appear unduly restrictive but we note that it is a common model for calibration of SPA as many consider it reasonable to assume that the errors in the abundance indices overwhelm other errors. Further, it allows us to compare results from the analytical approach and the nonparametric bootstrap to those of the parametric bootstrap. The analytical method and the nonparametric bootstrap have not been applied to more general models. We compare results for accuracy of estimates of projected catch and proportion at age for the Eastern Georges Bank haddock stock (Gavaris and Van Eeckhaute 1991). The SPA was calibrated using abundance-at-age estimates from the Canadian bottom trawl survey of Georges Bank. Although there are two USA surveys of the stock, only the Canadian survey was used in this study because it alone had large enough sample sizes within strata to permit adequate resampling for the nonparametric bootstrap.

Generation of Trawl Survey Data

Trawl Survey Indices

The Canadian trawl survey of Georges Bank has been conducted in February each year since 1986. The survey uses a stratified random design and is conducted using procedures similar to those of trawl surveys of the Scotian Shelf (Smith 1988). The stratification used in 1986 and 1987–1991 differs but the total area covered by the key strata for the stock is the same.

We will need the following definitions for quantities associated with the trawl surveys in any year y.

n_{hy}	=	the number of hauls or sets in stratum h ($h = 1, \ldots, L$),
n_y	=	$\sum_{h=1}^{L} n_{hy}$,
N_h	=	the total number of possible sets in stratum h,
N	=	$\sum_{h=1}^{L} N_h$,
I_{hiay}	=	the number of fish of age a caught in set i and stratum h,
$\bar{I}_{hay}$	=	the estimated mean abundance in stratum h,
I_{ay}	=	$\sum_{h=1}^{L} N_h \bar{I}_{hay}$, the estimated stratified total abundance for age a,

and $s^2_{hay} = \sum_{h=1}^{L} (I_{hiay} - \bar{I}_{hay})^2/(n_{hy} - 1)$ the estimated variance in stratum h for age a.

The variance of the stratified total abundance is estimated by

$$\widehat{\mathrm{Var}}\left(I_{ay}\right) = \sum_{h=1}^{L} N_h \left(N_h - n_{hy}\right) \frac{s^2_{hay}}{n_{hy}} . \quad (1)$$

Table 1. Comparison of bootstrap results with survey estimates for age=1 and y=1991. B= number of replications for bootstrap. The entry in the column labelled 'Variance' for the Survey estimate refers to the standard stratified estimate of variance of the total abundance (eq. (1) in text). The entries in the Variance column for the Bootstrap estimates were obtained from eq. (2) in the text. Note that the expectation of the variance for the naive bootstrap estimate was $\sum (N_h/n_{hy})^2 (n_{hy}-1) s^2_{hay} = 11473.69$ (eq. (3) in text).

Source	B	Estimated Total Abundance (1000's)	Variance
Survey		400	12195.26
Bootstrap			
$m_h = n_h$		404	12510.96
$m_h = n_h - 1$	100	403	12232.02
$m_h = n_h - 3$		391	11296.85
$m_h = n_h$		397	11214.81
$m_h = n_h - 1$	500	400	12262.90
$m_h = n_h - 3$		406	13042.73
$m_h = n_h$		395	11757.71
$m_h = n_h - 1$	1000	408	12256.73
$m_h = n_h - 3$		403	12656.02
$m_h = n_h$		401	11388.74
$m_h = n_h - 1$	5000	402	12362.76
$m_h = n_h - 3$		402	11769.77

Table 2. Sample sizes (n_{hy}) for each stratum from the Canadian spring surveys of unit areas 5Zj and 5Zm, Georges Bank. * Strata boundaries in 1986 were different from those in 1987–1991 but area covered was the same for all years.

	Strata			
Year	1	2	3	4
1986*	10	17	3	–
1987	9	7	4	6
1988	24	28	8	6
1989	18	26	8	6
1990	23	22	11	8
1991	22	26	11	8

Bootstrap Resampling of Survey Data

The main aim here is to create a population of bootstrap survey estimates of abundance for some age and year which have the same expectation and variance as the original survey estimate for that age and year. Five hundred sets of estimated numbers of haddock from the groundfish survey for ages 1–8 and for the years 1986–1991 were generated in one of the two ways described below.

i) Nonparametric Bootstrap

Bootstrap resampling is generally carried out by randomly sampling the original observations with replacement. This procedure assumes that the original observations were a random sample from the parent population. However, the survey abundances are estimated from data sampled using a stratified random survey design and this design complicates the bootstrap resampling. We have adopted the following procedure of Rao and Wu (1988) to resample data collected via a stratified random survey design.

1. Take a simple random sample $\left\{I^*_{hiay}\right\}_{i=1}^{m_{hy}}$, of size $m_{hy}\,(\leq n_{hy})$ with replacement from the given sample $\left\{I_{hiay}\right\}_{i=1}^{n_{hy}}$ in each stratum h. This was done independently for each strata.

$$\tilde{I}_{hiay} = \bar{I}_{hay} + \left(\frac{m_{hy}}{n_{hy}-1}\right)^{1/2} \left(I^*_{hiay} - \bar{I}_{hay}\right)$$

$$\tilde{I}_{hay} = m_{hy}^{-1} \sum_{i=1}^{m_{hy}} \tilde{I}_{hiay}$$

$$\tilde{I}_{ay} = \sum_{h=1}^{L} N_h \tilde{I}_{hay}$$

2. Replicate step 1 B times to obtain $\tilde{I}^1_{ay}, \ldots, \tilde{I}^B_{ay}$. The bootstrap estimate of the stratified mean is calculated as $\tilde{I}^{\bullet}_{ay} = \sum_{b=1}^{B} \tilde{I}^b_{ay}/B$ with variance estimated by

$$\text{(2)} \qquad \text{Var}\left(\tilde{I}^{\bullet}_{ay}\right) = \frac{1}{B-1} \sum_{b=1}^{B} \left(\tilde{I}^b_{ay} - \tilde{I}^{\bullet}_{ay}\right)^2.$$

Rao and Wu (1988) have demonstrated that the expectation of eq. (2) when $m_{hy} = n_{hy}$ and $\tilde{I}_{hiay} = I^*_{hiay}$ (referred to as the naive bootstrap), is

$$\text{(3)} \qquad \text{E}\left(\text{Var}\left(\tilde{I}^{\bullet}_{ay}\right)\right) = \sum_{h=1}^{L} N_h^2 \left(\frac{n_{hy}-1}{n_{hy}}\right) \frac{s^2_{hay}}{n_{hy}}$$

and therefore it is an inconsistent estimate of the variance in eq. (1). Efron (1982) showed that setting $m_{hy} = n_{hy} - 1$ will reduce the bias from eq. (2). Rao and Wu (1988) proposed setting $m_{hy} = n_{hy} - 3$ based upon comparing the bootstrap third moment $\text{E}_*\left(\tilde{I}^b_{ay} - \bar{I}^{\bullet}_{ay}\right)^3$ with the unbiased estimate of the third moment of $\bar{I}_{ay}$. We tested these suggestions by calculating the bootstrap estimates of mean and variance for each of $m_{hy} = n_{hy}, n_{hy}-1, n_{hy}-3$ for the abundance of age 1 haddock from 1991 (Table 1) and then compared these with the original survey estimates and the estimate from eq. (3). Note that as the number of replications increases from 100 to 5000 the variance of the bootstrap estimate for $m_{hy} = n_{hy}$ tends toward the quantity given by eq. (3) which is presented in the caption of Table 1. On the other hand, the variances for the other choices of m_{hy} are closer to the variance observed from

Table 3. Stratified total numbers (1000's) of haddock (a) and standard errors (b) for unit area 5Zj and 5Zm from the Canadian spring surveys of Georges Bank.

a) Stratified total numbers.

	Age							
Year	1	2	3	4	5	6	7	8
1986	5057	306	8175	997	189	348	305	425
1987	46	4286	929	3450	653	81	387	135
1988	971	49	12714	257	4345	274	244	130
1989	47	6473	959	2814	239	521	40	34
1990	726	108	12302	166	4465	299	1370	144
1991	400	2175	137	10776	116	1868	117	497

b) Standard errors.

	Age							
Year	1	2	3	4	5	6	7	8
1986	1656.58	109.06	3424.18	368.85	57.50	112.40	109.98	137.54
1987	38.33	2126.35	422.64	1437.77	242.11	30.87	144.22	48.05
1988	299.40	15.52	3798.65	73.86	1187.26	72.24	61.03	31.16
1989	33.71	1992.94	327.53	559.95	48.58	128.36	8.19	8.96
1990	270.03	35.69	2707.90	31.30	816.72	57.08	254.76	29.32
1991	110.43	416.30	27.32	2993.73	34.37	641.92	42.74	164.99

the survey. The results for $m_{hy} = n_{hy} - 1$ appeared to be more reliable than those for $m_{hy} = n_{hy} - 3$ and therefore we chose to do all of our bootstrap resampling with $m_{hy} = n_{hy} - 1$. A recent study by Kovar et al. (1988) on resampling from stratified random schemes came to similar conclusions regarding the merits of using $m_{hy} = n_{hy} - 1$ over $m_{hy} = n_{hy} - 3$.

An attractive characteristic of the Canadian Georges Bank survey for resampling is that there are relatively large numbers of survey sets in each of the strata (Table 2). Surveys on the Scotian Shelf often have only between two to eight sets (and sometimes one set) per stratum which would cause some difficulties in using resampling $m_{hy} = n_{hy} - 1$ observations per stratum.

ii) Parametric Bootstrap

We assumed that the stratified mean abundance of haddock of age a in year y was a lognormal random variate with mean and variance equal to that estimated from the survey for the same age and year. Pseudo-random mean abundances were generated in the following manner. Mean abundances were converted to total abundances by multiplying each number generated in step 4 by the N total number of possible sets in the survey.

1. The stratified mean and variance observed in the survey for year y and age a are used for the lognormal mean $\left(\alpha_{ay} = \exp\left(\mu_{ay} + \sigma_{ay}^2/2\right)\right)$ and variance $\left(\beta_{ay}^2 = \exp\left(2\mu_{ay} + \sigma_{ay}^2\right) \times \left(\exp\left(\sigma_{ay}^2\right) - 1\right)\right)$.

2. The mean $\left(\mu_{ay}\right)$ and standard deviation $\left(\sigma_{ay}\right)$ for the base normal distribution are derived as

$$\mu_{ay} = \ln\left(\alpha_{ay}\right) - \frac{\sigma_{ay}^2}{2}$$

$$\sigma_{ay} = \left[\ln\left(\frac{\beta_{ay}^2}{\alpha_{ay}^2} + 1\right)\right]^{1/2}$$

3. Generate B normal pseudo random numbers $\left(z_{ayb};\ b = 1, \ldots, B\right)$ with domain of support $N\left(z_{ayb} \mid \mu_{ay} = 0,\ \sigma_{ay} = 1\right)$.

4. Calculate $I_{ayb} = \exp\left(\mu_{ay} + \sigma_{ay} z_{ayb}\right)$ to give lognormal pseudo random variables with mean and variance as defined in step 1; $\left(\Lambda\left(I'_{ayb} \mid \mu_{ay},\ \sigma_{ay}\right)\right)$.

Results for Survey Estimates

The estimated stratified total numbers (1000's) and associated standard errors for haddock ages 1–8 from the Canadian survey of Georges Bank are presented in Tables 3a and 3b, respectively. These estimates, calculated using the standard formulae for stratified random survey designs (Cochran 1977; Smith 1988) were used to initiate the lognormal pseudo-random number generator for the parametric bootstrap estimates.

The frequency distributions of the stratified survey abundance estimates at age and year from the parametric and nonparametric bootstrap procedures were similar with respect to general shape, however the nonparametric bootstrap estimates

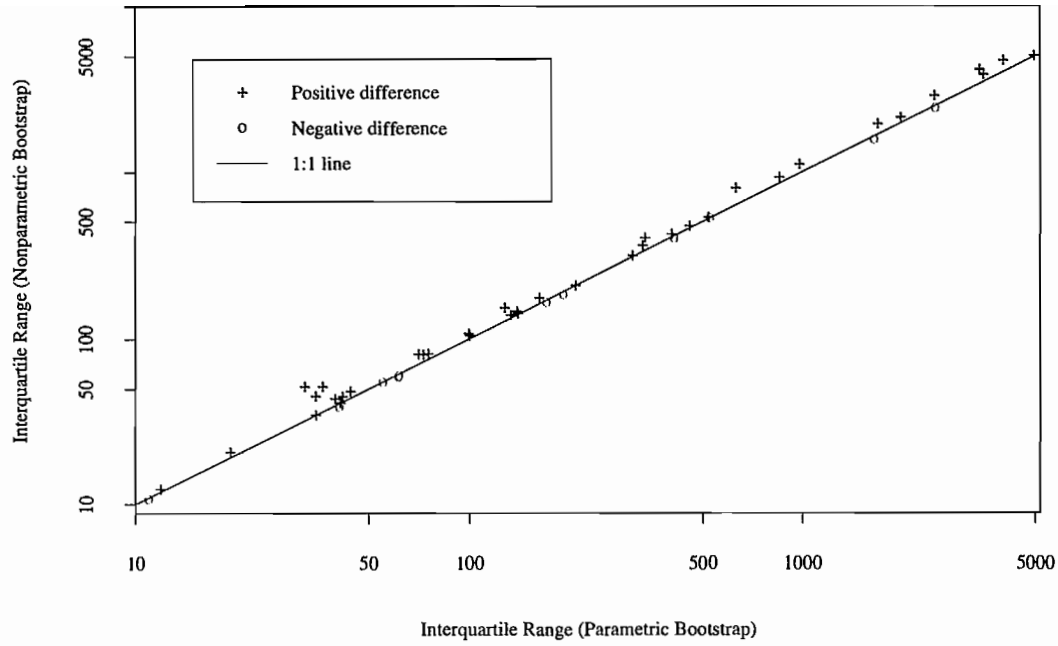


Fig. 1. Comparison of Interquartile ranges for parametric and nonparametric bootstrap samples for all ages and years. + = Interquartile (nonparametric) > Interquartile (parametric). o = Interquartile (nonparametric) ≤ Interquartile (parametric).

tended to be more variable. We measured this variability by the interquartile range of the 500 replications for each age and year from the nonparametric and parametric bootstrap resampling. While the differences between the interquartile ranges of the two types of bootstrap resampling were small, the interquartile ranges for the nonparametric bootstrap were more often larger than those for the parametric bootstrap (Fig. 1).

Bootstrap Analyses of Estimates From ADAPT

An SPA model was fit to each table (8 ages × 6 years) of above generated survey estimates using the ADAPT formulation described below. This resulted in 500 estimates of all of the parameters of interest, including projected catch.

Calibration of SPA model using ADAPT

Generally, the abundance indices are characterized as nonlinear functions of the commercial catch at age and the model parameters in the SPA model. The adaptive framework was introduced to allow flexibility in definition of model equations while maintaining a consistent statistically based (least squares) estimation procedure. The most common formulation in use employs population abundance, N_{at}, at age a ($a = 1, \ldots, A$) in the terminal year t as model parameters along with κ_a which is the catchability of the fish of age a to the survey. We depart from that strategy to use projected catch and proportion of the projected catch at each age group, for the year $t + 2$ as parameters of the model in place of population abundance. Note that stock assessments are generally conducted in year $t + 1$ with the SPA model updated for results of fishing in year t to give projections for year $t + 2$. The reason for the change in parameterization is that we were primarily interested in the statistical properties of the projected catch. By making projected catch an explicit parameter, these properties are immediately available. Though we could have estimated the projected catch as a function of terminal year population abundance, the relationships are not linear and introduce unnecessary computations.

A Levenberg-Marquardt algorithm was used to solve for parameters,

$$\begin{aligned} \psi_{t+2} &= \text{projected catch in target year.} \\ \pi_{a,t+2} &= \text{proportion of catch biomass at age } a, \\ &\quad a = 1, \ldots, 7. \end{aligned}$$

where

$$\pi_{8,t+2} = 1 - \sum_{a=1}^{7} \pi_{a,t+2}$$

for the following system of catch equations (Megrey 1989; Sparre et al. 1989),

$$\begin{aligned} Y_{a,t+2} &= \psi_{t+2} \times \pi_{a,t+2}, \ (a = 1, \ldots, 8); \text{ where } Y \\ &\quad \text{equals catch biomass at age.} \\ W_a &= \text{weight of fish at age } a. \\ C_{a,t+2} &= \frac{Y_{a,t+2}}{W_a}, \ (a = 1, \ldots, 8), \text{ where } C \text{ equals} \\ &\quad \text{catch numbers at age.} \\ F_{ay} &= \text{Fishing mortality.} \\ M &= \text{Natural mortality.} \end{aligned}$$

$$N_{a,t+2} = \frac{C_{a,t+2}\left(F_{a,t+2}+M\right)}{F_{a,t+2}\left(1-\exp\left(-\left[F_{a,t+2}+M\right]\right)\right)}, \quad (a=1,\ldots,8).$$

$$N_{a,t+1} = N_{a+1,y+1}\exp\left[F_{a,t+1}+M\right], \quad (a=1,\ldots,7),$$

$$N_{a-1,y-1} = N_{ay}\exp[M] + C_{a-1,y-1}\exp[M/2] \quad (a=2,\ldots,8 \text{ and } y=1,\ldots,t).$$

where,

$$F_{8,y} = \ln\left(\frac{\sum_{a=4}^{6} N_{ay}}{\sum_{a=5}^{7} N_{a,y+1}}\right) - M, (y=1,\ldots,t-1)$$

$$N_{8,y} = \frac{C_{8,y}\left(F_{8,y}+M\right)}{F_{8,y}\left(1-\exp\left(-\left[F_{8,y}+M\right]\right)\right)}, \quad (y=1,\ldots,t-1)$$

while minimizing the differences between the observed and predicted survey log numbers at age,

$$\sum_{y=1}^{t}\sum_{a=1}^{8}\left(\ln(I_{ay}) - \ln\left(\hat{\kappa}_a N_{ay}\right)\right)^2 ,$$

The calculation of the trawl survey indices, I_{ay}, was discussed in the previous section. To reduce computation time, the catchabilities were not explicitly searched for with the nonlinear minimization algorithm but were computed conditional on the other parameters according to

$$\widehat{\kappa}_a = \exp\left[\sum_{y=1}^{t}\left(\ln\left(I_{ay}\right) - \ln\left(N_{ay}\right)\right)/t\right] .$$

Natural mortality M, is assumed constant and equal to 0.2 for all ages and years. The catch is projected for a target F in year $t+2$ equal to the $F_{0.1}$ level of 0.25 assuming that the F for the intervening year, $t+1$, was also equal to $F_{0.1}$.

Bootstrap Measures of Accuracy

In general, the bootstrap method is used to investigate the properties of some estimator of a parameter θ given a finite set of observations x_i; $(i=1,\ldots,n)$ and an estimator of the form $\widehat{\theta} = g(x_1,\ldots,x_n)$. A series of n draws is made from the x_i by sampling with replacement and a new estimate, $\widehat{\theta}^*$ is calculated from these *resampled* data. This process is repeated a large number of times to obtain B estimates $\widehat{\theta}_i^*$ $(i=1,\ldots,B)$ of the original $\widehat{\theta}$. The average $\widehat{\widehat{\theta}}^*$ of the $\widehat{\theta}_i^*$ over B, is used to assess the bias and the standard error (SE) of $\widehat{\theta}$ as,

$$\text{BIAS} = \widehat{\widehat{\theta}}^* - \widehat{\theta} \tag{4}$$

and

$$\text{SE} = \left(\sum_{i=1}^{B}\left(\widehat{\theta}_i^* - \widehat{\widehat{\theta}}^*\right)^2/B - 1\right)^{\frac{1}{2}} . \tag{5}$$

In our situation the x_i are catches of haddock of a specific age from any one stratum in any one year. The $\widehat{\theta}$ that we are interested in is the estimated projected catch and proportions at age from ADAPT and $g(x_1,\ldots,x_n)$ is a nonlinear function of the stratified total abundances which are estimated from the x_i via a complex survey design.

Confidence Intervals

There are a number of ways of computing confidence intervals from bootstrap estimates (see DiCiccio and Tibshirani 1987; Efron 1987a) but we consider two of the more basic methods here. In the "percentile" method (**Perc**), an estimate of the distribution function of some estimator $\widehat{\theta}$, $G(s) = \text{Prob}\left(\widehat{\theta} \le s\right)$ is given as (Efron 1982),

$$\widehat{G}(s) = \sum_{i=1}^{B} I_i/B ,$$

where

$$I_i = \begin{cases} 1, & \text{if } \widehat{\theta}_i^* \le s; \\ 0, & \text{otherwise.} \end{cases}$$

and $\widehat{\theta}_i^*$ denotes the ith bootstrap estimate of $\widehat{\theta}$ (e.g., projected catch) and B as before, denotes the number of bootstrap replications. Upper and lower α confidence bounds are calculated as $\widehat{G}^{-1}(1-\alpha/2)$, $\widehat{G}^{-1}(\alpha/2)$, respectively. This is a central confidence interval for $\widehat{\theta}$ and assumes that the $\widehat{G}\left(\widehat{\theta}\right) = 0.50$. Efron (1987a) introduced a correction to the **Perc** method to account for differences between $\widehat{\theta}$ and the actual median of the frequency distribution. The α upper and lower confidence bounds for this bias-corrected method (**BC**) are obtained as $\widehat{G}^{-1}\left\{\Phi\left(z^{(1-\alpha/2)} + 2z_0\right)\right\}$, $\widehat{G}^{-1}\left\{\Phi\left(z^{(\alpha/2)} + 2z_0\right)\right\}$, respectively; where Φ is the cumulative distribution function for a standard normal variate, z^t is the tth percentile and $z_0 = \Phi^{-1}\left\{\widehat{G}\left(\widehat{\theta}\right)\right\}$. The term z_0 is the correction for median and equals zero when $\widehat{\theta}$ is the median of $\widehat{G}(s)$; the **BC** and **Perc** methods are equivalent in this case.

Efron (1987a) introduced the $\mathbf{BC}_a$ method as a further improvement; however, the jury is still out on the best way to estimate the acceleration constant required for this method in the general nonparametric case (see Efron 1987b) and no work has been done on estimation of the acceleration constant when resampling data from a complex survey design.

Results of Bootstrap Analyses

Estimates of projected catch and proportions at age for the haddock data from ADAPT using the survey data in Tables 3a,b are presented in Table 4. Included in this table are the bias estimates for these parameters from Box's method and standard error estimates (see Gavaris 1993) applied to the original ADAPT run as well as the bias and standard error estimates from the two bootstrap methods using equations (2) and (3). The biases associated with the proportion at age estimates appear to be minor while the bias for projected catch is more substantial. Bias estimates from both bootstrap methods appear to agree quite closely with those from the Box method for most of the parameter estimates considered here. The

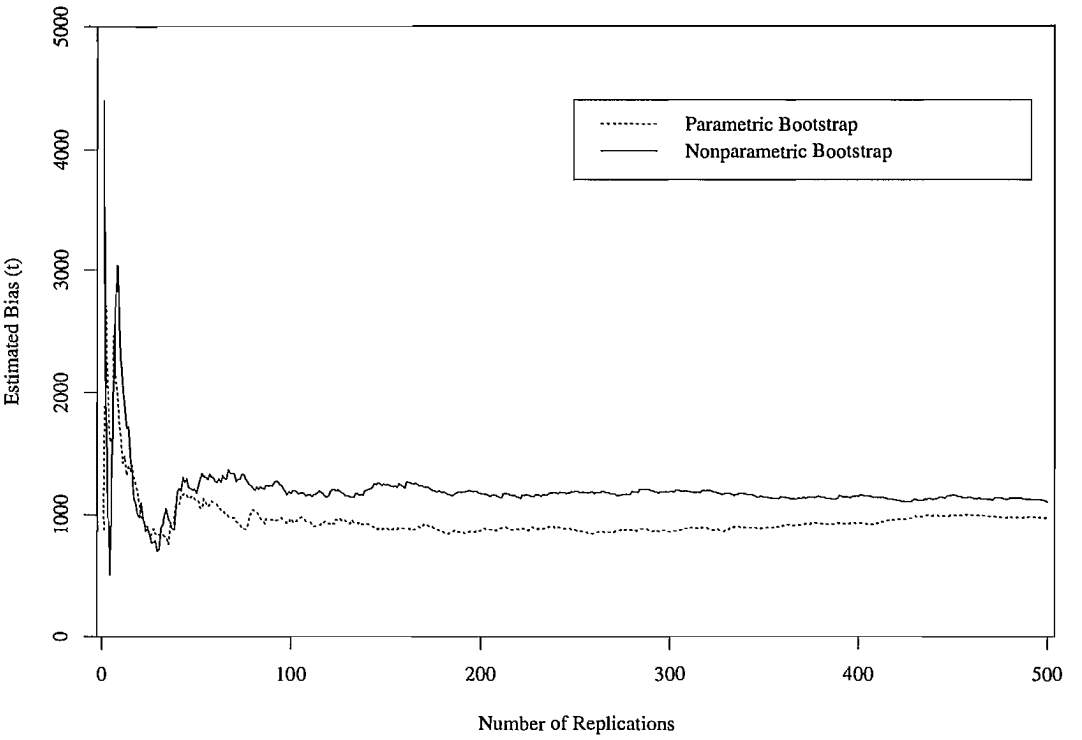


Fig. 2. Bootstrap estimates of bias of the estimated projected catch as functions of the number of resamplings used in the analysis.

stability of the bootstrap bias estimates for projected catch as a function of the number of replications was investigated in Fig. 2. We interpret this figure as indicating that, for this case the bootstrap bias estimates were relatively stable after $\approx$ 200 replications.

The bootstrap standard error estimates were both less than that from the analytical ADAPT estimate for projected catch and proportion at age estimates. As was the case for most of the bias estimates, the standard error estimates from the nonparametric bootstrap were intermediate between the parametric bootstrap estimates and the analytical estimates from ADAPT.

We have plotted the percentiles for the distributions of the parametric and nonparametric bootstrap estimates of projected catch on Fig. 3. The percentiles for the Student-*t* distribution for projected catch assuming a mean of 6218 t, standard error of 2558 t (Table 4) and 32 degrees of freedom (6 years $\times$ 8 years minus 16 parameter estimates) are also included on this plot for reference. This naive analytical confidence interval is included here for comparison only and is not recommended for routine application. More sophisticated techniques for deriving confidence regions are available (Ratkowsky 1983, p. 30) but require computationally intensive numerical methods. Hamilton (1986) describes a method for obtaining confidence regions which also accounts for the effect of non-linearity. For problems with appreciable non-linearity, as is the case here, these techniques result in assymetric bounds.

Note that all of the bootstrap cumulative curves are defined for positive values of projected catch only while the Student-*t* curve does include negative projected catches in the lower tail. The 90% central confidence bounds using the **Perc** method for the parametric and nonparametric bootstrap estimates of projected catch as well as the standard Student-*t* bounds are presented in Table 5. In our case, it appears that the assumption of a lognormal distribution for the parametric bootstrap has resulted in confidence intervals which are shorter than those obtained from bootstrapping the raw data. Note that the both of the **Perc** bootstrap intervals are shorter than that given by the Student-*t* method. The 'Shape' of each of the intervals is calculated as the logarithm of the ratio of the upper limit minus the median to the median minus the lower limit (Efron 1992). A shape of zero (0.0) indicates a symmetric distribution such as in the Student-*t* case. However, the **Perc** intervals are highly asymmetric in both cases. This diagnostic indicates that $\widehat{G}\left(\widehat{\theta} = 6218\,t\right) \neq 0.50$, as shown on Fig. 3 where the cumulative probability for the nonparametric bootstrap is 0.332 for $\widehat{\theta} = 6218\,t$. Applying the **BC** method reduces the asymmetry substantially and also reduces the length of the confidence intervals (Table 5). However, the **BC** interval for the parametric bootstrap continues to be shorter than that for nonparametric bootstrap.

Discussion

Although least-squares estimates for nonlinear models are asymptotically unbiased and minimum variance among linear estimators, there are no general guidelines on how large the sample size must be to achieve acceptable behaviour. Reparameterization may result in a model which is close to linear in behaviour and thus possess these desirable properties (Bates and Watts 1980), however the new parameters may not be

Table 4. Original estimates of projected catch and proportion at age from ADAPT. The bias estimates from ADAPT analysis were calculated using Box's method for estimating the bias of nonlinear models. Bias estimates from the Parametric bootstrap and the Nonparametric bootstrap are obtained from average values over the 500 replications (eq. (4) in text). The bootstrap estimates of standard error were calculated using eq. (5) in text.

Method	Projected Catch	Proportion at Age						
		1	2	3	4	5	6	7
				Original Parameter Estimates				
ADAPT	6218	0.0509	0.2356	0.0085	0.4485	0.0067	0.1732	0.0112
				Bias Estimates				
ADAPT	1353	0.0078	0.0045	0.0002	-0.0154	0.0002	-0.0009	0.0003
Nonparametric	1108	0.0051	0.0035	-0.0003	-0.0023	0.0000	-0.0034	-0.0005
Parametric	980	0.0040	0.0018	-0.0002	-0.0043	-0.0002	-0.0007	-0.0002
				Standard Error Estimates				
ADAPT	2558	0.0331	0.0931	0.0036	0.1047	0.0030	0.0630	0.0049
Nonparametric	2118	0.0198	0.0510	0.0026	0.0555	0.0018	0.0366	0.0023
Parametric	1773	0.0174	0.0449	0.0022	0.0465	0.0016	0.0342	0.0023

Table 5. Central 90% confidence interval of Projected Catch from bootstrap percentiles and Student-t distribution.

Measure	Parametric		Nonparametric		Student-t
	Percentile	Bias-Corrected	Percentile	Bias-Corrected	
Upper	10646	8401	11731	8595	10541
Lower	4891	4301	656	3730	1895
Length	5755	4100	7075	4865	8646
Shape	6.786	0.130	1.261	-0.046	0.000

the characteristics of interest. Pope and Stokes (1989) have reparameterized the sequential population analysis as a generalized linear model. Although the SPA could be expressed in a generalized linear model form, the resulting parameterization was such that ad hoc methods had to be developed to relate the parameters of the final model to the quantities normally obtained from SPA (fishing mortalities, etc.). John Shepherd (MAFF, Lowestoft, NR33 0HT, UK. pers. comm.) has exploited the fact that the fisheries assessment problem is close to linear when expressed in terms of logarithms of population abundance. However to obtain characteristics of interest to fisheries decision makers (projected catch, population abundance, fishing mortality rates) would involve non-linear transformations and necessitate consideration of bias at that stage. Given these considerations, examination of uncertainty must include bias and precision.

The two bootstrap methods and Box's method indicated that the estimates of projected catch for the problem studied were biased. The estimates of bias from each of the three methods were in reasonable agreement with the parametric bootstrap estimate giving the lowest estimate. The non-parametric bootstrap estimate requires no assumptions while the parametric bootstrap assumes an error distribution. Box's estimator is conditional on the model being correct and the residuals exhibiting a normal distribution. We conclude that for this problem, and perhaps for many fisheries assessment situations, Box's method for estimating the bias can provide a reasonable approximation which is easily computed and appears to be in close agreement with methods having fewer assumptions.

Both bootstrap estimates of the standard errors were smaller than the estimate from the linear approximation near the solution. The nonparametric bootstrap estimate of standard error was about half way between the estimates from the parametric bootstrap and the analytical result. The estimates of variance for parameters from non-linear models which exhibit bias in those parameters, will be an underestimate of the precision (Ratkowsky 1983). Seber and Wild (1989, p. 182) derive an approximation for the variance of the parameter estimates which incorporates additional terms accounting for intrinsic and parameter effects curvature. We are not aware of any attempts to estimate this approximation. These considerations suggest that the bootstrap estimates of standard error may be underestimating the precision or that the linear approximation near the solution is very poor and is distorting estimates.

It was in the calculation of confidence intervals that we found the most differences between the two bootstrap methods and the Student-t method which was based on estimates from the linear approximation method. The choice of an arbitrary statistical distribution (e.g., lognormal) to model the uncertainty of the survey estimates results in shorter confidence intervals than those obtained based on resampling the original data. The interquartile ranges from the bootstrap data (Fig. 1)

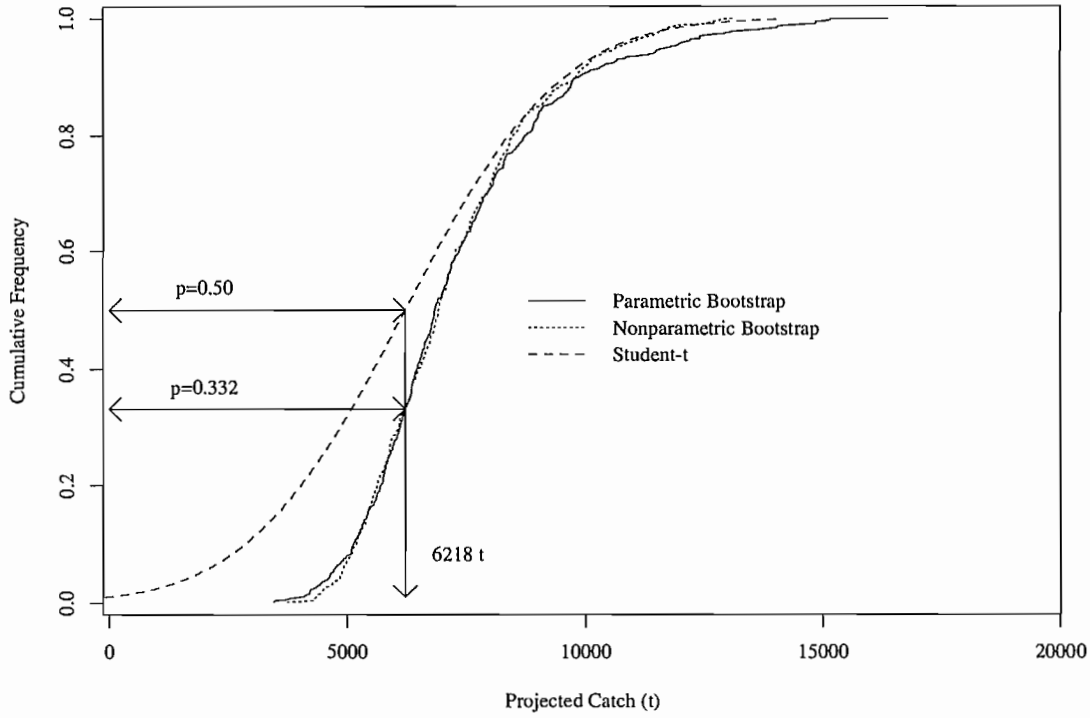


Fig. 3. Frequency percentiles for Projected Catch from the Parametric and Nonparametric bootstrapping. Student-t percentiles for projected catch with standard error equal to 2558 (Table 4) and degrees of freedom equal to 32, also presented for reference. Position of the ADAPT estimate of projected catch (6218 t) with respect to cumulative probability given for Bootstrap and Student-t curves.

suggest that the nonparametric bootstrap samples were somewhat more variable than those from the parametric bootstrap and this variation appears to have been carried over into the wider limits for nonparametric bootstrap confidence intervals for the projected catch. Restrepo et al. (1990) and Hoenig et al. (1990) both use the percentiles from $\widehat{G}(s)$ to characterize the distribution of their estimates from ADAPT. We do not know what the impact would be of applying **BC** percentile method to their data but in our case the differences were dramatic (Fig. 3, Table 5). The shape parameters for the **Perc** bootstrap intervals indicate that the associated frequency distributions of bootstrap projected catch were not symmetric and cast doubt on the appropriateness of the Student-t assumption for the case studied here. Use of the **BC** method resulted in confidence intervals which were more symmetric than **Perc** method and shorter than either the **Perc** method or the Student-t method.

In closing, we note that the problem studied here had an exceptionally short time series for an abundance index. Indeed, the six years available did not even allow us to follow a year-class through the eight age groups which were included in the analysis. Though such situations may occur, they are not common. Typically, abundance index time series span one or more decades. This problem was studied here because it was the only one available to us which was suitable for the nonparametric bootstrap experiment. Though we may not want to draw conclusions regarding the typical magnitude of bias or excess variance in fisheries assessment problems from this case study, it was useful for comparing the performance of alternative techniques. These techniques should not be viewed as competing methods. Rather, comparison of results from them provides insight regarding the validity of model and error assumptions and can ultimately guide us to improvements in fish stock assessments.

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Bootstrap Estimates of ADAPT Parameters, Their Projection in Risk Analysis and Their Retrospective Patterns

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The estimation of standing stock, fishing mortality and recruitment are the principal elements of biological advice for stock management. There is an increasing realization that the uncertainty associated with these estimates should also be provided. The ADAPT framework, which is commonly used in Canadian stock assessment, provides estimates of the parameter variation from a linear approximation near the solution. Using data from 4VsW cod, we compare these linear approximations with distributions from conditioned bootstrapping. As well as the estimated parameters, the bootstrapped standing stock and recruitment series are preserved for input into a risk analysis. Results for 4VsW cod show that the coefficients of variation for the calibration coefficients are overestimated by the linear approximation when compared to bootstrap results. Conversely, the CV's for F or numbers at age in the terminal year are underestimated. Risk analyses using the bootstrap distributions are compared to those derived with Monte Carlo inputs from normal distributions. Retrospective patterns for 4VsW (which shows the pattern in point estimates) and 4X (which does not) cod are investigated using bootstrapping. The mean of the bootstrap terminal year values is shown to remove the retrospective pattern for 4VsW cod. The retrospective problem is based on the observation that contemporaneous estimates of stock parameters do not agree with subsequent estimates for the same year as additional data are added to the same model. Furthermore, this disagreement displays a trend rather than a random pattern. The point estimates of fully recruited F for the 4VsW cod stock display a strong retrospective pattern. However, when the bootstrapped fully recruited F's are viewed retrospectively, the distributions are centered about a relatively stable mean which corresponds to the converged estimate after a number of years of data are accumulated.

L'estimation du stock actuel, de la mortalité par pêche et du recrutement sont les principaux facteurs biologiques pris en considération pour la gestion de stock. On commence toutefois à réaliser qu'il serait probant d'associer à ces facteurs leur valeur d'incertitude respective. Le cadre ADAPT, couramment utilisé pour l'évaluation de stock au Canada, offre des estimations de la variation de paramètre à partir d'une approximation linéaire à proximité de la solution. À l'aide des données de la morue 4VsW, nous comparons ces approximations linéaires aux distributions de méthode bootstrap conditionnée. Tout comme les paramètres estimés, les séries de données du stock actuel de la méthode bootstrap et du recrutement sont conservées afin d'être incorporées dans l'analyse de risque. Les résultats relatifs à la morue 4VsW révèlent que les coefficients de variation pour les coefficients d'étalonnage sont surestimés par l'approximation linéaire lorsqu'on les compare aux résultats de la méthode bootstrap. Inversement, les VC pour F ou la distribution selon l'âge en année terminale sont sous-estimés. Les analyses du risque à l'aide de la distribution bootstrap sont comparées à celles dérivées des données de la technique Monte-Carlo pour des distributions normales. Les répartitions rétrospectives avec les données de morue 4VsW (qui tracent la répartition par estimations ponctuelles) et de morue 4X (qui ne le font pas) sont étudiées à l'aide de la méthode bootstrap. La moyenne des valeurs bootstrap en année terminale contrecarre la répartition rétrospective de la morue 4VsW. Le problème rétrospectif repose sur l'affirmation selon laquelle les estimations modernes des paramètres de stock divergent des estimations subséquentes pour la même année lorsqu'on ajoute des données additionnelles au même modèle. De plus, la divergence semble être une tendance plutôt qu'une répartition aléatoire. Les estimations ponctuelles de F entièrement recruté du stock de morue 4VsW révèlent une importante répartition rétrospective. Toutefois, lorsque les F entièrement recrutés de la méthode bootstrap sont étudiées rétrospectivement, les distributions gravitent autour d'une moyenne relativement stable qui correspond à l'estimation convergée après plusieurs années de cueillette de données.

Bootstrap resampling will be used to investigate two issues in the provision of scientific advice for fisheries resource management. The first issue is the precision of that advice, or equivalently its uncertainty which is a key component of risk analysis. Risk analysis has been a topic of recent interest in the provision of scientific advice in fisheries management (e.g., NAFO Workshop 1991; Prospero Workshop 1992). Risk analysis may be thought of as being comprised of two components. The first is the propagation of uncertainties for a given course of action and the second is a metric of the consequences. We will focus on the estimation of uncertainties in catch projections. Metrification of the consequences of under- or over-fishing has proved to be elusive. The second to be investigated is the retrospective problem

(ICES 1991; Sinclair et al. 1990) which is a symptom of the lack of consistency in the estimation of stock abundance.

A previous report (Mohn 1991) used the diagnostics in ADAPT (Gavaris 1988) and analysis of underlying data to estimate confidence regions and sensitivities for 3 year constant catch strategies for NAFO Subarea (SA) 4VsW cod. This work was a Monte Carlo investigation of error propagation in stock projections. Hoenig et al. (1991) performed a conceptually similar, though more extensive, analysis but included the VPA process in their bootstrap analysis of uncertainty. A word about nomenclature. We will use Monte Carlo to mean replication of a procedure with data/parameters drawn from parametric distributions, however determined; although Efron (1982) would call this parametric bootstrapping. Herein, bootstrapping will be used to denote replication by sampling with replacement from observations. In the following, bootstrap methods are incorporated into the assessment procedure to compare the confidence limits of a projection with the Monte Carlo estimates previously reported. Because we are particularly interested in error propagation, it is the errors (residuals) in the nonlinear least squares (NLLS) fit of survey which are bootstrap sampled. This is known as conditioned bootstrapping.

Currently the ADAPT framework uses a version of the Marquardt-Levenberg algorithm to fit catch data and abundance indices to a model of exponential survivorship and catch which is proportional to effort and mean abundance. The Marquardt-Levenberg algorithm is known to have difficulties near the convergence point (Press et al. 1988). Also in the Marquardt as is a common practice in regression techniques, the goodness of fit is estimated by inverting an approximate Hessian matrix (with Marquardt parameter lambda set to zero) which provides an estimate of the covariance of the errors in the parameters. We will denote such estimates as linear approximation estimates. Press et al. (1988) have reported that the linear approximations are not robust to non-normal error distributions. In the following we compare the linear approximation estimates to those derived from bootstrapping.

After the bootstrap distribution of parameters has been completed, we explore the projection of the bootstrap distributions of standing stock, recruitment and selectivity under constant catch strategies for three years. A three year period is chosen because it is consistent with current management practice in Atlantic Canada. The state variables which we shall investigate are biomass, catch, fishing mortality (F) and profits. Profits are estimated from vessel performance data from 1985 (ICES 1986). A model suggested by P. MacGillivray (Dept. Fisheries and Oceans, P. O. 550 Halifax, N. S. B3J 2S7, pers. comm.) with a fixed cost and two variable costs is used. One variable cost contains the crew's share of operating expenses which is subtracted from the gross landed value before determining the labour costs. The other variable cost does not affect the labour cost.

The retrospective problem is based on the observation that contemporaneous estimates of stock parameters do not agree with subsequent estimates as additional data are added to the same model. Furthermore, this disagreement displays a trend rather than a random pattern. These biases may be seen in terms of either fishing mortality or population abundance estimates. Usually, but by no means always, the subsequent estimates of stock size for the same year are smaller; or equivalently the contemporaneous estimates are too large. This problem has been reported by Sinclair et al. (1990) and was a theme at the 1991 ICES Methods Working Group Meeting in St. John's (ICES 1991). The problem appears in a number of stocks and for a number of methods. However, the ICES results suggest that the problem is more prevalent in ADAPT than in the other methods they tested. The 4VsW cod stock displays such a strong retrospective pattern (Fanning and MacEachern 1991) the 1991 analyses were rejected. Fully recruited F estimates from the bootstrap replicate ADAPT runs were compiled to investigate their retrospective distribution and compare these to the point estimates.

Methods

Bootstrapping is the name applied to a broad range of computer intensive techniques which are used to estimate the statistical characteristics of a system. (See for example, Efron 1982). The type of bootstrapping used in this study is conditioned bootstrapping. The original abundance data (Y_{Obs}) are fit to a model in a base run of ADAPT (Y_{Base})

$$Y_{\text{Obs}} = Y_{\text{Base}} + \varepsilon \tag{1}$$

and the residuals (ε) are saved for resampling. Catch data are assumed to be error free and not included in the bootstrapping. Bootstrap data (Y_{Boot}) are then formed by sampling the residuals with replacement and adding to (Y_{Base}). The resampling is within a single age and survey and negative values, if any, are truncated to zero.

$$Y_{\text{Boot}} = Y_{\text{Base}} + \text{Resample}(\varepsilon) \tag{2}$$

The bootstrap abundance data are then re-analyzed using ADAPT. Similarly for the Monte Carlo replicates, random variation is introduced to Y_{Base} using a lognormal distribution $\left(\Lambda\left(\mu, \sigma^2\right)\right)$ parameterized by the coefficient of variation (CV).

$$Y_{\text{Monte Carlo}} = Y_{\text{Base}}\Lambda\left(0, \text{CV}^2\right) \tag{3}$$

The lognormal distribution was chosen for two reasons. Firstly, an examination of the residuals shows that they have long tails and the lognormal approximates this better than a normal distribution. Secondly, because the lognormal assumption is the most frequent choice in CAFSAC (Canadian Atlantic Fishery Advisory Committee) assessments, it is of more value to investigate than less common functions which might have fit the data better.

An age structured population, based on the 4VsW cod data (Fanning and MacEachern 1990, 1991), is the basis for this study. The catch at age, spring and summer surveys provide the inputs for our analysis. We have chosen ages 1 to 15 where the oldest age is not modelled as a plus group and the

21 year period from 1970 to 1990. A sequential population model is fit to these data using ADAPT with a model which has 19 parameters: F at ages 3–8 in 1990, 6 calibration coefficients (q's) for the summer survey for ages 3–8, and 7 q's for the spring survey for ages 3–9. The summer survey spans the 21 year catch data while the spring survey is from 1979 to 1990 with 1985 excluded. The objective function is the sum of squares of the log of the residuals to the surveys. After convergence which establishes the base run the projection bootstrapping phase begins. The residuals from the ADAPT fit for each survey are saved for bootstrap resampling. The resampling is by age and survey with replacement. The new data sets are analyzed with ADAPT and a summary of the results saved for further analysis. The recruits at age 1 and the standing stocks in 1990 are saved, as well as the average, maximum and fully recruited F's and the parameter values. The fully recruited F was the defined as the average of F's on ages 7, 8 and 9. This measure is adopted because the maximum over ages, a common definition of full recruitment, is relatively unstable.

For comparison, two types of projections are included. In either case, variation is added to the initial standing stock, recruitment, selectivity, natural mortality, weight at age, total catch and price per ton. The first type of projection, although called bootstrapping, is actually a hybrid of bootstrap and Monte Carlo techniques. Some Monte Carlo (parametric) simulation is required to include the effects of parameters which are not bootstrapped. The standing stock at the beginning of the projection, recruitment and selectivity are drawn, with replacement, from the results of 1000 ADAPT runs and hence we denote them as bootstrapped. The natural mortality, weight at age, TAC compliance and price per ton are determined parametrically from lognormal distributions, i.e., Monte Carlo. The projections are for three years with constant nominal TAC's. The modelled landings in each year are derived from a nominal TAC. To model compliance, variation is applied to the TAC from a distribution biased so that over-runs are likely to be of a greater magnitude than under-runs. This is done by halving the difference between under-runs and the nominal TAC as they are drawn from a lognormal distribution. For each projection the F's are determined iteratively; the iterations continue until projected landings and the target landings are matched within 100t or until the F exceeds 2.

The second type of projections is pure Monte Carlo with the recruitment, initial standing stock and selectivity now being replicates determined from lognormal distributions with CV's from the linear approximation and historical values. For more detail see Mohn (1991). Otherwise, for the other parameters the noise is lognormal. The noise is added before each replicate. Thus the natural mortality with noise added at each age remains unchanged throughout the 3 year projection. A series of test runs were performed with the noise added each year for the age specific parameters (natural mortality, selectivity, weights) but the results differed by only a few percent which does not justify the increased complexity. Typically 5000 replicates were performed.

Coefficients of variation are needed for the Monte Carlo projections and for the weight, mortality, TAC's and price per tonne both in the Monte Carlo and the bootstrap projections. The coefficient of variation for the recruitment series estimated in the base run is at 0.38 from which we set a value of 0.4 for use in the projections. This value compares well to the linear approximation CV's for F's for ages 3 through 8 in the base run (Table 1). Further, these values may be compared to the CV's given in Fanning and MacEachern (1990, Table 17a) for the parameters estimating numbers at age from their ADAPT run; for ages 4 to 9, their values are 0.44, 0.34, 0.31, 0.29, 0.38 and 0.38 respectively. Similarly, a value of 0.4 was assigned to the CV for initial numbers at age for the projections. Weights at age are assumed to be fairly easy to estimate and are assigned a CV of 0.1. Given the absence of any data, a CV of 0.2 has (conservatively) been assigned to the natural mortality. The CV's for the selectivities in the base run are large, particularly for younger animals. A fair proportion of this variation may be in response to year class strengths and changes in management regulations and markets over the 21 year data period. A relatively low CV of 0.1 is assigned for selectivity. Although this value is lower than might seem appropriate, it was chosen because higher values produced bizarre selectivity patterns which caused outliers in projections. The CV for the landings is set at 20% . This value was based on a comparison of the landings and TAC's for 4VsW cod (Fanning and MacEachern 1991) and 4X cod (Campana and Hamel 1991).

To summarize the overall scheme, in both 'bootstrap' and 'Monte Carlo' projections, the weight at age, natural mortality, price and total landings are resampled parametrically. In the bootstrap projections, the recruitment, initial population at age and selectivity are sampled from the bootstrapped ADAPT runs. For the Monte Carlo projections these three data items are drawn from lognormal distributions with coefficients of variation as described in the preceding paragraph.

In addition to the usual biological state variable, economic variables have been added to the projections. Of these economic variables, we only report the fleet profits which are based on the 1985 vessel performance report (ICES 1986). This report gives the average costs and earnings for a sample of vessels from various fleet sectors. We have chosen the sector containing trawlers of 45–64′ and separate the average costs into fixed and two variable costs. The variable costs are partitioned into those which are related to vessel operation (VC1 = \$378/day fishing) and are subtracted from the revenues (price times tonnage landed) before the crews labour cost is calculated. The remaining variable costs (VC2) are estimated at \$424/day fishing and the average vessel of those that responded to the survey fished for 73 days. The annual fixed costs are \$38000 per vessel. Once the landings are corrected for the crews share labour costs are defined to be 48% of the remainder. The average vessel performance was linked to a fictitious fleet of such vessels by relating the average catch and effort to the fishing mortality reported in the 4VsW cod assessment (Fanning and MacEachern 1991) and then the catch from a projection at the F (= 0.6) reported in the assessment for 1984. It was further assumed that a vessel could adjust its number of days fishing more easily than it

Table 1. Summary ADAPT and Bootstrap results for 4VsW cod. CV is the coefficient of variation and t is the Student's t statistic.

	Base Run[1]			Bootstrap[2]		
Age	Parameter	CV	t	Parameter	CV	t
F_{1990}						
3	0.00515	0.39	2.6	0.00544	0.52	1.9
4	0.07166	0.30	3.4	0.08118	0.51	2.0
5	0.18359	0.28	3.5	0.23935	0.69	1.4
6	0.23969	0.32	3.1	0.34421	0.83	1.2
7	0.19958	0.33	3.0	0.26583	0.78	1.3
8	0.29729	0.35	2.8	0.31548	0.35	2.8
q Summer Survey						
3	0.00016	0.12	8.2	0.00016	0.19	5.3
4	0.00017	0.12	8.0	0.00018	0.14	7.0
5	0.00015	0.14	7.3	0.00016	0.13	7.6
6	0.00013	0.16	6.1	0.00014	0.12	8.4
7	0.00013	0.21	4.8	0.00013	0.12	8.1
8	0.00009	0.35	2.8	0.00009	0.14	7.3
q Spring Survey						
3	0.00012	0.18	5.7	0.00013	0.36	2.8
4	0.00014	0.17	5.7	0.00015	0.26	3.9
5	0.00016	0.18	5.5	0.00016	0.20	5.0
6	0.00018	0.19	5.2	0.00019	0.18	5.4
7	0.00016	0.24	4.2	0.00017	0.22	4.5
8	0.00014	0.33	3.0	0.00015	0.19	5.2
9	0.00011	0.61	1.6	0.00011	0.21	4.8

[1] Denotes the linear approximation from base run ADAPT estimates.
[2] Denotes results of the 1000 bootstrap estimates.

could enter or leave the fishery. The total effort for the fleet in boat-days was partitioned (arbitrarily) such that the days per vessel would be twice as responsive as the number of active vessels. This assumption has only a minor effect of overall costs but is retained because of its intuitive appeal.

The main form of graphical representation for the risk analysis is a two dimensional distribution having biomass as one axis and fishing mortality, catch or fleet profits as the second. The appropriate state variables are aggregated into a rectangular grid to approximate a two dimensional probability distribution. Then the distributions are integrated and displayed as contours (e.g., Fig. 4). The distribution is integrated from the most probable cell to the least, however, instead of the usual cumulative distribution which integrates from the lower end of the range to the upper. See Mohn (1991) for details of the integration scheme. This has the affect of accumulating the cells so that the most probable states may be identified with a contour which approximates the confidence region. This technique has the advantage that the confidence region need not be continuous.

Results for the retrospective bootstrap analysis are based on 250 replicates and are given in tables and graphs. A single data set spanning the entire data period is generated using the conditioned bootstrap as described above. From that replicate five successive estimates are made using data from 1979–1986, then 1979–1987 etc. until the full data set used, 1979–1990. The tabular output has rows of average F over ages 7 through 9 estimates for each year's data set. Each row down the table contains the results when one more year's data is included. Therefore looking down a column for a given year will show the successive estimates of the average F as more years' data are added. The average F may be either a point estimate from the ADAPT base run or the mean of a number of bootstrap replicates. The graphical format is superimposed frequency distributions for subsequent estimates of a given year; the 1986 estimate of 1986 F's, the 1987 estimate, etc. They have been smoothed by a moving average filter with weights of 1, 2 and 1. The magnitude of these distributions is normalized to give constant areas.

ADAPT uses a nonlinear least squares, NLLS, algorithm (Marquardt-Levenberg) to estimate the model parameters. ADAPT provides estimates of the parameters, and from a linear approximation the coefficient of variation for each parameter. Also, the correlation matrix for the parameters is estimated. We shall compare the linearly estimated standard errors of the parameters to errors estimated by bootstrapping (Table 1). Also, the correlation matrix from the ADAPT estimation will be compared to the correlation of the parameter estimates from bootstrapping. In order to more clearly see the underlying patterns, the estimates of correlation are presented schematically in Table 2. A strong positive correlation is denoted by an asterisk, moderate positive by a plus sign, weak correlation by a period, moderate and strong negative

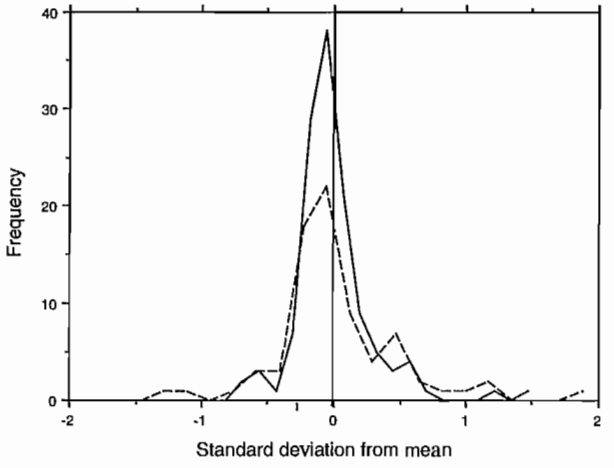


Fig. 1. Distributions of residuals for summer (solid line) and spring (dashed line) surveys. The residuals are normalized by subtracting the mean and dividing by the standard deviation for each survey and age.

respectively by a minus sign and an equals sign. Although they are both indices of interdependence among the parameters, these two correlation matrices are determined differently. The ADAPT matrix is the linear approximation derived from the approximate Hessian in the NLLS. The bootstrap correlation matrix is the correlation among the bootstrap replicates of the parameters.

Results

A base run of ADAPT was performed and the residuals of the two surveys saved for bootstrapping. The residuals are shown in Fig. 1. They have been normalized by subtracting the mean and dividing by the standard deviation. This was done to emphasize the pattern of the distributions. The frequency distribution for the spring survey is of a smaller magnitude because it contains 77 residuals as opposed to 126 in the summer survey distribution. Thus the model used for this data set fit 19 parameters with 203 observations. The distributions of the residuals are neither normal nor lognormal as determined by an analysis of their skewness and kurtosis. Table 1 contains the parameter estimates from this base run as well as their linearly approximated CV's and t values. Subsequently, 1000 bootstrap runs of ADAPT were carried out. For display purposes, the bootstrap parameters of these runs are averaged over groups; the F's the summer survey q's and the spring survey q's (Fig. 2). The shape of the distributions of biomass and F's from the bootstrap replicated ADAPT's were analyzed for normality and lognormality by examining the skewness (b_1) and kurtosis (b_2) of the distributions. The normal distribution has zero skewness and a kurtosis of 3. In the following table a single asterisk denotes significance at a 5% level and two denote less than 1%.

	Normal		Lognormal	
	b_1	b_2	b_1	b_2
Average F	1.69**	6.88**	0.65**	3.44**
Maximum F	2.87**	13.16**	1.20**	4.93**
Summer q	0.25*	3.06	0.04	2.93
Spring q	0.27**	2.96	-0.05	2.89
Biomass	0.71*	3.86**	0.08	2.76*
Recruits (1987)	1.17**	4.67**	0.01	2.75**

The average F is strongly non-normal and, although log transforming the data improved the skewness and kurtosis, F remains strongly non-lognormal. The distribution of average F is shown in Fig. 2a where a long tail to the right is observed. As expected, the maximum terminal F is more poorly behaved than the average. The summer and spring q's (Fig. 2b) are much more normally distributed. Figure 2 also shows the point estimate from the base run from the original data. Biomass and recruits are not parameters but rather state variables of interest. Log transforming the biomass data brings it closer to a normal distribution. The recruits in 1987 are neither normally nor lognormally distributed, but the log transformed data do not appear to exhibit a skewed distribution.

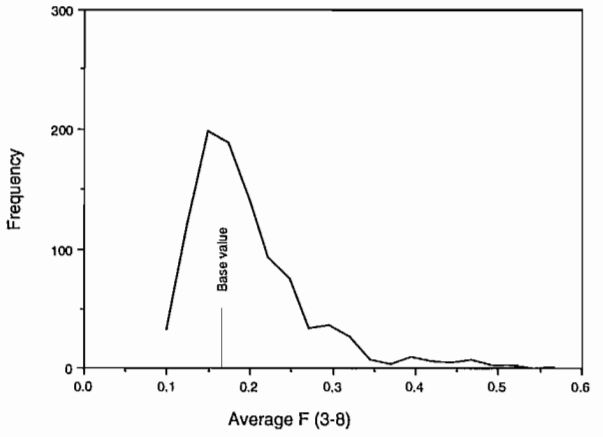
Table 1 contains parameter estimates from the base run and the mean parameters from the bootstrap experiment. The bootstrap F's are larger than those from the base run. The coefficients of variation for the bootstrap F's are considerably larger than those of the base run; in some cases approximately by a factor of 2. Figure 2a shows that the distribution of the average of the 6 F-parameters from the bootstrap runs is positively skewed. The q's display a better agreement between the base run and the average of the bootstraps. Also, the CV's are of a similar magnitude. One difference though is the pattern of the CV's. The CV's derived from the linear approximation show an increasing trend with age. The bootstrap CV's for the q's show a smaller but opposite trend. The distributions of the average q's are more symmetric than the average F's (Fig. 2b).

The estimates of correlation from the NLLS are presented schematically in the upper half of Table 2. Most of the correlations are negative. The correlations among the 19 parameters from the 1000 bootstrap assessments are given in the lower half Table 2. Most of the correlations are positive. Thus, a particular set of bootstrap residuals which causes a smaller than average population to be estimated implies that F will be higher (for the same catch) and the q's will be smaller because on average the survey values will be unchanged. As well as the sign differences between the correlations, there are also differences in pattern. For example, the age 8 F versus the summer survey in the linear approximation estimates is small. The same band in the bootstrap based correlations is strongly positive.

The 5000 bootstrap projections are summarized by both distributions and confidence regions. Figure 3 shows the distribution of biomass spreading for the three years of constant

Table 2. Correlation matrices from ADAPT and bootstrapped estimates. '=' denotes a correlation coefficient less than −0.15, '-' from −0.15 to −0.05, '.' from −0.05 to 0.05, '+' from 0.05 to 0.15 and '*' greater than 0.15.

	Age	F						q_{summer}						q_{spring}						
		3	4	5	6	7	8	3	4	5	6	7	8	3	4	5	6	7	8	9
ADAPT Base Run																				
F	3	.	.	-	-	-	=	+	-	-	-	-	-	*	-	-	-	-	-	-
	4	.	.	.	-	-	-	+	+	.	-	-	-	+	+	-	-	-	-	-
	5	-	.	.	-	-	-	.	.	.	-	-	-	+	+	+	-	-	-	-
	6	-	-	-	.	-	=	.	.	.	.	-	-	.	.	.	.	-	-	=
	7	-	-	-	-	.	-	.	.	.	.	.	-	.	.	.	.	.	-	=
	8	=	-	-	=	-	.	.	.	.	.	.	.	-	.	.	.	+	+	=
q_{summer}	3	+	+	.	.	.	.	.	.	.	.	.	-	.	.	.	.	-	-	-
	4	-	+	.	.	.	.	.	.	.	.	.	-	.	.	.	.	.	-	-
	5	-	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	-
	6	-	-	-	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	-
	7	-	-	-	-	.	.	.	.	.	.	.	.	-	.	.	.	.	.	-
	8	-	-	-	-	-	.	-	-	.	.	.	.	-	-	-	-	-	-	-
q_{spring}	3	*	+	+	.	.	-	.	.	.	.	-	-	.	.	.	-	-	-	-
	4	-	+	+	.	.	.	.	.	.	.	.	-	.	.	.	.	-	-	-
	5	-	-	+	.	.	.	.	.	.	.	.	-	.	.	.	.	.	-	-
	6	-	-	-	.	.	.	.	.	.	.	.	-	-	.	.	.	.	.	-
	7	-	-	-	-	.	+	-	.	.	.	.	-	-	-	.	.	.	.	-
	8	-	-	-	-	-	+	-	-	.	.	.	-	-	-	-	.	.	.	=
	9	-	-	-	=	=	=	-	-	-	-	-	-	-	-	-	-	-	=	.
Bootstrap																				
F	3	.	+	.	+	.	+	*	+	.	+	.	+	*	.	.	.	+	+	.
	4	+	.	.	.	.	+	+	*	.	+	.	+	+	*	.	.	.	+	+
	5	.	.	.	.	+	.	+	+	+	+	+	+	.	+	+	+	.	+	+
	6	+	.	.	.	.	+	+	+	+	*	+	*	+	.	*	*	+	*	+
	7	.	.	+	.	.	+	+	.	+	+	*	*	.	-	+	*	*	+	*
	8	+	+	.	+	+	.	*	*	*	*	*	*	.	+	+	*	*	*	*
q_{summer}	3	*	+	+	+	+	*	.	+	.	.	+	+	+	.	+	.	+	+	+
	4	+	*	+	+	.	*	+	.	+	+	+	+	.	+	+	.	.	+	+
	5	.	.	+	+	+	*	.	+	.	+	+	+	.	.	+	.	+	+	+
	6	+	+	+	*	+	*	.	+	+	.	+	+	.	.	+	+	+	*	+
	7	.	.	+	+	*	*	+	+	+	+	.	*	.	.	+	+	+	+	+
	8	+	+	+	*	*	*	+	+	+	+	*	.	.	.	+	+	+	*	+
q_{spring}	3	*	+	.	+	.	.	+	.	.	.	.	.	.	.	.	.	.	.	+
	4	.	*	+	.	-	+	.	+	.	.	.	.	.	.	.	.	.	.	.
	5	.	.	+	*	+	+	+	+	+	+	+	+	.	.	.	+	.	+	+
	6	.	.	+	*	*	*	.	.	.	+	+	+	.	.	+	.	+	+	+
	7	+	.	.	+	*	*	+	.	+	+	+	+	.	.	.	+	.	+	+
	8	+	+	+	*	+	*	+	+	+	*	+	*	.	.	+	+	+	.	+
	9	.	+	+	+	*	*	+	+	+	+	+	+	+	.	+	+	+	+	.



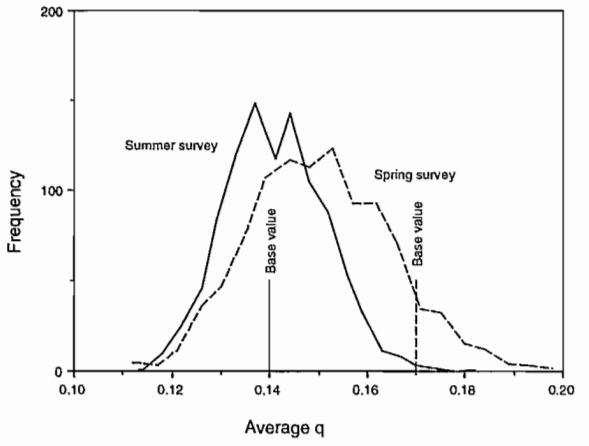


Fig. 2. Bootstrap distributions of the average F parameter (Fig. 2a) and average summer (solid line) and spring (dashed line) survey q parameters (Fig. 2b). The base run value associated with each parameter is also shown as a vertical line.

35 kt projections. Under this strategy the mean is slowly growing but the right hand margin is growing faster. Figure 4 shows the 50, 68 and 90% confidence regions from 45 kt annual TAC's after 3 years. The 68% line extends from approximately 220 to 460 kt with a small amount of this region located at the right margin of the plot. This is due to truncation of the distribution in preparation for integration and is artifactual.

The 5000 bootstrap projections are described by a state space of biomass, yield, average F and fleet profits. This space is projected onto two dimensional plots. Figure 5 shows the 50% confidence regions for yield and biomass for 4 levels of annual TAC's. The confidence regions are roughly elliptical for the 30, 45 and 60 kt TAC's with the principal axes parallel to the axes of the graph. The 60 kt TAC shows a broadening on its left hand side. The 75 kt distribution displays a sagging on the left hand side which indicates a correlation between biomass and yield. This correlation reflects the probability that the modelled fishery is becoming resource limited increases as the TAC rises.

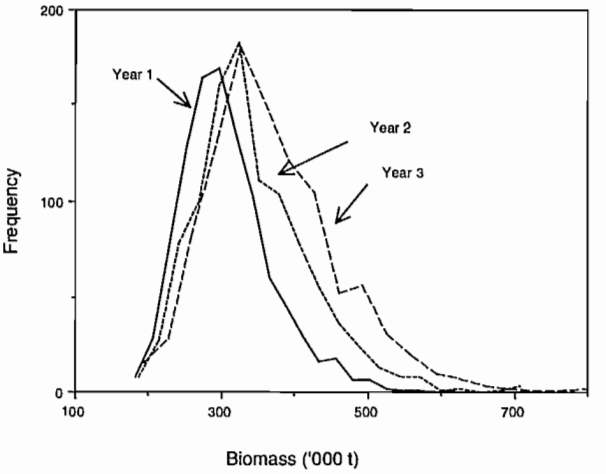


Fig. 3. Bootstrap distributions of total biomass in three successive years of a projection with a constant 35 kt TAC for 4VsW cod. This figure is derived from 500 replicates.

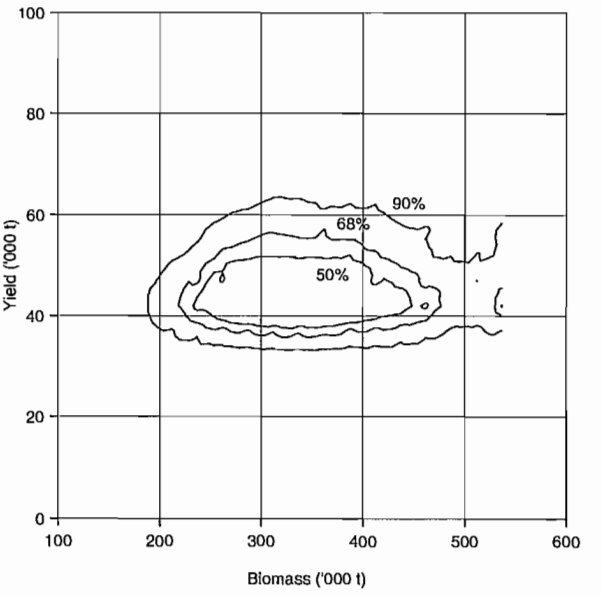


Fig. 4. Bootstrap confidence regions at 50, 68 and 90% levels for 4VsW after 3 years of a constant 45 kt TAC. This figure is derived from 500 replicates.

Figure 6 has the 50% confidence regions projected onto average F and biomass. In this figure the confidence region for the 60 kt TAC shows a splitting into two domains. This is because the fleet is limited to a maximum F of 2 in the projections and the catch is resource limited. The projection of the F's in Fig. 6 onto the biomass axis do not exactly correspond to the projection of the yields onto the biomass axes in Fig. 5 even though the biomass distribution is the same in both plots. This is because of the aggregation used in the integration and because the Y axes in the two plots have different distributions. The true marginals are identical but the coarseness of the aggregation in this mode of display aliases the details.

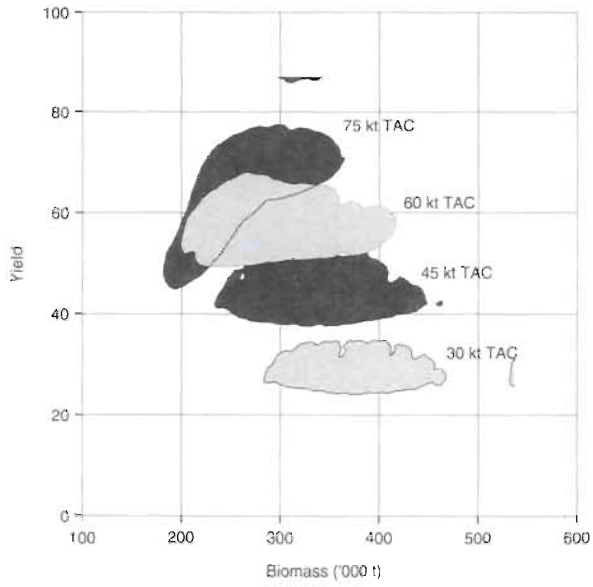


Fig. 5. 50% yield vs biomass confidence regions for 4VsW cod after 3 years of constant TAC strategies. The TAC's range from 30 to 75 kt. This figure is derived from 5000 replicates using bootstrap recruitment, initial population and selectivity.

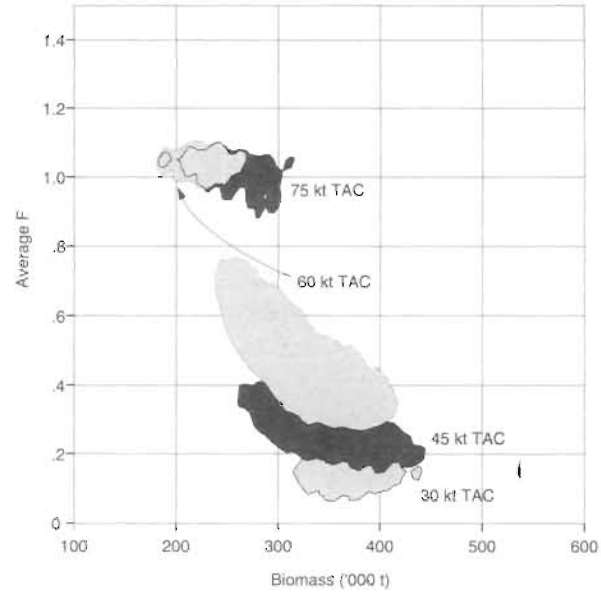


Fig. 6. 50% average F vs biomass confidence regions for 4VsW cod after 3 years of constant TAC strategies. The TAC's range from 30 to 75 kt. This figure is derived from 5000 replicates using bootstrap recruitment, initial population and selectivity.

Figure 7 has the same biomass axis as the previous two figures but the Y axis is now fleet profits. The confidence regions are quite broad and overlap considerably. The overlap is caused by the high relative sensitivity of profits to fluctuations in landings and effort. The heavy losses shown for the 75 kt TAC would not be expected because the fleet would change operations rather than lose money by fishing so hard with a diminished resource. We have assumed that the market would not respond with higher prices as landings are reduced because of the availability of cod from other fisheries.

Figure 8 is analogous to Fig. 5 except that the results are from the Monte Carlo projections instead of bootstrapping. The main difference seen is that the distributions do not extend so far to the right when Monte Carlo errors are used instead of bootstrapped. This suggests that the bootstrap distribution is more skewed than the lognormal distribution which is consistent with Fig. 1.

The equilibrium profits are shown in Fig. 9. The maximum profit occurs at an F of 0.25 and the profits drop to zero at an F of 0.95. These values may be compared to $F_{0.1}$ of 0.23 and F_{Max} of 0.46. When comparing this figure to Fig. 6, keep in mind that the F here is maximum F while it is average F in Fig. 6. The difference is approximately a factor of 2. Also, Fig. 9 is for an equilibrium fishery whereas Fig. 6 is from the estimated standing stock in 1991 which is not in equilibrium.

Table 3 contains the points estimates of the average (ages 7–9) F from the base run. The columns for 1985, 1986 and 1987 show the retrospective pattern. Reading down the columns as more data are added, the estimates of F get larger. The corresponding bootstrap means in Table 4 do not show the retrospective pattern seen in the point estimates. The means in Table 4 also are close to the point estimates after they have converged (after 3 or 4 years' data are available). The standard

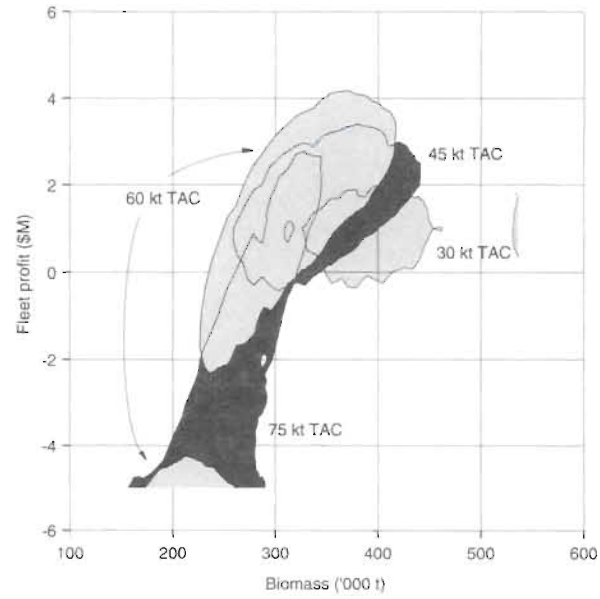


Fig. 7. 50% profit vs biomass confidence regions for 4VsW cod after 3 years of constant TAC strategies. The TAC's range from 30 to 75 kt. This figure is derived from 5000 replicates using bootstrap recruitment, initial population and selectivity.

deviations of the means are seen to decrease down columns in Table 5. The distributions of the successive estimates (1986 to 1989) of the 1986 average F are shown in Fig. 10. As more data are added, the distributions become tighter and the skewness decreases.

Data from the 4X cod fishery (Campana and Hamel 1991) do not show this strong pattern and for this reason they are

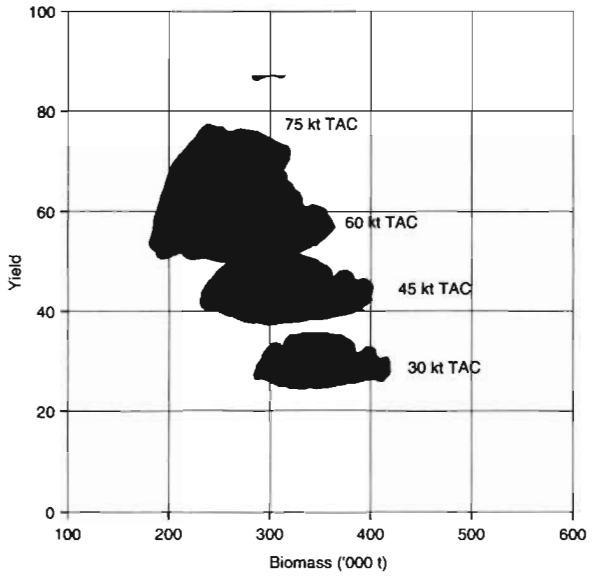


Fig. 8. 50% yield vs biomass confidence regions for 4VsW cod after 3 years of constant TAC strategies. The TAC's range from 30 to 75 kt. This figure is derived from 5000 replicates using Monte Carlo recruitment, initial population and selectivity.

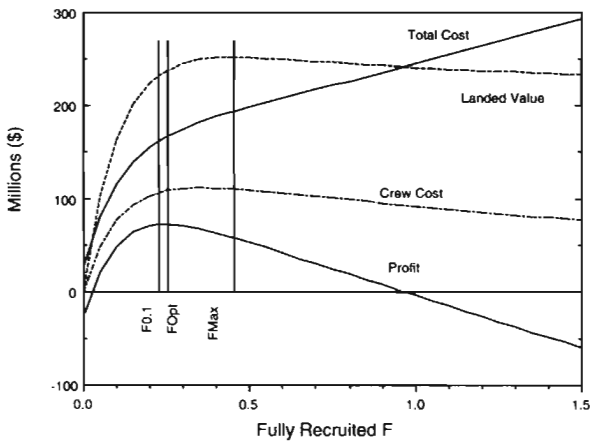


Fig. 9. Equilibrium fleet economic variables for 4VsW cod. Potential target fishing mortalities, $F_{0.1}$, F_{Opt} (point of maximum profits) and F_{Max} (point of maximum yield) are identified by vertical lines.

used to contrast the above results. The model fits terminal F to ages 3 through 7 and q's for the same ages for a single survey. The point estimates of the fully recruited (ages 7–9) F are shown in Table 6, which may be compared to Table 3. When data from this stock were bootstrapped and assessed by ADAPT the distributions of average F parameter and average q parameter were compiled and plotted in Figures 11a and b. The distribution of the estimated F's in Fig. 11a is quite different from that seen in Fig. 2a. In Fig. 11a the distribution shows a spike which is superimposed on a distribution which is skewed to the right. The distributions of the average q parameter (Fig. 11b) are not strongly skewed compared to the F's. The distributions of the retrospective estimates for

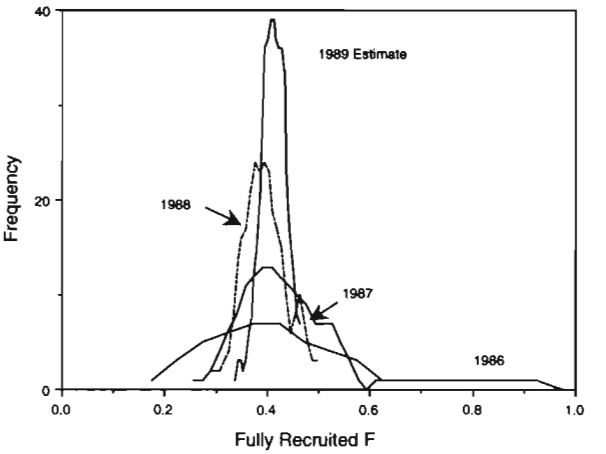


Fig. 10. Bootstrap distributions of successive (1986–1989) estimates of the 1986 fully recruited F for 4VsW cod.

F in 1986 are shown in Fig. 12. Although the distributions are wide for 1986 and 1987 estimates, they do not show the skewing to the right seen for 4VsW cod (Fig. 10).

Discussion

The comparison of errors between estimates derived from the inverted linear approximation to the Hessian in ADAPT and bootstrapped estimates show the former to be much more optimistic about the goodness of fit of F in the terminal year. If the bootstrap estimates are believed then we are less certain of terminal F and hence terminal stock size than previously estimated in ADAPT. The average bootstrap CV is about 60%. The linear approximation is known to be a poor estimator of confidence regions and covariance when measurement errors are not normally distributed (Press et al. 1988). Although assuming a logarithmic objective function helps reduce the discrepancy, the bootstrap estimates suggest that problems still exist, at least for these data.

The positive skewness of the bootstrap distribution of F parameters may be a warning of the retrospective problem. The point estimate would be expected to be near the mode of the distribution which is to the left of the relatively stable mean. The mean of the distributions in the 4VsW cod case does not move as additional years data are added. Rather, the distribution gets tighter and more symmetric. Thus, the retrospective problem can be removed by considering the means of the distributions rather than the point estimates. The distribution from 4X cod data shows a similar right-skewed distribution but has a spike superimposed upon it. This stock does not have a retrospective problem and one conjectures that the difference in the distributions may be related to this. It would be instructive to do a retrospective analysis with bootstrapping each year.

Because negative indices of abundance are not possible, any negative values in the bootstrap replicates were truncated to zero. As the bootstrap resampling is constrained within a given survey and age range the amount of truncation will

Table 3. Successive annual patterns of point estimates of average F's (ages 7–9) from base run from 1986 to 1990. Each row contains one more year's data using NAFO Subarea 4VsW cod data.

Terminal	Estimate Year											
Data Year	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990
1986	.188	.366	.354	.322	.235	.209	.251	.123				
1987	.200	.405	.383	.376	.257	.256	.327	.176	.147			
1988	.210	.429	.425	.423	.298	.287	.400	.257	.228	.209		
1989	.219	.446	.419	.424	.301	.340	.508	.366	.399	.418	.285	
1990	.222	.452	.430	.441	.312	.343	.515	.403	.485	.559	.446	.287

Table 4. Successive annual patterns of mean estimates of average F's (ages 7–9) from 250 bootstrap replicates using NAFO Subarea 4VsW cod data.

Terminal	Estimate Year									
Data Year	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990
1986	.437	.448	.355	.366	.590	.496				
1987	.439	.457	.343	.363	.556	.437	.549			
1988	.448	.463	.343	.348	.530	.400	.467	.588		
1989	.432	.444	.322	.358	.547	.414	.489	.590	.510	
1990	.432	.444	.316	.347	.523	.410	.499	.591	.489	.345

be less frequent and less severe than if residuals were resampled from an entire survey. In a test run of 500 replicates, truncation increased the mean of the 203 points data used in tuning by 0.7% and decreased its variance by 0.4% on average. Therefore the effects of the truncation are not expected to be an important factor.

The economic dimension was added to demonstrate that although biological assessment and prediction are quite uncertain, the variances related to economics are even worse. The economic model is quite simple and would benefit from further development. The uncertainty shown by fleet profits are so large that they would not be expected to be changed greatly by model refinement. The equilibrium yield results show that the maximum fleet profits ($F = 0.25$) are quite close to $F_{0.1}$ but that the break-even F is nearly 4 times higher. This suggests that economic pressures would be pulling the fishery towards economic equilibrium and away from an $F_{0.1}$ strategy, thereby creating a certain tension. We have modelled constant catch strategies for three years which well approximates current practice in Atlantic Canada. As Mohn (1990) reported, constant catch strategies tend to have more stable profits than constant F strategies. Therefore, we would expect that uncertainties, related to fleet profits under a constant F strategy, would be very large (Fig. 7).

The results are for demonstration of the bootstrap method and are not meant to constitute an assessment of a stock. The estimates of standing stock and hence projections are considerably larger than the values in the most recent assessment. This is because the base run does not agree with that used in Fanning and MacEachern (1991). Also, the bootstrap runs used different assumptions about recruitment than those of Fanning and MacEachern. Until these, or similar, techniques are applied to other stocks one cannot say to what degree the patterns and conclusions above are typical.

The distributions of the F-related parameters for 4VsW cod (Fig. 2a) and 4X cod (Fig. 11a) show dissimilar patterns. The former is unimodal and skewed to the right while the latter is bimodal with a spike superimposed on a right skewed distribution. The ranges are broad which reflects the difficulty in determining F, or stock size, in the terminal year.

The distributions for respective average q parameters (Figures 2b and 11b) are tighter than the F parameters and show

Table 5. Successive annual patterns of standard deviations of average F's (ages 7–9) from 250 bootstrap replicates using NAFO Subarea 4VsW cod data.

Terminal	Estimate Year									
Data Year	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990
1986	.015	.025	.026	.039	.127	.217				
1987	.009	.013	.015	.020	.051	.078	.175			
1988	.005	.009	.011	.015	.034	.043	.080	.221		
1989	.007	.010	.011	.008	.019	.024	.048	.116	.195	
1990	.004	.007	.009	.008	.016	.015	.026	.059	.087	.127

Table 6. Successive annual patterns of point estimates of average F's (ages 7–9) from base run from 1986 to 1990 for NAFO Subarea 4X cod data.

Terminal Data Year	Estimate Year									
	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990
1986	.473	.504	.643	.401	.370	.258				
1987	.494	.542	.692	.431	.402	.325	.452			
1988	.491	.520	.659	.403	.398	.306	.389	.344		
1989	.501	.542	.709	.472	.519	.467	.709	.675	.532	
1990	.500	.539	.703	.475	.505	.451	.624	.631	.355	.327

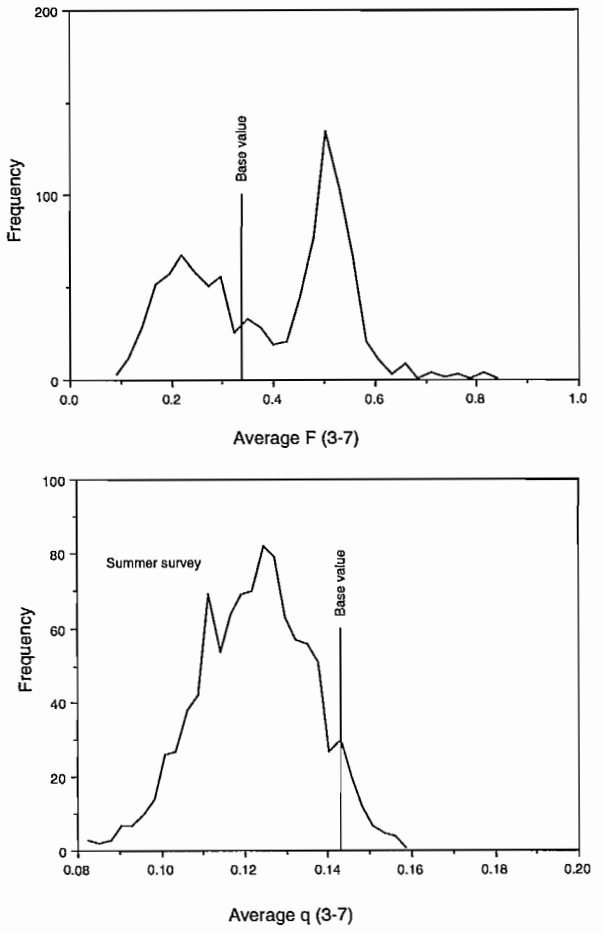


Fig. 11. Bootstrap distributions of the average F parameter (Fig. 11a) and average summer survey q parameter (Fig. 11b). The base run average associated with each distribution is also shown as a vertical line.

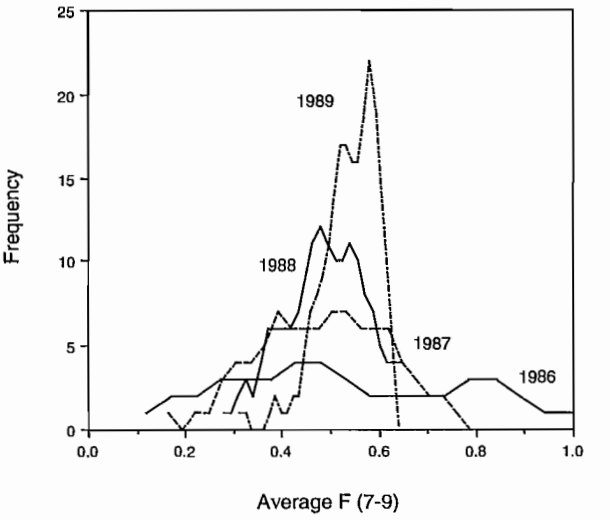


Fig. 12. Bootstrap distributions of successive (1986–1989) estimates of the 1986 fully recruited F for 4X cod.

much less skewing. This observation may explain the stability of the Laurec-Shepherd method (Laurec and Shepherd 1983) compared to ADAPT. Laurec-Shepherd does not attempt to directly estimate the stock in the terminal year directly but rather infers it from q estimates.

In the retrospective study, we compare the 4VsW cod stock, which has a retrospective pattern in the point estimates of average F, to 4X which does not. For either stock, the contemporaneous estimate is very broad, ranging from an F of 0.1 to greater than 1. Such a wide distribution suggests that it would be difficult with these data to directly estimate the terminal F, which is equivalent to estimating the terminal stock size. The difference between the two stocks is that the 4X distributions are relatively more symmetric. This suggests that the pattern of F's may be used as a diagnostic of the susceptibility to a retrospective problem. Fortunately, in the 4VsW case the successive distributions are nested so that the means are well behaved. The stability of the means of the successive bootstrap estimates suggests that their estimation may be a solution for the retrospective problem, at least for stocks that behave like 4VsW cod. There has been debate for stocks having the retrospective pattern as to whether the contemporaneous estimate or the estimate after an number of years was the true value. The observation that the bootstrap means correspond well with the point estimates after several years of data and the low precision of the contemporaneous estimates favours the estimate after a number of years.

A more complete analysis would begin with bootstrapping from the survey samples themselves and carry through to projections. See for example Smith and Gavaris (1993). We chose a less ambitious strategy and began with the aged surveys and bootstrapped the residuals from a base run. The cost/benefits of choosing a conditioned approach are not

known. A study comparing Monte Carlo, conditioned and unconditioned bootstrap parameter estimation for ADAPT would represent a valuable contribution to our understanding of the assessment process. Finally, we have advocated the use of the mean of the bootstrap estimate instead of the point estimate to remove the retrospective pattern. This practice is contrary to that usually suggested for bootstrapping as it suggests that the bootstrap mean is a better estimate of the position of underlying distribution than the point estimate. The bias correction implied from the results of the bootstrap does not correct for the retrospective pattern and is in fact in the opposite direction. Noting that the mean of the bootstrap estimates removed the retrospective pattern in the case studied, it is suggested that other cases where strong retrospective patterns occur be investigated to identify the universality of this approach for removing retrospective patterns.

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Analytical Estimates of Reliability for the Projected Yield from Commercial Fisheries

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The decision making process for fishery resource management would be enhanced by the provision of information regarding the reliability of assessment results. A common product of marine fish stock assessments is the projected yield corresponding to a target fishing mortality rate. The stock assessment problem involves non-linear models and derived statistics are based on linear approximations. The precision and bias of the estimate of projected yield were derived directly by reparameterizing the model formulation of ADAPT to include this quantity as an explicit parameter. These statistics were alternatively derived as functions of the parameters from a more typical model formulation. The results were virtually identical, as they should be. The analytical estimates were also compared to Monte Carlo simulation results to examine the effect of non-linearity on the analytical approximations. The study suggest that for typical stock assessment problems, the analytical estimators can provide a useful summary if the model and error assumptions are appropriate.

Le processus de prise de décisions pour la gestion des ressources de pêche pourrait être enrichi par des données relatives à la fiabilité des résultats d'évaluation. Un des paramètres qui apparaît souvent dans l'évaluation du stock de poissons en eaux salées est le rendement prévu, qui correspond à un taux cible de mortalité par pêche. Le problème d'évaluation des stocks titent de l'incompatibilité des modèles non linéaires avec les statistiques dérivées d'approximations linéaires. La précision et le biais de l'estimation du rendement prévu ont été obtenus directement en reparamétrant la formulation de modèle ADAPT afin d'inclure cette valeur à titre de paramètre explicite. Les statistiques ont ensuite été dérivées des paramètres d'une formulation de modèle plus typique. Comme prévu, les résultats obtenus étaient presque identiques. Les estimations analytiques ont aussi été comparées aux résultats de simulation de Monte-Carlo afin d'examiner l'incidence de la non-linéarité sur les approximations analytiques. L'étude suggère que, en ce qui a trait aux problèmes d'évaluation de stock typique, les estimateurs analytiques peuvent offrir un sommaire utile si le modèle et les suppositions d'erreurs sont pertinents.

In Atlantic Canada, biological advice is one of the principle elements upon which the management of marine fishery resources is founded. Results from stock (population) assessments form the basis of the scientific advice. Decision making is facilitated and enhanced by an appreciation of the reliability of the assessment results. Ideally, the uncertainty surrounding parameters would be encapsulated by probability distributions of their estimators and these would be integrated with objectives, costs and risks in a more formal statistical treatment of the decision problem (Berger 1980; Raiffa and Schlaifer 1961).

The Canadian Atlantic Fishery Scientific Advisory Committee (CAFSAC) has typically provided advice for marine fisheries in the form of projected yield corresponding to a target fishing mortality rate (Anon 1991). The reliability of the projected yield has not been derived routinely. In the absence of this information, a formal statistical decision making process has not evolved and subjective judgment regarding the uncertainties has been employed to qualify advice. Building consensus on subjective interpretations is a difficult process. Some progress towards an informed decision making process can be achieved through knowledge of the precision and bias of the projected yield.

Many analytical assessments reviewed by CAFSAC currently employ the adaptive framework, ADAPT, (Gavaris 1988; Gavaris 1991). ADAPT is not associated with a specific model. It is a foundation which provides for expression of the stock assessment problem in a common statistical estimation framework based on least squares theory. ADAPT allows sufficient flexibility in the definition of model relationships and structures to accommodate diverse stock assessment situations. By virtue of being based on common non-linear least squares, ADAPT can be used to exploit available techniques to examine the statistical properties of estimators. This paper examines two approaches for employing analytical methods based on linear approximation to derive estimates of precision and bias for projected yield. One approach is based on development of a formulation for ADAPT which reparameterizes the assessment problem to explicitly include the projected yield as a parameter and thereby obtain direct measures of its precision and bias. Alternatively, the projected yield and its precision and bias are derived as functions of the statistical results from the more typical formulation which uses population abundance in the terminal year as parameters.

The sampling distribution and derived statistical properties of estimators can also be studied by replication (Bard

1974; Ratkowsky 1983; Seber and Wild 1989). Since it is generally not practical to replicate the physical system, the replication method is often carried out by computer simulation. The two analytical approaches are compared and their results are evaluated against those of a Monte Carlo simulation study where the model assumptions are satisfied.

Methods

Data

Data for eastern Georges Bank haddock (Gavaris and Van Eeckhaute 1991) were used for these analyses. The Georges Bank haddock fishery is characteristic of the stock assessment problems encountered for groundfish in the Northwest Atlantic. The primary gear used is the otter trawl but there is a significant longline component. Catches are taken year round though there are area closures during the spring spawning season. Haddock are relatively long lived and abundant year-classes contribute significantly to the commercial catch through age 8. Catch numbers at age were available from 1969 to 1990 for age groups 1 to 8. Annual stratified random bottom trawl surveys have been conducted by the Department of Fisheries and Oceans, Canada, during the spring since 1986 and by the National Marine Fisheries Service, USA, during the spring since 1968 and during the fall since 1963. In these analyses, spring survey results were compared to the beginning-of- year age specific population numbers, while the fall survey results were compared to the age specific population numbers for fish one year older at the beginning of the subsequent year. The following notation was used for the data where a indexes age and y indexes year:

$C_{a,y}$	=	catch
$I_{1,a,y}$	=	USA fall survey
$I_{2,a,y}$	=	USA spring survey
$I_{3,a,y}$	=	Canadian spring survey

Model

Groundfish stocks in Atlantic Canada are typically managed by regulating the amount caught by the fishery to correspond with a target fishing mortality rate determined from consideration of population dynamics. The typical situation encountered requires that the target yield be projected to the second year following the terminal year for which the catch at age statistics are available. Though the target yield may be projected for other years, the analyses are demonstrated for this scenario.

The projected yield is a function of the state of the stock and not a forecast in the ordinary sense. It can be regarded as the translation of the estimated current state of the stock to a quantity which can be used to regulate the fishery. Accordingly, we desire estimators of the mean projected yield and its associated precision and bias.

Two cases were considered; a model using all three available survey indices as was done in the assessment by Gavaris and Van Eeckhaute (1991) and a model using only the Canadian spring survey. The latter was conducted for comparison with bootstrap results from this same data in Smith and Gavaris (1993). The model formulations were the same for the two cases considered excepting that the objective function being minimized reflected those surveys included.

It was assumed that natural mortality, M, was known and equal to 0.2. The errors in the survey data, after taking natural logarithms, were assumed to be independent, identically distributed and with constant variance. It has been recognized that the errors in the survey data for a given year are unlikely to be independent among ages. This assumption has been made however in the interest of simplicity and parsimony. Though further study of this issue is warranted, common practice is followed here. The bias estimator also requires that these errors are normally distributed. The errors in the catch at age were assumed negligible relative to the errors in the survey data. Though the fishing mortality rate for the projected years is considered a target which is controllable, often the exploitation pattern by age group is estimated as an average from the fishery. Similarly, the average weight at age for the projected years is derived as an average over some recent time period. For the projected years, errors in the exploitation pattern as well as errors in the average weight at age were also considered negligible.

The derivatives required by the analytical methods and the minimization algorithm were approximated by finite differences. Computational efficiency was improved by taking advantage of the conditionally linear nature of the calibration coefficients. These parameters can be removed from the nonlinear search and computed analytically at each step from the remaining parameters (Bates and Watts 1988, p. 85; Seber and Wild 1989 p. 654). They are considered explicit parameters in the computation of statistical properties.

Projected Yield Formulation

This approach employs a model formulation which includes projected yield as an explicit parameter. Letting t represent the terminal year for which catch at age statistics are available, define the model parameters

ψ_{t+2}	=	projected yield,
$\pi_{a,t+2}$	=	proportion of projected yield at age.

for $a = 1$ to 7, where the proportion at age 8 is defined by the constraint for the proportions at age to sum to 1,

$$\pi_{8,t+2} = 1 - \sum_{a=1}^{7} \pi_{a,t+2}$$

and

$\kappa_{1,a}$	=	calibration constants for USA fall survey

$\kappa_{2,a}$ = calibration constants for USA spring survey

$\kappa_{3,a}$ = calibration constants for Canadian spring survey

for $a = 1$ to 8

ADAPT, which incorporates a Levenberg-Marquardt algorithm (Bard 1974) to obtain least squares estimates, was used to solve for the parameters by minimizing the objective function

$$Q(\hat{\psi}, \hat{\pi}, \hat{\kappa}) = \sum_{s,a,y} \left(q_{s,a,y}(\hat{\psi}, \hat{\pi}, \hat{\kappa})\right)^2 = \sum_{s,a,y} \left(\ln I_{s,a,y} - \ln \hat{\kappa}_{s,a} N_{a,y}(\hat{\psi}, \hat{\pi})\right)^2$$

where s indexes survey and $N_{a,y}(\hat{\psi}, \hat{\pi})$ is the estimate of the population abundance at age. The objective function being minimized defines a linear relationship through the origin between the survey abundance and population abundance for each age group, with slope equal to an age specific calibration constant. Equivalently, the function can be viewed as defining a linear relationship between log survey abundance and log population abundance with a constant slope set to 1 and the intercept equal to the log of an age specific calibration constant.

Established catch equations for sequential population analysis (Rivard 1982) were used to compute the population abundance as follows, where to avoid confusion, $N_{a,y}(\hat{\psi}, \hat{\pi})$ is abbreviated to $N_{a,y}$.

The yield at age was computed as the product of the projected yield and the proportion at age,

$$(1) \qquad Y_{a,t+2} = \psi_{t+2} \pi_{a,t+2}$$

the catch numbers at age were derived by dividing the yield at age by the average weight at age, W_a,

$$(2) \qquad C_{a,t+2} = Y_{a,t+2} / W_a$$

and the population abundance at the beginning of the year was derived using the common catch equation,

$$(3) \qquad N_{a,t+2} = \frac{C_{a,t+2}\left(F_{a,t+2} + M\right)}{F_{a,t+2}\left(1 - \exp\left[-\left(F_{a,t+2} + M\right)\right]\right)}$$

where F designates the fishing mortality rate.

The population abundance at the beginning of the previous year was obtained from the exponential decay model which underlies the catch equation,

$$(4) \qquad N_{a,t+1} = N_{a+1,t+2} \exp[F_{a,t+1} + M]$$

The target fishing mortality rate, $F_{a,y}$ in both years, $y = t + 1$ and $y = t + 2$ for $a = 1, \ldots, 8$, was set at:

Age	1	2	3 to 8
$F_{a,y}$	0	0.125	0.25

For all other years, $y = 1$ to t, the population abundance was computed sequentially using the cohort approximation (Pope 1972),

$$(5) \qquad N_{a,y} = N_{a+1,y+1} \exp[M] + C_{a,y} \exp[M/2]$$

for ages 1 to 7 and the catch equation,

$$(6) \qquad N_{8,y} = \frac{C_{8,y}\left(F_{8,y} + M\right)}{F_{8,y}\left(1 - \exp\left[-\left(F_{8,y} + M\right)\right]\right)}$$

for age 8, where the fishing mortality rate for age 8 was assumed equal to a self-weighted average for ages 4 to 7,

$$(7) \qquad F_{8,y} = \ln\left(\frac{\sum_{a=4}^{6} N_{a,y}}{\sum_{a=5}^{7} N_{a,y+1}}\right) - M$$

A common estimator of the covariance matrix of the parameters based on a linear approximation was employed (Kennedy and Gentle 1980 p. 476).

$$(8) \qquad \mathrm{Cov}(\hat{\psi}, \hat{\pi}, \hat{\kappa}) = \hat{\sigma}^2 \left[\boldsymbol{J}^T(\hat{\psi}, \hat{\pi}, \hat{\kappa}) \boldsymbol{J}(\hat{\psi}, \hat{\pi}, \hat{\kappa})\right]^{-1}$$

where $\hat{\sigma}^2$ is the mean square residual and $\boldsymbol{J}(\hat{\psi}, \hat{\pi}, \hat{\kappa})$ is the Jacobian matrix (first derivatives with respect to parameters) of $\boldsymbol{q}(\hat{\psi}, \hat{\pi}, \hat{\kappa})$

$$(9) \qquad \boldsymbol{J}(\hat{\psi}, \hat{\pi}, \hat{\kappa}) = \partial \boldsymbol{q}(\hat{\psi}, \hat{\pi}, \hat{\kappa}) / \partial(\hat{\psi}, \hat{\pi}, \hat{\kappa})$$

where $\boldsymbol{q}(\hat{\psi}, \hat{\pi}, \hat{\kappa})$ is a vector with elements $q_{s,a,y}(\hat{\psi}, \hat{\pi}, \hat{\kappa})$. The superscript ^{T} denotes transpose.

The method of Box (1971), which is also based on a linear approximation and in addition assumes that the errors are normally distributed, was used to estimate the bias of parameters.

$$(10) \qquad \mathrm{Bias}(\hat{\psi}, \hat{\pi}, \hat{\kappa}) = \frac{-\hat{\sigma}^2}{2} \times \left(\sum_{s,a,y} \boldsymbol{J}_{s,a,y}(\hat{\psi}, \hat{\pi}, \hat{\kappa}) \boldsymbol{J}^T_{s,a,y}(\hat{\psi}, \hat{\pi}, \hat{\kappa})\right)^{-1} \times \sum_{s,a,y} \boldsymbol{J}_{s,a,y}(\hat{\psi}, \hat{\pi}, \hat{\kappa}) \times \mathbf{tr}\left[\left(\sum_{s,a,y} \boldsymbol{J}_{s,a,y}(\hat{\psi}, \hat{\pi}, \hat{\kappa}) \boldsymbol{J}^T_{s,a,y}(\hat{\psi}, \hat{\pi}, \hat{\kappa})\right)^{-1} \times \boldsymbol{H}_{s,a,y}(\hat{\psi}, \hat{\pi}, \hat{\kappa})\right]$$

where $\boldsymbol{J}_{s,a,y}(\hat{\psi}, \hat{\pi}, \hat{\kappa})$ are vectors of the first derivatives for each $q_{s,a,y}(\hat{\psi}, \hat{\pi}, \hat{\kappa})$ (these are rows of the Jacobian matrix defined above) and $\boldsymbol{H}_{s,a,y}(\hat{\psi}, \hat{\pi}, \hat{\kappa})$ are the Hessian matrices (second derivatives with respect to parameters) for each $q_{s,a,y}(\hat{\psi}, \hat{\pi}, \hat{\kappa})$.

$$(11) \qquad \boldsymbol{H}_{s,a,y}(\hat{\psi}, \hat{\pi}, \hat{\kappa}) = \frac{\partial^2 q_{s,a,y}(\hat{\psi}, \hat{\pi}, \hat{\kappa})}{\partial(\hat{\psi}, \hat{\pi}, \hat{\kappa}) \partial(\hat{\psi}, \hat{\pi}, \hat{\kappa})}$$

The expression **tr** represents the trace (sum of major diagonal) operator.

Function of Parameters Method

A more typical model formulation using population abundance at the end of the terminal year (beginning of year $y = t + 1$) as parameters was considered. The projected yield can then be derived as a function of the estimated parameters. Natural log population abundance was used instead however, because this parameterization displayed a more "close to linear" behaviour as will be discussed later. Define the model parameters

$\phi_{a,t+1}$ = ln population abundance at age

and the $\kappa_{1,a}$, $\kappa_{2,a}$ and $\kappa_{3,a}$ are calibration constants as before for $a = 1$ to 8.

As above, ADAPT was used to solve for the parameters by minimizing the objective function

$$Q(\hat{\phi}, \hat{\kappa}) = \sum_{s,a,y} \left(q_{s,a,y}(\hat{\phi}, \hat{\kappa})\right)^2 = \sum_{s,a,y} \left(\ln I_{s,a,y} - \ln \hat{\kappa}_{s,a} N_{a,y}(\phi)\right)^2$$

Again, to avoid confusion, $N_{a,y}(\hat{\phi})$ is abbreviated by $N_{a,y}$. For year $y = t + 1$, the population abundances are derived from the parameter estimates,

$$N_{a,t+1} = \exp\left[\hat{\phi}_{a,t+1}\right] \tag{12}$$

For all other years, $y = 1$ to t, the population abundance was computed using equations 5–7. The approximations described above, equations 8–11 (replacing $\hat{\psi}, \hat{\pi}, \hat{\kappa}$ with $\hat{\phi}, \hat{\kappa}$), were used to obtain estimates of the covariance and bias of the estimated parameters, $\hat{\phi}$ and $\hat{\kappa}$.

To derive the projected yield for the target year, $y = t + 2$, the target fishing mortality rate at age and weight at age, as identified above, were used in the following calculations.

The population abundance at the beginning of year $y = t + 2$ is obtained from the exponential decay model,

$$N_{a,t+1} = N_{a+1,t+2} \exp\left[-\left(F_{a,t+1} + M\right)\right] \tag{13}$$

The catch numbers at age and projected yield in year $y = t + 2$ are derived using the catch equation and then applying the weight at age,

$$C_{a,t+2} = F_{a,t+2} N_{a,t+2} \times \frac{\left(1 - \exp\left[-\left(F_{a,t+2} + M\right)\right]\right)}{\left(F_{a,t+2} + M\right)} \tag{14}$$

$$\psi_{t+2} = \sum_a C_{a,t+2} W_a \tag{15}$$

It is seen from these calculations that the projected yield, ψ_y, is a function of the estimated parameters from a more typical formulation. Let $\psi_y = g(\hat{\phi}, \hat{\kappa})$ denote that transforming function. Estimates of the variance and bias of the projected yield can be derived using the methods described in Ratkowsky (1983).

$$\text{Var}(\psi) = \mathbf{tr}\left[\left(\boldsymbol{G}\boldsymbol{G}^T\right) \text{cov}\left(\hat{\phi}, \hat{\kappa}\right)\right] \tag{16}$$

$$\text{Bias}(\psi) = \boldsymbol{G}^T \text{Bias}\left(\hat{\phi}, \hat{\kappa}\right) + 1/2\mathbf{tr}\left[\boldsymbol{W}\text{cov}\left(\hat{\phi}, \hat{\kappa}\right)\right] \tag{17}$$

where $\boldsymbol{G}$ is the vector of first derivatives of $g()$ with respect to the parameters

$$\boldsymbol{G} = \partial g\left(\hat{\phi}, \hat{\kappa}\right) / \partial\left(\hat{\phi}, \hat{\kappa}\right) \tag{18}$$

and $\boldsymbol{W}$ is the matrix of second derivatives of $g()$ with respect to the parameters

$$\boldsymbol{W} = \partial^2 g\left(\hat{\phi}, \hat{\kappa}\right) / \partial(\hat{\phi}, \hat{\kappa}) \partial(\hat{\phi}, \hat{\kappa}) \tag{19}$$

Monte Carlo Simulation

A Monte Carlo simulation employing the projected yield formulation was used to study the statistical properties of the estimator for projected yield. Following common practice (Bard 1974, p. 46; Ratkowsky 1983, p. 23), the true values of the parameters ψ, π, κ and of the variance σ^2, used to generate replicates of $I_{s,a,y}$ were assumed equal to the estimates (i.e., $\hat{\psi}, \hat{\pi}, \hat{\kappa}$) obtained from the case studies. First, the "true" population abundance $N_{a,y}$, was computed with equations 1–7 using the observed catch at age, the target fishing mortality rate and the average weight at age for the projected years with natural mortality assumed to be equal to 0.2. Then the "true" ln survey abundance was computed as

$$\mu_{s,a,y} = \ln \kappa_{s,a} N_{a,y} \tag{20}$$

Pseudo-random survey data were generated by

$$_rI_{s,a,y} = \exp\left[\mu_{s,a,y} + \sigma\, {}_rZ_{a,y}\right] \tag{21}$$

where r indexes replicates and ${}_rZ_{a,y}$ is a pseudo-random number from a standard normal distribution, N(0,1). This corresponds to the assumptions made in estimation, that the errors in the survey abundance indices are independent and identically distributed with constant variance after taking natural logarithms. The projected yield formulation was applied to each of 500 replicate survey data sets, generating 500 estimates of projected yield.

The 500 estimates describe the sampling distribution of the estimator of projected yield. An estimate of the mean of the sampling distribution is

$$\overline{\psi} = \sum_r {}_r\psi / 500 \tag{22}$$

and its variance

$$\text{Var} = \sum_r \left({}_r\psi - \overline{\psi}\right)^2 / 499 \tag{23}$$

An estimate of the bias of the estimator of projected yield is obtained from

$$\text{Bias} = \overline{\psi} - \widehat{\psi} \tag{24}$$

Results

Case A: Three Survey Indices

The results in Table 1 show that the calculations based on the projected yield formulation and the function of parameters method are virtually identical, as should be expected. The estimated projected yield was 5,543 t with a standard error of 1,357 t and a bias of 556 t. The estimated mean from the Monte Carlo simulation was 6,183 t indicating that the estimator of projected yield had a bias of 639 t. The standard error from the simulation study was 1,477 t. Since the estimated mean projected yield from the simulation study does not correspond to that from the analytical calculations, the results for standard error and bias are evaluated by comparing their magnitudes relative to the estimated means. The results show that the analytical approximations correspond very well with the simulation results, both giving 10% relative bias and 25% and 24% respectively for the relative error. The marginal distribution of the projected yield from the simulation results appears skewed (Fig. 1).

Table 1. The mean, standard error and bias were obtained using two analytical approaches and a Monte Carlo simulation experiment. For Case A where all three survey data sets were used, both the relative precision and relative bias obtained with the analytical approaches corresponded well with the results from Monte Carlo simulations. For Case B where only the Canadian spring survey data were used, the relative precision obtained with the analytical approaches did not correspond well with the results from Monte Carlo simulations, though the relative bias agreed reasonably well.

Case A

	Projected Yield	Function of Parameters	Monte Carlo
Mean	5543.71	5543.57	6183.00
Standard Error	1357.03	1357.03	1476.71
Relative Error	0.25	0.25	0.24
Bias	556.46	556.42	639.30
Relative Bias	0.10	0.10	0.10

Case B

	Projected Yield	Function of Parameters	Monte Carlo
Mean	6217.74	6217.86	8212.08
Standard Error	2558.06	2558.27	5397.06
Relative Error	0.41	0.41	0.66
Bias	1353.48	1353.55	1994.33
Relative Bias	0.22	0.22	0.24

Case B: Canadian Spring Survey Only

As with Case A, the results from the two analytical formulations are virtually identical (Table 1). The projected yield for this case was similar at 6,218 t but the estimate was considerably less reliable with a standard error of 2,558 t and a bias of 1,353 t. The Monte Carlo simulation study resulted in a standard error of 5,397 t and a bias of 1,994 t. Though the relative bias compared well between the analytical and simulation computations, 22% versus 24%, the relative errors were substantially different, 41% against 66%. The marginal distribution of the projected yield from the simulation results was markedly skewed (Fig. 1) and is associated with larger variance for this case study.

Discussion

The stock assessment problem involves non-linear models. Least squares estimators for linear models are unbiased minimum variance linear estimators. For non-linear models, these properties are approached asymptotically as sample size increases. There are no general guidelines for how large the sample size must be before the estimators can be considered reliable because the behaviour is also dependent on the model equations. Ratkowsky (1983) defines "close to linear" behaviour as the condition when bias is negligible, variance is close to the minimum bound and the sampling distribution is approximately normal. Bates and Watts (1988) discuss how the relative curvature measures of non-linearity which they developed can be used to indicate the adequacy of linear approximation inference such as confidence intervals and regions.

Analytical estimates of precision and bias for projected yield can be equivalently obtained from a model formulation which explicitly includes projected yield as a parameter, providing direct estimates, or through the use of suitable methods which derive projected yield as a function of other parameters (e.g., equations 14–15). The advantage of this latter method is that the model can be reparameterized to approach "close to linear" behaviour thereby improving the performance of the search algorithms. With respect to this characteristic, I found that the parameterization using natural logarithms of population abundance was among the best, displaying almost negligible bias and small standard errors. It is important to note however, that the statistical properties of the quantity of interest cannot be altered by the reparameterization. That is, the bias of the projected yield was not altered by solving for a model using the natural logarithms of population abundance as parameters.

Due to non-linearity in the model equations, the analytical estimators of precision and bias may not be reliable even when the model and error assumptions are satisfied. An estimator of covariance which compensates for the effect of non-linearity is described in Seber and Wild (1989, p. 183) but involves third derivatives, complicating implementation. The performance

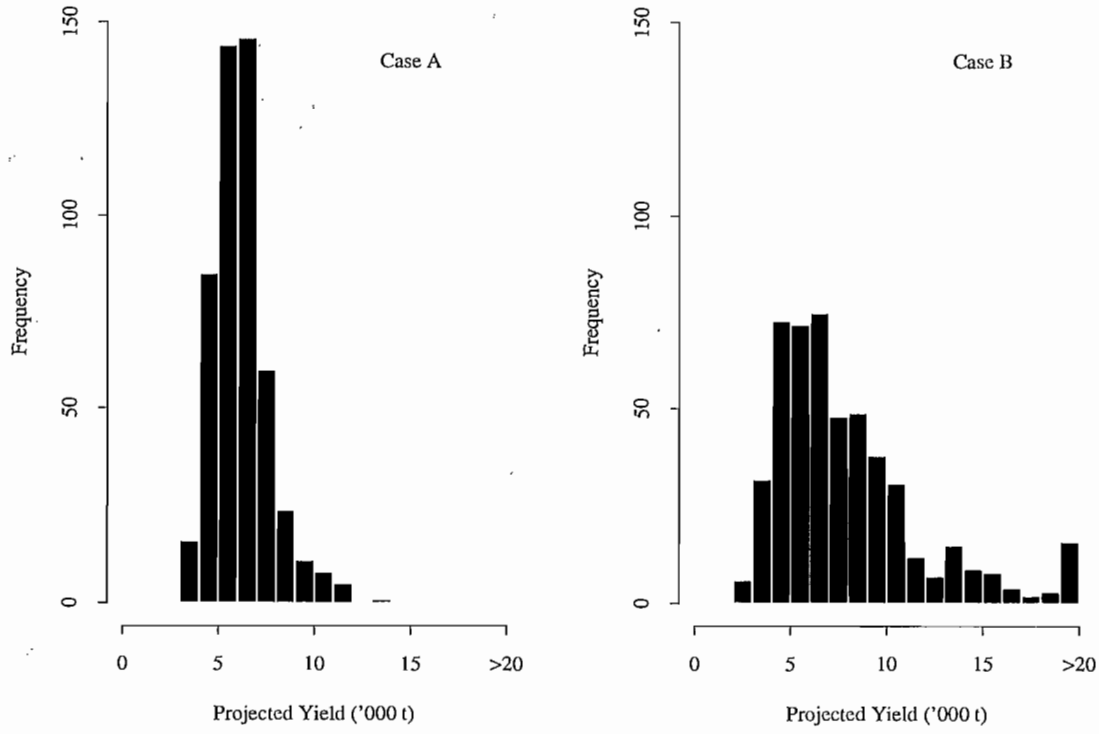


Fig. 1. The marginal frequency distributions of the projected yield from the Monte Carlo simulation experiments show that both cases were skewed. The variance for Case B was greater resulting in a longer tail.

of analytical methods for a class of problems can most readily be evaluated by comparison with results from a Monte Carlo simulation experiment. A simulation study is used to obtain the sampling distribution of an estimator from which estimates of precision and bias can be derived. The replication method used here can be considered a model conditioned parametric bootstrap as the generated pseudo-random data were based on the model equations and on an assumed parametric error distribution.

The results for Case A indicate that the analytical estimates of relative precision and bias were in good agreement with the simulation results suggesting that the linear approximations were adequate. Case A is fairly typical of assessment problems with respect to the number of years for which data is available and the magnitude of the variance. For Case B, the analytical estimate of relative bias was similar to the results from simulation but the relative errors differed substantially. Since the "true" parameter values and variance used for the simulation were the estimates from the analysis of the original data, the asymptotic minimum bound of the variance for the simulation study is equivalent to the variance from the analytical results. Ratkowsky (1983) refers to the discrepancy between the minimum bound and the simulated sampling variance as the "excess variance" and regards it as a diagnostic of the severity of non-linearity. Both cases A and B displayed skewed sampling distributions, indicating that linear approximation inferences should not be used. Techniques which compensate for the effect of non-linearity (Hamilton 1986) should be applied to make valid inferences.

Stock assessments are rarely based on abbreviated data sets such as Case B. As mentioned earlier, this scenario was included for comparison with other replication experiments using this data. Smith and Gavaris (1993) employed non-conditioned non-parametric bootstrap and non-conditioned parametric bootstrap techniques and their results indicated closer agreement to the analytical approximations. As was noted earlier, the simulation study done here can be considered a conditioned parametric bootstrap. With any parametric simulation study, a measure of the "true" variance is required to obtain replicates. For the non-conditioned parametric bootstrap experiment, Smith and Gavaris (1993) employed the estimated sampling error based on the survey design to derive the "true" variance. They obtained age-year specific values of σ which were predominantly in the range of 0.2–0.4. For the conditioned parametric bootstrap described here, the "true" variance was set equal to the estimate of the mean square residual obtained from solving the available realization. The corresponding resultant σ was 0.6. As is generally the case, the "true" variance for this study is not known. Each estimate, the one based on survey design sampling error and the one based on model conditioned results, carries assumptions. For each study, these assumptions need to be evaluated to determine which are more supportable. Often it may not be evident that one or the other estimate is preferred but the

differences in the simulation results provide insight on the implications of each option. For this case study, examination of the distribution of the estimated squared residuals from the model conditioned results showed that there was substantial skewness. The square root of the median squared residual was about 0.3 which corresponds better with the estimates of σ based on survey design. This result coupled with the observation that the non-conditioned bootstrap results were in closer agreement to the analytical approximation suggest that, for this case study, the model conditioned estimate of the true variance, based on the one available realization, probably resulted in a value of σ which was too large. These difficulties serve to caution against indiscriminately using any single estimate of σ in parametric bootstrap experiments for the purpose of making inferences.

An interesting and important issue associated with the identified bias relates to the retrospective pattern which has been reported for assessments (Sinclair et al. 1991). It has been observed that as additional years of data were added to the time series, the estimates of the past were generally lower than what they had been estimated previously. The "statistical" bias estimated here may account for some of the retrospective problem however bias arising from model misspecification is likely to be an important contributing factor.

It can be concluded that for typical stock assessment situations resembling the conditions in Case A, analytical estimates of precision and bias of projected yield are readily computed and would provide acceptable approximations if the model and error assumptions are appropriate. Linear approximation inferences of projected yield should be avoided. Comparison of analytical results with those from simulation studies are encouraged to accumulate experience with this class of problem. Furthermore comparison with simulation results using other replication approaches (e.g., non-parametric bootstrap or those incorporating violations of model assumptions) would provide useful insight on the robustness and sensitivity of estimators.

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Risk Evaluation of the 10% Harvest Rate Procedure for Capelin in NAFO Division 3L

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The fishery for capelin in NAFO Div. 3L is almost exclusively on mature fish. Management of the fishery is based on a total allowable catch (TAC) of 10% of the projected mature biomass one year into the future. The projection is made from a survey estimate of numbers at age in the current year. Survival rates of mature and immature fish are a chief source of uncertainty in the projection. It is demonstrated that average survival rates can be estimated separately for mature and immature fish using the survey data. These estimates differ substantially from those in use. The performance of the existing management procedure, which uses the old estimates of survival rates, is assessed by Monte Carlo simulation. The results indicate that it is highly unlikely that the actual exploitation in any year exceeded 30%. Commonality among capelin fisheries in the North Atlantic may facilitate the development of appropriate assessment procedures and methods for the quantification of risk.

La pêche au capelan dans la division 3L du NAFO en est presque exclusivement une de poissons matures. La gestion de cette pêche repose sur un total des prises admissibles (TPA) équivalant à 10 % de la biomasse mature prévue pour l'année suivante. Cette prévision s'inspire d'une estimation des âges de la piscipopulation de l'année en cours. Les taux de survie des poissons matures et immatures sont une source principale d'incertitude dans cette prévision. Il est démontré que les taux de survie moyens des poissons matures et immatures peuvent être estimés séparément à l'aide des données de relevé. Ces estimations divergent considérablement de celles qui sont actuellement utilisées. Le rendement de la méthode de gestion actuelle, qui a recours aux anciennes estimations des taux de survie, est évalué selon la technique de Monte-Carlo. Les résultats révèlent qu'il est fort improbable que l'exploitation réelle de quelque année ait dépassé 30 %. La relation de communauté entre les pêches au capelan de l'Atlantique Nord peut contribuer à faciliter la mise au point de méthodes et de techniques appropriées pour l'évaluation de la quantification du risque.

Capelin off the east coast of Newfoundland and Labrador were caught for decades in a small inshore fishery on mature fish to satisfy local demands for bait, fertilizer for gardens, human food and food for dog teams. An offshore fishery developed rapidly in the early 1970's and in 1974 the fishery was brought under management by ICNAF (precursor of NAFO, Northwest Atlantic Fisheries Organisation) with a total allowable catch (TAC) of 250,000 tons based on an estimate of the average surplus yield following the decline in predator stocks of cod, seals and whales (Winters and Carscadden 1978). The TAC was increased to 500,000 tons in 1975 and remained unchanged through 1978. A declining capelin stock resulted in the introduction of a new, conservative management procedure in 1979, which we will refer to as the 10% rule. At the same time, the large offshore fishery which took immature and maturing fish was all but eliminated and a much smaller commercial fishery operating on prespawning fish in the nearshore regions developed.

The 10% rule, which has been implemented for most capelin stocks in the Northwest Atlantic, is to recommend an exploitation rate that does not exceed 10% of the projected spawning stock biomass. No analysis was carried out to determine the appropriate exploitation rate for capelin stocks in the Northwest Atlantic prior to the implementation of the 10% rule and, despite the appearance of strong year classes in the 1980's, the 10% rule has been retained. The importance of capelin as a forage species, particularly for cod (Lilly 1991), and the uncertainty associated with the projections are reasons that have been presented for maintaining a conservative management approach.

In this paper we first demonstrate that average survival rates can be estimated separately for mature and immature capelin using the annual acoustic survey data for the stock in NAFO Div. 3L. We then evaluate the existing projection procedure and application of the 10% rule, using Monte Carlo simulation to determine the effect of uncertainty in the assessment inputs (estimated biomass, age structure, proportion mature and survival rates). We present the results as a frequency distribution of the *perceived* TAC (i.e., as perceived by the assessment scientist) as a proportion of the *true* mature biomass (i.e., the biomass of capelin that actually exists in NAFO Div. 3L in the projected year). Finally, we make a brief comparison with capelin fisheries elsewhere to determine the degree of commonality with a view to the development of improved assessment procedures and the quantification of risk.

Materials and Methods

Data Sources Used in Projections

The NAFO Div. 3L (northern Grand Bank/Avalon) stock (Fig. 1) is one of five stocks (or stock complexes) identified in the Northwest Atlantic. This stock occupies the northern Grand Bank as juveniles and once mature, migrates inshore to spawn on Newfoundland beaches in June and July (Carscadden 1983). It is during this prespawning period that the inshore commercial fishery harvests mature females for the Japanese roe market. Annual spring (April–May) acoustic surveys prior to the inshore spawning migration have been conducted on the northern Grand Bank since 1982 and results have been reported to NAFO as a basis for the provision of management advice. Standard echo integration techniques (Miller and Carscadden 1984, Miller 1985) are used in acoustic data analysis. Details of survey design, sampling techniques and results for individual surveys (1982–89) used in this analysis can be found in Miller (1984, 1985, 1986), Miller and Carscadden (1983, 1987, 1988, 1989) and Miller et al. (1982). Data on length and age composition, weights at length, sex composition, and proportion mature are obtained from biological samples taken from midwater trawls carried out during the acoustic surveys. Numbers and weights at age in the catches are estimated from a sampling programme conducted on the inshore commercial fishery, described by Nakashima and Harnum (1984–90).

Estimation of Survival Rates

A simple model for the survival of capelin cohorts is

$$(1) \qquad N_{a+1,t+1} = N_{a,t}\left(1 - p_{a,t}\right) s_1 + \left(N_{a,t}\, p_{a,t} - C_{a,t}\right) s_2 + \varepsilon_{a+1,t+1}$$

where N is the number of fish estimated in the acoustic survey, C is the commercial catch by number, p is the proportion mature by number, s_1 is the average annual (finite) survival rate of immature fish, s_2 is the average annual (finite) survival rate of mature fish and ε is an error term. The subscripts a and t denote age and year, while $a + 1$ and $t + 1$ denote the next older age and the next year. The catch is assumed to be taken at the beginning of the year and the duration of the fishery is assumed to be short, so that fishing and natural mortality act sequentially rather than concurrently (Ricker 1975; type 1 fishery). These assumptions are valid for the capelin fishery in NAFO Div. 3L if the year is taken to commence on 1 June. If $N_{a,t}$, $p_{a,t}$ and $C_{a,t}$ are assumed to be known without error, and the $\varepsilon_{a+1,t+1}$ are assumed *iid* $\mathrm{N}\left(0, \sigma^2\right)$ random variables, then ordinary least squares multiple linear regression can be used to estimate average survival rates s_1 and s_2. Under these assumptions the estimates are also maximum likelihood. The estimates of s_1 and s_2 from the survey and catch data (Table 1) are given in Table 2. From Table 2 it can be seen that, whereas s_1 is reasonably well defined, s_2 has a large standard error. This is taken to be more a reflection of substantial variation in annual survival rate rather than a measure of precision.

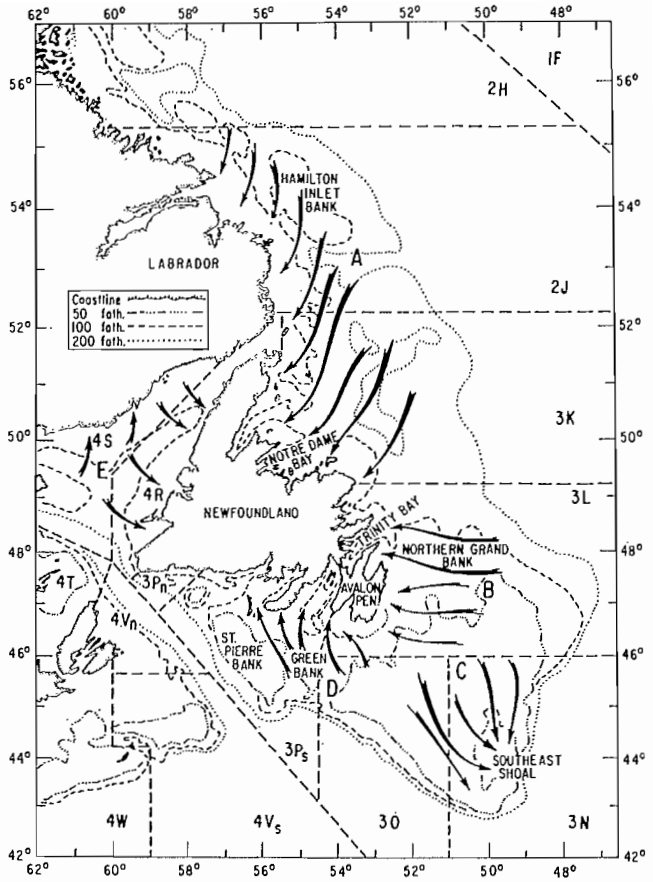


Fig. 1. Map showing major capelin stocks (A–E) and spawning migration routes of capelin in the Northwest Atlantic. A - Labrador/Northeast Newfoundland (NAFO Div. 2J3K) stock; B - Northern Grand Bank/Avalon (NAFO Div. 3L) stock; C - South Grand Bank (Southeast Shoal - NAFO Div. 3NO) stock; D - St Pierre Bank-Green Bank (NAFO Div. 3Ps) stock; E - Gulf of St Lawrence Stock. Arrows indicate the direction of the spawning migration undertaken by mature fish each summer.

Evaluation of past applications of the 10% rule

Projections were initially based on sequential population analysis estimates of numbers at age, but from 1982 onwards acoustic population estimates have been used. In the case of capelin in Div. 3L, an annual acoustic survey of the entire stock is conducted in May and the results are reported to NAFO immediately (June). At the same time, projections for the next fishing season (June/July the following year) are provided. The procedure for calculating the recommended TAC based on the 10% rule is now described.

The following notation is used: $\hat{}$ denotes an estimate from survey and/or catch data; $\bar{}$ denotes an average from the survey or catch estimates over all years; $\tilde{}$ denotes an assumed value or a value taken from the literature; B is the biomass of

Table 1. Survey numbers at age, proportions mature by age, and catch at age data used in the estimation of survival rates for capelin in NAFO Div. 3L.

Survey numbers at age $(\times 10^{-9})$.

	Age			
Year	2	3	4	5
82	9.594	16.195	2.4034	0.8973
83	3.383	1.904	0.8003	0.1008
84	20.132	6.209	3.0950	0.4942
85	368.612	80.379	3.7662	2.3480
86	59.473	158.003	21.2381	1.0083
87	87.513	18.292	38.8983	3.9910
88	382.869	65.696	9.7307	16.7765
89	317.094	96.011	15.3080	1.3936

Survey proportions mature

	Age			
Year	2	3	4	5
82	0.072	0.813	0.972	0.991
83	0.140	0.600	0.975	1.000
84	0.085	0.846	0.989	0.994
85	0.029	0.435	0.900	1.000
86	0.007	0.368	0.933	1.000
87	0.059	0.731	0.978	0.994
88	0.020	0.763	0.992	0.993
89	0.014	0.505	0.977	0.977

Catch numbers at age $(\times 10^{-9})$

	Age			
Year	2	3	4	5
82	0.00653	0.70147	0.08898	0.03416
83	0.03247	0.54312	0.27225	0.01106
84	0.01914	0.42060	0.57322	0.04788
85	0.10180	0.59016	0.20346	0.05849
86	0.00803	1.04304	0.55666	0.04909
87	0.03608	0.10250	0.40304	0.02988
88	0.18302	1.03253	0.25627	0.26753
89	0.02444	1.24383	0.33976	0.03663

fish from the survey; Q is the total allowable catch; w is the weight of an individual fish in the surveys; u is the weight of an individual fish in the catch; h is the proportion by weight of an age group of fish in the survey; k is the proportion by weight of an age group in the catch, and the superscripts i and m refer to immature and mature fish respectively.

The projected mature biomass and TAC in year $t+1$ is obtained from the following equations. The current population at age for immatures and matures is estimated from survey data by

$$\hat{N}^m_{a,t} = \frac{\hat{B}_t \hat{h}_{a,t} \hat{p}_{a,t}}{\hat{w}_{a,t}} \tag{2}$$

Table 2. Multiple linear regression estimates of s_1 and s_2 from survey and catch data for NAFO Div. 3L for the period 1982–89.

	s_1	s_2
Estimate	0.350	0.227
Standard error	0.043	0.239
Covariance	-0.00298	
r^2	0.8079	

and

$$\hat{N}^i_{a,t} = \frac{\hat{B}_t \hat{h}_{a,t} \left(1 - \hat{p}_{a,t}\right)}{\hat{w}_{a,t}} \tag{3}$$

respectively.

The next equation is used to project the number mature at the start of the next year, and equation (5) converts numbers to biomass and sums over all ages.

$$N^m_{a+1,t+1} = \left(\hat{N}^i_{a,t}\tilde{s}_1 + \hat{N}^m_{a,t}\tilde{s}_2\right)\tilde{p}_{a+1} \tag{4}$$

$$B^m_{t+1} = \sum_a \left(N^m_{a+1,t+1}\tilde{u}_{a+1}\right) \tag{5}$$

Equation (6) provides an estimate of the TAC by applying the 10% rule to the projected biomass.

$$Q_{t+1} = 0.1B^m_{t+1} \tag{6}$$

Note that the right side of (4) is incomplete; it should have the recorded commercial catch of fish age a in year t removed. This has not been done in past applications of the projection procedure because catches are small relative to the biomass and are comprised of mature fish, the majority of which die after spawning. In fact the catch between 1982 and 1989 has averaged only 4.3% of the estimated mature biomass. Survey sample estimates of proportions mature, weights at age and age composition are used in equations (2) and (3) to obtain the estimates of numbers at age. However, in the projections (equations (4), (5), and (6)) literature values are used for the survival of spawners (Carscadden et al. 1985) and proportion mature (Carscadden et al. 1981). Constant values derived from inshore sampling are used for mature weights at age. An assumed value is used for the survival rate of immatures. Values used in projections are listed in Table 3.

In order to evaluate the procedure, equations (2) to (6) were incorporated into a Monte Carlo simulation, described in detail in the Appendix. The simulations consisted of repeating 3000 times two sets of parallel computations, one dealing with the possible *true* state of nature and the other with the state *perceived* by the assessment scientist in implementing the current harvest rate procedure for capelin in NAFO Div. 3L. This provided realisations of the TAC (from the application of the harvest rate procedure) as a proportion of the *true* mature biomass. In the realisation of the *true* system, survival rates were assumed to vary in a time dependent (rather than

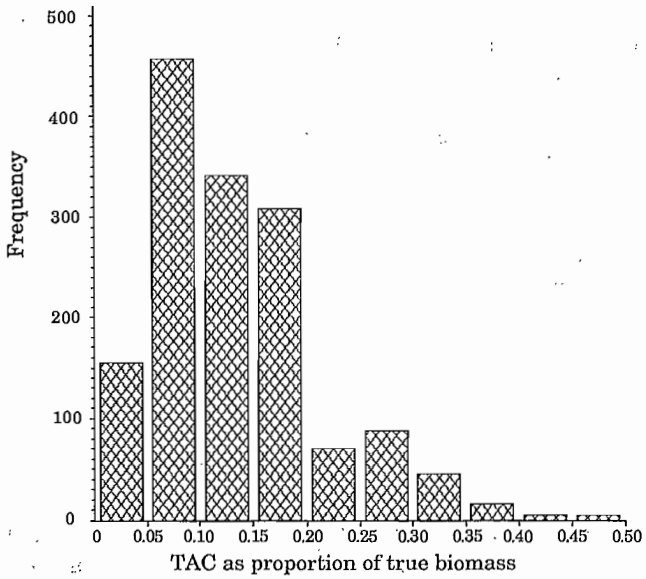


Fig. 2. The simulated TAC as a proportion of the true mature biomass using the 10% rule and assuming a survey sample error CV of 30%. Number of simulations = 3,000.

Table 3. Old parameter values used in the projections to date, compared with new values estimated from the catch and survey data for 1982 to 1989 in this study.

	Age				
Parameter	2	3	4	5	6
Mature weights (g)					
Old		21.20	28.40	31.10	32.40
New	14.18	28.25	36.00	34.31	36.98
Proportion mature					
Old[1]	0.00	0.47	0.87	0.93	1.00
New	0.05	0.63	0.97	0.99	1.00
Annual survival rates					
Immatures					
Old[1]			← 0.74 →		
New			← 0.35 →		
Matures					
Old[1]	0.18	0.18	0.14	0.08	0.08
New			← 0.23 →		

[1] Carscadden et al. (1985).

age dependent) manner, thus yeat specific pairs of s_1 and s_2 estimated from the survey and catch data were randomly resampled to simulate this variability. Of course, the variability in year-specific survival rates also reflects measurement error. Thus the simulation procedure would tend to incorporate too much uncertainty in the survival rates, and thus be conservative. Data for ages 3 and 4 in 1984 and age 3 in 1985 were omitted in the calculation of survival rates because inclusion would result in an estimate of a survival rate greater than 1. This tends to give a negative bias to the simulated survival rates because years with high survival rates are more likely to be excluded. Again, this is conservative.

The results are presented in Fig. 2 as a frequency distribution for the simulated TAC as a proportion of the true mature biomass, assuming an acoustic survey sample error CV of 30% and no bias. Cumulative probabilities of a TAC that is less than or equal to a specific proportion of mature biomass are plotted in Fig. 3 for the case illustrated in Fig. 2 (a) as well as for a sample error CV of 40% on the acoustic estimate (b) and for a CV of 30% and a positive bias of 30% (c).

It is apparent that the existing procedure (under the assumption of an unbiased acoustic survey estimate with a sample error CV of 30%) performs reasonably well in achieving the objective with about 50% of the trials below 12% of the mature biomass and 50% above. There is only a small probability (< 0.1) that a TAC would be recommended that was actually 30% or more of the mature biomass.

This result is somewhat surprising, considering that we now believe the survival rates, particularly for immature fish, to be quite different from the values used up until now (Table 3). The reason that the existing procedure performed reasonably is that the values for weights at age in the projection are less than the mean of the estimates of weight at age of fish from the catch samples, and the proportions mature are less than the mean of those estimated from the survey samples (Table 3). These two factors compensate for the higher survival rate for immatures used in the projections.

An increase in the CV of the acoustic estimate from 30% to 40% moved the cumulative probability curve to the right as expected, but only by a small amount (Fig. 3). A 30% positive bias in the acoustic estimate had a greater effect on the cumulative probability curve with the median of the distribution of TAC as a proportion of true biomass changing from 0.12 to 0.16. The simulation thus suggests that, under the assumptions made, the performance of the 10% rule has been relatively insensitive to sample error in the acoustic survey and that a positive bias in the acoustic estimate would have to be greater than 30% in order to result in less than an 80% probability that the TAC has remained below 30% of the true biomass.

Discussion

Although the 10% rule has been exercised in NAFO Div. 3L using proportions, weights and survival rates which do not correspond to those estimated from the surveys and catches, it has performed reasonably well in achieving the desired goal of providing a TAC of around 10% of the mature biomass. The approach used appears to be robust to sample error and bias in the acoustic estimate, and even with a positive bias of 30%, there appears to be more than an 80% chance of choosing a TAC which will be less than 30% of the true biomass (curve c, Fig. 3).

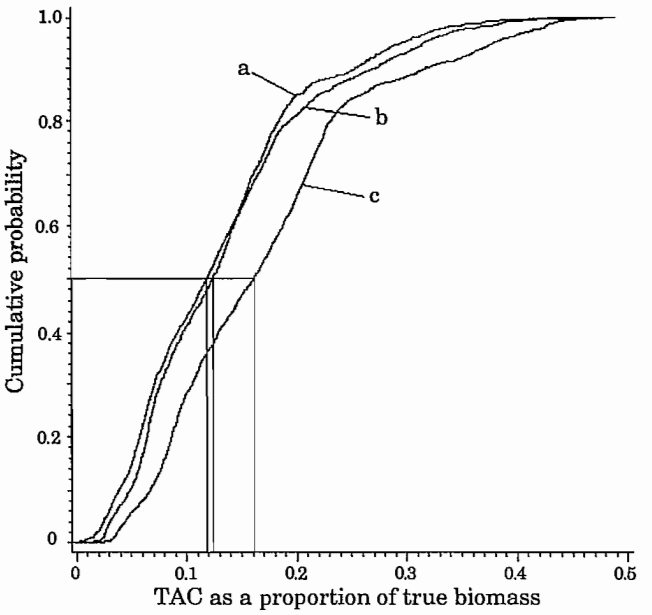


Fig. 3. The cumulative probability of the 10% rule giving rise to a TAC that is less than or equal to a specified proportion of the mature biomass for cases with different variance and bias associated with the acoustic estimate: a- CV=30%, bias=0%; b- CV=40%, bias=0%; c- CV=30%, bias=+30%.

The estimate of survival rate for the spawners has a large standard error which, under the assumptions of the model, must be attributed to considerable variability in survival among years or ages or both. In reality it could equally be a result of error in the estimates of population size, proportion mature and catch at age. Of particular importance is the estimate of survival rate of the non-spawners which, at 0.35, is substantially less than the value of 0.74 assumed up until now. This result emphasises the role of capelin as a forage species in the North West Atlantic system and reinforces the need to maintain an appropriately small fishing mortality.

There is some commonality in the capelin fishery in NAFO Div. 3L and the capelin fishery in Div. 2J3K, as well as those fisheries in the Barents Sea and off Iceland. In each case an annual acoustic estimate is made from which projections are carried out to serve as a basis for setting TAC's. All four stocks are similar in that the mature spawning biomass is composed of three-, four-, and five-year-olds and survival rate after spawning is low (assumed to be zero in Barents Sea and Iceland and estimated to be below 0.25 for NAFO Div. 2J3KL (Carscadden et al. 1985)), and an average of 0.23 in this study. The average annual survival rate estimated for immature capelin in the Barents Sea is 0.38 (Bjarte Bogstad, Marine Research Institute, Bergen, Norway, pers. comm.) compared to an average value of 0.35 estimated in this study. These values are substantially lower than the assumed value currently used in assessments in the Northwest Atlantic (0.74). It would be informative to have estimates of the average survival rate of immatures from NAFO Div. 2J3K and Iceland for comparison with the values from Div. 3L and the Barents Sea.

The Barents Sea capelin fishery is managed on the basis of a constant escapement of 500,000 tons of spawners to 31 March (Hamre and Tjelmeland 1982). Fish spawn on the bottom at depths ranging from 10–100m in March and April. An acoustic survey is carried out in September to provide an estimate of the biomass and numbers at age of fish two years old and older. The stock is assessed in October and projections of mature and immature fish are made based on estimates of survival rates and the assumption that fish exceeding a specified length are mature and will spawn. Separate TAC's are set for winter and summer/fall fisheries. The TAC for the winter fishery is set at the appropriate level to achieve the target escapement of spawners. Post-spawning survival is assumed to be zero. The immature portion of the population (mostly three-year-olds) is projected forward to the next fall to provide a preliminary TAC for the next fall and winter fishery. This TAC is revised during the October assessment taking into account the new survey data.

The Iceland fishery is managed on the basis of a constant escapement strategy of 400,000 tons of spawners (Vilhjalmsson 1983). Spawning takes place in March and the first half of April in shallow coastal waters. Fisheries take place in both the fall and winter. An acoustic survey is carried out in August and the annual assessment is carried out in October. As for the Barents Sea, a TAC for the winter fishery is set so as to reach target spawner escapement. A preliminary TAC is also set for the next fall and winter fisheries. A second acoustic survey is carried out in February–March to check on spawner escapement and in some years to provide estimates of recruitment. The TAC for the fall and winter fisheries is revised on the basis of the new survey data.

The fishery in Div. 2J3K is divided into an inshore component in summer and an offshore component in fall/winter. The summer fishery is essentially similar to the one pursued in Div. 3L (i.e., on mature fish, mainly 3- and 4-year olds), whereas the fall/winter fishery is on immature fish (mainly aged 2- and 3-years old), most of which will spawn the following year. Acoustic surveys are carried out in October each year. The stock is assessed in February of the following year. The TAC for the inshore fishery is calculated as for Div. 3L, and in addition a TAC of 10% of the total biomass as of 1 September (i.e., after spawning) is calculated for the fall offshore fishery. In the projection for the fall fishery, the geometric mean number of 2 year olds estimated in past acoustic surveys is used to represent the size of this (as yet unsurveyed) age class in the current year.

The basic similarities between the NAFO Div. 3L capelin fishery and the fisheries described above suggest that it is likely that the method used here to estimate survival rates, or some variant of it, would be appropriate in the other fisheries, and that assessment procedures could be developed for each of the four systems which are basically similar. Projections now carried out use only the current biomass estimate and ignore the serial correlation in estimates due to the persistence of year-classes between surveys. A Bayesian-like estimation model of the kind described by Butterworth and Bergh

(1993), which accounts for the time-series nature of the problem may be an appropriate approach and should be explored. A comparison of parameter estimates among systems and a consideration of the value of the alternative management goals (i.e., constant escapement in the Barents Sea and off Iceland versus constant proportion in the Northwest Atlantic) could prove extremely useful in the development of assessment procedures and the quantification of risk associated with these procedures.

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Appendix

Monte Carlo Simulations for Evaluating the Existing Procedure Used to Manage Capelin in NAFO Div. 3L

Two sets of parallel computations are repeated 3000 times, one dealing with the possible "true" state of nature and the other with the state "perceived" by the assessment scientist. The true state starts with an arbitrary level of biomass. A known catch is subtracted from the mature portion of the biomass. The remaining biomass is then projected ahead one year and the proportion mature is computed. The appropriate total allowable catch is 10% of the projected mature biomass. The assessment scientist attempts to quantify each of these steps to estimate what is 10% of the mature biomass one year into the future.

The steps of the model are of two types. For some steps, we can assume a value for a true parameter and assume that the scientist attempts to estimate this parameter. For example, we assume the true biomass at the beginning of the simulation is an arbitrary value B, and that the assessment biologist can obtain an unbiased estimate of this with a coefficient of variation (CV) of 30%. In each run of the simulation, we represent this by generating a perceived biomass as a normal random variable with expectation B and CV of 30%. In other steps of the model, the assessment scientist uses assumed parameter values. Consequently, in the simulation we must generate a possible value of the true parameter. For example, in the past the assessment scientist has assumed values for the survival rates of immature and mature capelin. To represent the possible values of the true parameter values, we used the new method described in the main text to estimate the survival rates for each pair of adjacent years from the survey data. In our simulations, we randomly resample the pairs of year-specific estimates of s_1 and s_2 estimated from the survey and catch data.

A third type of step, not used in our simulations, is where the particular value of the true parameter is influential and the assessment scientist estimates the parameter. It is then necessary to specify possibilities for the true value and, conditioual on the specified true value, to generate an estimate that the scientist might obtain. For example, in the future the assessment scientist might estimate the survival rates each year from the survey data instead of assuming a value. We may only have a poor idea of the true survival rates and the survival rates are influential. Therefore, we would want to include many possible values of the true rates in the simulations. For each possible true survival rate generated, we would need to generate a perceived survival rate centered on the current value of the true rate.

We now discuss the differences between type 1 and type 3 steps. In a type 1 step, we assume a particular value for a true parameter rather than specifying a range of possibilities for the parameter as in a type 3 step. In general, if the simulation results depend heavily on the particular value of a true parameter, then all reasonable values of the true parameter should be represented in the simulation. Otherwise, an arbitrary value of the true parameter may suffice. In some cases, it may be possible to show analytically the sensitivity of the results to the value of a particular parameter. In other cases, a sensitivity analysis may be used to establish if the additional complexity of a type 3 step is needed over the simplicity of a type 1 step.

In the example above, the initial biomass was set at an arbitrary level B^*. Since the simulations are concerned with the proportion of the biomass harvested, rather than with absolute biomass, the actual value of the biomass drops out and any value of the initial biomass would be suitable. This is only true because the actual catch averaged only 4.3% of the mature biomass estimated by acoustics (i.e., the catch was a trivial proportion of the true biomass). If the exploitation rate had varied considerably so that the catch sometimes accounted for a large portion of the biomass then we would have had to specify a distribution of the biomass to be projected ahead one year. This distribution would have to account for the variability due to the initial biomass level and catch level.

In some instances we assume that the true parameter value is the long-term historical mean of the available estimates, and that the collection of annual estimates comprises an empirical distribution of what might be observed in any year. For example, the proportion mature at age 3 has been estimated annually from 1982 to 1989 from the survey data. We assume in our simulations that the true proportion mature is the arithmetic mean of these historical estimates. In the "perceived" part of the simulation, the assessment scientist might make any of the historical estimates of the proportion mature. This approach is conservative. We are assuming that the variability or error in estimates of proportion mature is equal to the variability of the estimates about the long-term mean. However, the estimates vary from year to year due to both sampling error and actual change in the proportion mature. Therefore, our simulations include an extra component of variability and are conservative in this respect.

In what follows, we denote a realisation of a quantity in the true system by an asterisk (*). A realisation of an estimate of a parameter is denoted with a hat ($\widehat{\ }$) symbol. Other symbols are as defined in the text.

Simulation Procedure

Realisation of true system

1. The true biomass is taken to be an arbitrary value B.

$$B_t^* = B$$

2. The true values for numbers at age in the simulation are obtained from the true biomass, the mean weight at age over all the surveys, and the mean proportion by age over all the surveys.

$$N_{a,t}^* = \frac{B_t^* \overline{h}_a}{\overline{w}_a}$$

3. The true catch at age by number is derived from a catch of 4.3% of the true mature biomass. First, the mature number at age is determined.

$$N_{a,t}^{m*} = \frac{B_t^* \overline{h}_a \overline{p}_a}{\overline{w}_a}$$

Then the mature number at age is converted to biomass using mean weights at age in the catch, reduced by the average catch as a proportion of the mature biomass, and finally reconverted to numbers at age.

$$C_{a,t}^* = \left(0.043 \left(\sum_a N_{a,t}^{m*} \overline{u}_a\right)\right) \frac{\overline{k}_a}{\overline{u}_a}$$

4. The true projected numbers at age in year $t+1$ is then simulated using survival rates s_1 and s_2 drawn randomly from pairs of values estimated from adjacent years of historic data.

$$N_{a+1,t+1}^* = N_{a,t}^* \left(1 - \overline{p}_a\right) s_1 + \left(N_{a,t}^* \overline{p}_a - C_{a,t}^*\right) s_2$$

5. The true mature biomass in year $t+1$ is simulated using the mean weight at age in the catch (over all years) from annual estimates and the mean proportion mature at age (over all years) from the annual survey estimates.

$$B_{t+1}^{m*} = \sum_a \left(N_{a+1,t+1}^* \overline{u}_{a+1} \overline{p}_{a+1}\right)$$

Realisation of perceived system

1. The perceived biomass (i.e., estimated biomass) is generated randomly from a normal distribution with mean B and variance σ_2.

$$\widehat{B}_t \sim \mathrm{N}\left(B, \sigma^2\right)$$

2. The perceived biomass is apportioned to numbers at age based on vectors of proportion by weight and individual fish weights. These vectors are selected randomly with replacement from the matrix of survey estimates from 1982 to 1989 (same year selected for both vectors to account for possible covariance).

$$\widehat{N}_{a,t} = \frac{\widehat{B}_t \widehat{h}_{a,t}}{\widehat{w}_{a,t}}$$

3. The perceived catch is assumed to be zero, as done in practice.

$$\widehat{C}_{a,t} = 0$$

4. The perceived numbers at age in year $t+1$ is simulated using literature values for survival rates ("old" values in Table 3) and estimates of proportion mature simulated by randomly selecting a vector of proportion mature from the matrix of annual survey estimates for 1982 to 1989.

$$N_{a+1,t+1} = \widehat{N}_{a,t} \left(1 - \widehat{p}_{a,t}\right) \widetilde{s}_1 + \left(\widehat{N}_{a,t} \widehat{p}_{a,t} - C_{a,t}\right) \widetilde{s}_2$$

5. The perceived biomass in year $t+1$ is calculated using assumed and literature values for weight-at-age in the catch and proportion mature (see "old" values in Table 3).

$$B_{t+1}^m = \sum_a \left(N_{a+1,t+1} \widetilde{u}_{a+1} \widetilde{p}_{a+1}\right)$$

6. Not applicable.

6. The perceived TAC for the realisation is then calculated, following the 10% rule.

$$Q_{t+1} = 0.1 B^{m}_{t+1}$$

7. Finally, the true exploitation rate (relative to mature biomass) associated with the perceived TAC is calculated using the true mature biomass.

$$E^{*}_{t+1} = \frac{Q_{t+1}}{B^{m*}_{t+1}}$$

Using jackknife and Monte Carlo simulation experiments to evaluate forecast models for Atlantic salmon (*Salmo salar*)

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A method is presented which can be used to test the expected performance of assessment models using jackknife and Monte Carlo simulation techniques. Our example is a forecast model, based on probability distribution functions, which incorporates pre-season run-size and in-season run-timing data for an Atlantic salmon population. We tested the model's expected performance by Monte Carlo simulation. Simulations incorporated variability in the following population parameters: natural mortality at sea, fisheries exploitation rates, total post-spawning mortality, proportion returning to the river, and annual variation in run-timing. We also introduced variation in the two parameters used to estimate river returns, estuarine harvest and research trapnet efficiency. We applied jackknife and Monte Carlo techniques to determine the expected performance of the forecast model under conditions of increasing parameter uncertainty. We found the model was robust to the levels of uncertainty which have been estimated from research data. These results lend confidence to the forecast model and emphasize the importance of evaluating a model's performance before it is applied to assessments. Jackknife and Monte Carlo procedures are likely to provide a useful testing procedure for a wide variety of marine and freshwater assessment models.

L'article fait état d'une méthode qui peut être utilisée pour faire l'évaluation du rendement prévu de modèles d'évaluation à l'aide des techniques de simulation du Jackknife et de Monte-Carlo. Notre exemple est un modèle de prévision, reposant sur les fonctions de distribution de probabilité, qui conjugue les données de la taille de montaison avant saison et de la durée de montaison en saison pour une population de saumon de l'Atlantique. Nous avons fait l'évaluation du rendement prévu du modèle à l'aide de la simulation de Monte-Carlo. Les simulations tenaient compte de la variabilité des paramètres de population suivants : mortalité marine de cause naturelle, taux d'exploitation des pêches, mortalité totale après la fraie, proportion de remonte et variation annuelle de la durée de montaison. Nous avons aussi étudié la variation des deux paramètres utilisés pour l'estimation des remontes, soit la prise en estuaire et l'efficacité des filets de recherche. Nous avons utilisé les techniques du Jackknife et de Monte-Carlo pour déterminer le rendement prévu du modèle de prévision en situation d'augmentation de l'incertitude des paramètres. Nous avons déterminé que le modèle était efficace pour les niveaux d'incertitude estimés selon les données de recherche. Ces résultats permettent de croire en l'efficacité du modèle et soulignent l'importance d'évaluer le rendement d'un modèle avant de l'appliquer aux évaluations. Tout porte à croire que les techniques du Jackknife et de Monte-Carlo pourront s'avérer utiles pour l'évaluation d'un grand nombre de modèles d'évaluation en eaux douces et en eaux salées.

Forecasting population numbers and projecting catches are an important part of fish stock assessments. If managers know in advance the number of fish expected to return or to be caught in a fishery the next year, and have accompanying estimates of uncertainty, they can set harvest and/or effort levels that are consistent with a given probability of meeting management targets. Fishery forecast models are often presented without estimates of uncertainty and without evaluating how they may perform given the variability inherent in parameter estimates. This deficiency has meant that models are often not properly evaluated prior to implementation and fail due to unanticipated causes. These failures frequently result in *ad hoc* changes to advice which may erode a manager's confidence in forecasts and stock assessments. In this paper we provide an example of a method, which employs jackknife and Monte Carlo simulation techniques, to evaluate a forecast model's expected performance.

The forecast model uses probability distribution functions to predict returns of Atlantic salmon (*Salmo salar*) to the Miramichi River. The pre-season probability distribution functions follow the ideas found in Evans and Rice (1988), Rice and Evans (1988), and Noakes (1989); while the in-season methodology relies primarily on that of Noakes (1989). Forecast uncertainty is easily incorporated into the provision of

management advice from these models because the pre-season and in-season probabilities of returns can be read directly from cumulative probability curves.

The model's performance was tested by performing a series of Monte-Carlo simulation experiments with increasing parameter uncertainty (Miller 1974). Forecast returns were compared to known returns from simulated data to determine the expected performance of the model. Each of these comparisons used jackknife leave-one-out procedures and allowed us to test the model on each year without including the forecast year in the data set; thus removing a potential source of bias in performance testing (Srivastava and Carter 1983). Simulated populations were created using a model with the following parameters: natural mortality at sea, fisheries exploitation rates, total post-spawning mortality, proportion of the population returning to the river, and annual variation in run-timing. Variability in the two parameters used to estimate returns to the river, estuarine harvest and research trapnet efficiency were also incorporated into the experiments.

While our example relates specifically to freshwater assessments, the technique of comparing model results to known simulated populations is applicable to many assessment models. Regardless of the model, simulation experiments offer a powerful means of investigating model sensitivity and expected performance under conditions that could not be examined in any other way. We expect wider application of these techniques will improve assessment models and the provision of biological advice.

Materials and Methods

Objectives

Our objective is to forecast numbers of one-sea-winter (1SW) and multi-sea-winter (MSW) salmon returning to a river. 1SW salmon are those which have spent one-winter maturing at sea before returning to spawn. MSW salmon are those which have spent two or more winters maturing at sea and are returning to a river to spawn for the first time but may also include those which have previously spawned. In the Miramichi River most MSW salmon (60–95%) have spent two winters at sea and are returning to spawn for the first time (2SW salmon). On average fewer than 5% are virgin three (3SW) or four-sea-winter (4SW) salmon while repeat spawners have ranged from 5% to 40% (Randall 1985; Moore et al. 1991). Forecasting MSW salmon returns are important because they have 10–20 times more eggs than 1SW salmon (Randall 1989). Thus, MSW salmon account for up to 95% of the egg deposition in some Atlantic Canada rivers and on average 85% of the egg deposition in the Miramichi River (Randall 1985). Forecasting 1SW salmon is important because current regulations restrict angling harvests exclusively to this age group, while MSW salmon must be released if hooked by anglers. Both MSW and 1SW salmon are, however, subject to harvest in non-recreational fisheries.

A forecast model for 1SW salmon has never been developed for the Miramichi River, and previous attempts to forecast MSW salmon returns have relied on linear multiple regression models. Recent analyses have shown that parametric linear regression and time-series models are not reliable means of forecasting MSW salmon returns to the Miramichi River, but that non-parametric probability distribution function models may provide reliable forecasts (Claytor et al. 1991).

Our proposed forecasting models use probability distribution functions (Silverman 1986; Evans and Rice 1988; Rice and Evans 1988; Noakes 1989) to provide a pre-season forecast of MSW salmon returns which is then updated and improved with in-season run-timing data. 1SW salmon forecasts must rely exclusively on in-season data because there is no abundance index which is correlated to subsequent 1SW salmon returns. Smolt abundance which has been useful for forecasting 1SW salmon returns in other rivers (Chadwick 1987) has not recently been estimated for the Miramichi River (Kerswill 1971). In addition, juvenile parr densities, estimated since 1971, do not seem to be able to predict 1SW salmon returns, although we are currently investigating this question. Below, we describe how we test the expected performance of these models using jackknife and Monte Carlo simulation techniques and demonstrate how these pre-season and in-season models could be used to provide management advice.

Pre-Season Analysis

Observed data — We use 1SW salmon returning in one year (year i) to forecast MSW salmon returns the next year (year $i+1$). We may expect a relationship between these two life-history forms because the 1SW salmon and most of the MSW salmon (as 2SW salmon) are from the same smolt class. As a result, 1SW salmon returns provide the first indication of expected returns from a smolt-class.

Estimates of salmon returns to the Miramichi River are obtained from counts using a Department of Fisheries and Oceans research trapnet located in the Miramichi estuary at Millbank (Fig. 1). Trapnet efficiencies determined from mark-recapture experiments in 1973 and 1985–1987 (Randall et al. 1989) and updated annually from tag recoveries at counting fences and angling camps (Moore et al. 1991) are used to estimate returns. Estuarine harvest below the Millbank trapnet is added to returns at Millbank to estimate total returns to the river. Trapnet operation and estimation procedures are described in Randall et al. (1989). Total return estimates, based on these trapnet efficiencies, are available from 1971–1990 (Moore et al. 1991).

Probability distribution functions — We used probability distribution function models (PDF) (Silverman 1986; Evans and Rice 1988; Rice and Evans 1988; Noakes 1989) with a Gaussian kernel estimator as described by Noakes (1989) to model pre-season MSW salmon returns, y, to the Miramichi River as shown below:

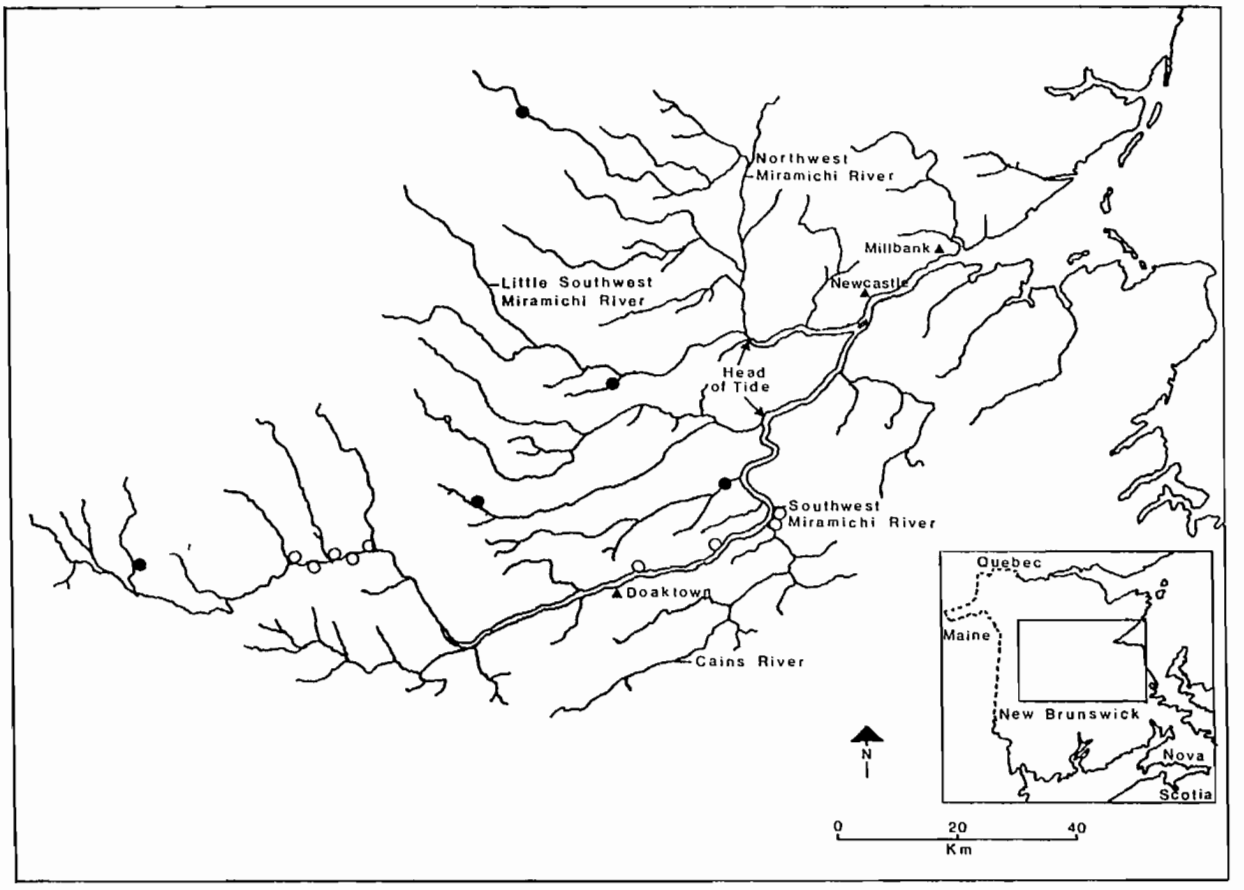


Fig. 1. Location of Millbank trapnet used to estimate salmon returns to the Miramichi River after estuarine harvest. Solid circles represent counting fence sites and open circles represent angling camp sites used as tag-recovery sites in mark-recapture experiments.

$$(1) \qquad f(\mathbf{y}) = \frac{1}{n}\sum_{i=1}^{n} \frac{1}{(2\pi)^{\frac{d}{2}} \prod_{j=1}^{d} h_j} \times \prod_{j=1}^{d} \exp\left[-\frac{1}{2}\left\{\frac{(y_j - x_{ij})}{h_j}\right\}^2\right]$$

where,

n = the number of observations (years);

d = the number of variables, in this case $d = 2$;

y = a vector of length d, y_j refers to the jth element of this vector;

x_{ij} = the observed values for the variables entering the model which in our case are 1SW (year i) and MSW (year $i+1$) returns; and

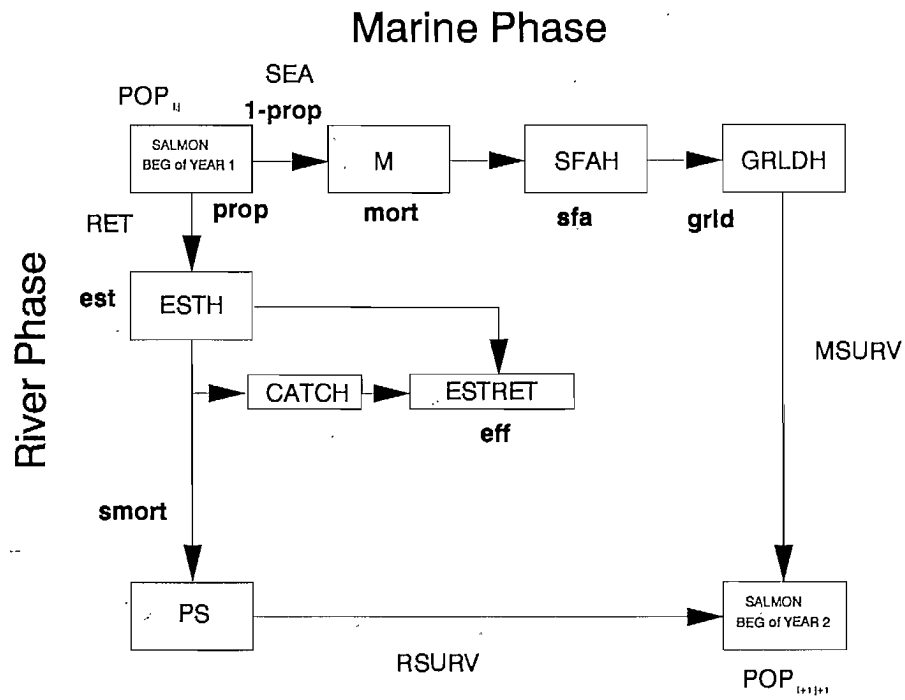
$\boldsymbol{h} = (h_1, \ldots, h_d)$, the vector of smoothing parameters.

Smoothing parameters were obtained by maximizing the likelihood function described by Noakes (1989) and shown below:

$$(2) \qquad L(\boldsymbol{h}) = \prod_{i=1}^{n} f_i(\boldsymbol{x}_i; \boldsymbol{h})$$

where, $f(\boldsymbol{x}_i; \boldsymbol{h})$ is the function in (1) evaluated at each row in the matrix x_{ij}.

The algorithm developed by Powell (1964) and implemented in FORTRAN by Noakes (1989) was translated into SAS (1990) to calculate the smoothing parameters and forecast returns for each year. Separate smoothing parameters were calculated for each variable. Logarithms of the data were used to avoid forecasting negative returns; these transformed data were normalized to avoid extreme differences in spread among the data sets examined; and outliers greater than one standard deviation from the mean were weighted by the inverse of the distance from the mean to dampen their effect on the smoothing parameter estimate. The anti-log of the maximum likelihood value provided the forecast (Noakes 1989).



Rates:

River phase

prop: Proportion back to river

est: Estuarine exploitation

eff: Research trapnet efficiency

smort: In-river mortality

Marine phase

1-prop: Proportion at sea

mort: Natural marine mortality

sfa: Newfoundland exploitation

grld: Greenland exploitation

Population numbers:

River phase

RET (River Returns) = POP x **prop**;

ESTH (Estuarine harvest) = RET x **est**;

CATCH (Research trapnet catch) = (RET-ESTH) x **0.015**

ESTRET (Estimate of river returns) = ESTH + (CATCH)/**eff**

PS (Post-spawners) = (RET-ESTH)-((RET-ESTH) x **smort**)

RSURV (River survivors) = RET - (ESTH-PS)

Marine phase

SEA (Salmon remaining at-sea) = POP x (**1-prop**);

M (Natural sea mortalities) = SEA x **mort**;

SFAH (Newfoundland harvest) = (SEA - M) x **sfa**;

GRLDH (Greenland harvest) = (SEA - M - SFAH) x **grld**;

MSURV (Marine survivors) = SEA - (M - SFAH - GRLDH)

$POP_{i+1,j+1} = RSURV_{i,j} + MSURV_{i,j}$

Fig. 2. Schematic of model used to simulate data for known populations of salmon returns. Upper case abbreviations refer to population numbers; lower case bold abbreviations to rates used to derive numbers. POP, the initial number of salmon at sea, was chosen randomly from a uniform distribution from 35,000 to 400,000 at the beginning of each year.

Pre-Season Model Comparisons

Observed data — The procedures of Evans and Rice (1988) and Noakes (1989) used cross-validation or jackknife procedures to calculate PDF smoothing parameters. We added an additional jackknife leave-one-out procedure, as described by Srivastava and Carter (1983) for discriminant function analysis, to evaluate the performance of pre-season PDF forecasts relative to those using mean numbers of returning MSW salmon (MEAN). For example, forecasts for MSW salmon returns in 1971 were obtained by deleting 1971 from the data set and calculating the PDF forecast using all other years, while the MEAN forecast for 1971 was simply the mean number of MSW salmon returning in all other years. Then 1971 was replaced in the data set and the next year was deleted. This procedure continued until all years had been deleted and replaced in the data set.

Jackknife residual sum of squares from the PDF and MEAN models and the difference between PDF and MEAN absolute residuals (Cleveland 1985) were used to evaluate the relative performance of each model. Absolute differences were plotted against the range of observed ratios to compare the relative bias of each model. PDF residuals were calculated from the anti-log of the maximum likelihood value which provided the forecast so that PDF and MEAN residuals were calculated on the same scale.

Monte Carlo experiments: Pre-season simulated data — We tested the expected performance of the pre-season models using Monte-Carlo simulation experiments including the previously described jackknife procedure for observed data. The simulation creates two types of data: 1) known numbers of 1SW and MSW salmon returning to the river using age-at-maturity, exploitation, and mortality rates; and 2) estimated numbers returning to the river using exploitation and research trap efficiency rates. The estimated number of 1SW and MSW salmon were used to develop the MSW pre-season forecasts and these forecasts were then compared to the known MSW salmon returns created by the simulations.

Simulated populations began with a pool of fish at sea (POP) and a proportion (**prop**) of these which return to the river. Those remaining at sea are subject to a natural mortality rate (**mort**), then are harvested sequentially in the Newfoundland fishery with an exploitation rate designated (**sfa**) and then in the Greenland fishery with an exploitation rate designated (**grld**). Those returning to the river are first harvested in an estuarine fishery, with an exploitation rate designated (**est**). Survivors are subject to mortality from angling, poaching, disease, and post-spawning mortality. These sources were combined into one mortality rate designated (**smort**). Survivors from these mortalities re-enter the population as repeat spawners the following year (Fig. 2).

The number of 1SW salmon returning to the river depends only on POP and **prop**: while the number of MSW salmon returning to the river is a combination of virgin 2SW, 3SW, and 4SW spawners and repeat-spawners from 1SW, 2SW, 3SW, and 4SW age-classes (Fig. 2).

Monte Carlo experiments: rate estimates — Mean values for the population parameters **mort, sfa, grld,** (Fig. 2) were those used by the Canadian Atlantic Fisheries Scientific Advisory Committee (CAFSAC 1989) and the International Council for the Exploration of the Sea (ICES 1990) with ranges set to what we believed to be realistic limits. Ranges for **est** (Fig. 2) were obtained by dividing estuarine harvest by the estimate of total returns (Moore et al. 1991). Ranges for **smort** (Fig. 2) were determined by dividing the number of repeat spawning salmon, year $i + 1$, by the number of virgin spawning salmon, year i, for 1SW and MSW age classes. Values for **prop** were estimated by calculating the ratio of virgin 1SW:virgin 2SW salmon and selecting the range in **prop** that produced the ratios seen in the data (Claytor; unpublished data). The range for the initial number of salmon at sea was determined using the Miramichi River smolt production estimates of Kerswill (1971) and applying the minimum and maximum estimates of smolt marine survival for Gulf of St. Lawrence rivers (Chadwick 1987). Pre-season simulation 1 provides the initial limits given to these parameter ranges (Table 1).

Monte Carlo experiments: Pre-season model behaviour — We first examined model behaviour by holding all parameters constant and letting one parameter or one group of similar parameters vary according to the ranges shown in pre-season simulation 2 (Table 1).

Seven pre-season simulation experiments of increasing uncertainty were conducted to test the PDF and MEAN models (Table 1). The first three experiments examined the effect of increasing uncertainty in estimating salmon returns (ESTRET) (Fig. 2). These tests were done by increasing the variation in **eff** as follows: constant (pre-season simulation 1), 95% confidence interval from the mean of the three-year experiment (pre-season simulation 2), and a range equivalent to the largest 95% confidence interval observed in a single year during the mark-recapture experiment (pre-season simulation 3) (Randall et al. 1989). In pre-season simulation 4, uncertainty effects in ESTRET was tested by assuming a 40% coefficient of variation in ESTH rather than a 20% coefficient of variation as was assumed for all other simulations (Table 1). The effects of uncertainty in **prop, mort, sfa, grld,** and **smort** on the forecast model were examined in pre-season simulations 5–7 (Table 1).

Pre-season simulation 2 was the control in these experiments, its ranges represented the most realistic values, and all values were returned to the conditions of this simulation before changes were made for the next experiment (Table 1). For each simulation experiment the parameters were randomly selected from a uniform distribution. Selection of this distribution assumes the least knowledge about the error structure of the parameters and should provide the most robust test of the forecast model. We used the same methods to evaluate the simulation experiments as we did for the forecast model using observed data. Jackknife PDF and MEAN forecasts were compared using differences in absolute residuals.

Table 1. Parameter ranges used in simulations to test pre-season model. The initial number of salmon at sea (POP) ranged from 35,000 to 400,000. A * indicates parameter ranges which are different from those of simulation 2. Simulation 4 is the same as simulation 2 except that a 40% coefficient of variation, rather than a 20% coefficient of variation is applied to estuarine harvest. Except for PROP, MSW refers to all 2, 3, and 4 sea-winter salmon. % success represents the percentage that the PDF model performed better than the mean. The three values shown for % success of simulation 2 are for the three simulations of 100 iterations each.

		Simulation						
Parameter	Sea-age	1	2	3	4	5	6	7
PROP	1SW	0.15–0.55	0.15–0.55	0.15–0.55	0.15–0.55	*0.10–0.90	0.15–0.55	0.15–0.55
	2SW	0.8–0.99	0.8–0.99	0.8–0.99	0.8–0.99	0.8–0.99	0.8–0.99	0.8–0.99
	3,4SW	0.95–0.99	0.95–0.99	0.95–0.99	0.95–0.99	0.95–0.99	0.95–0.99	0.95–0.99
EST	1SW	0.0–0.1	0.0–0.1	0.0–0.1	0.0–0.1	0.0–0.1	0.0–0.1	0.0–0.1
	MSW	0.0–0.6	0.0–0.6	0.0–0.6	0.0–0.6	0.0–0.6	0.0–0.6	0.0–0.6
EFF	1,MSW	*0.015	0.009–0.023	*0.005–0.027	0.009–0.023	0.009–0.023	0.009–0.023	0.009–0.023
SMORT	1SW	0.9–0.99	0.9–0.99	0.9–0.99	0.9–0.99	0.9–0.99	0.9–0.99	*0.45–0.99
	MSW	0.6–0.99	0.6–0.99	0.6–0.99	0.6–0.99	0.6–0.99	0.6–0.99	*0.3–0.99
MORT	1SW	0.1–0.3	0.1–0.3	0.1–0.3	0.1–0.3	0.1–0.3	*0.05–0.60	0.1–0.3
	MSW	0.1–0.2	0.1–0.2	0.1–0.2	0.1–0.2	0.1–0.2	*0.05–0.40	0.1–0.2
SFA	1SW	0.4–0.6	0.4–0.6	0.4–0.6	0.4–0.6	0.4–0.6	*0.2–0.60	0.4–0.6
	MSW	0.7–0.95	0.7–0.95	0.7–0.95	0.7–0.95	0.7–0.95	*0.35–0.95	0.7–0.95
GRLD	1SW	0.4–0.8	0.4–0.8	0.4–0.8	0.4–0.8	0.4–0.8	*0.2–0.95	0.4–0.8
	MSW	0.6–0.9	0.6–0.9	0.6–0.9	0.6–0.9	0.6–0.9	*0.3–0.95	0.6–0.9
% Success		72	62,67,62	69	71	34	61	70

Each iteration generated 20 years of data, the same number of years of observed data used in the forecast model. We used 100 iterations, which produced 2000 (20 × 100) populations, to test the effect of each pre-season simulation experiment on the forecast model. We examined whether or not 100 iterations were sufficient by performing 300 iterations of pre-season simulation 2, and examining the change in percentage of cases when PDF had lower residual sum of squares than MEAN forecasts after each 100 iterations. There was little change in the success rate of PDF versus MEAN in each set of 100 iterations (Table 1) and because each set of 100 iterations was the result of 2000 simulated populations we concluded that this number was sufficient to provide stable results for each experiment.

In-Season Analysis

Observed data — In-season forecasts were made for early, late, and total seasons as defined by returns estimated for each standardized week of the season. Standardized weeks number the weeks of the year beginning with week 1 from January 1–7 for each year (Ash and O'Connell 1987). Thus, the early-run begins at week 20 (begins May 14) and ends week 34 (ends August 26). The late-run begins at week 35 (begins August 27) and ends week 43 (ends October 28) (Fig. 3). A preliminary analysis demonstrated that in-season forecasts for MSW salmon required separate forecasts for early and late runs but that an in-season forecast for 1SW salmon could be obtained by using the total season. Average late 1SW salmon returns are a relatively small percentage of total 1SW salmon returns, while late MSW salmon returns sometimes account for half the returns in a year (Fig. 3 and Moore et al. 1991).

Weekly run-timing was available for Millbank return estimates from 1974-1990 but timing of estuarine harvest had to be estimated for most years. Run-timing data from 1981-1983 (Claytor; unpublished data) indicated that catches in the estuarine fishery coincided with those at Millbank and all estuary harvest occurred during the early season. As a result, all estuary catches were placed into the early season in direct proportion to the returns at Millbank for all years.

Probability distribution functions — In-season forecasts using probability distribution functions followed the same procedures described previously and outlined by Noakes (1989). In this procedure cumulative returns in a season forecast salmon yet to return. The forecast of salmon yet to return is then added to the cumulative returns to date to produce an in-season forecast. A final forecast for the season is obtained by weighting the relative confidence of the pre-season and in-season forecasts as shown below:

$$\begin{aligned} \text{Final forecast} &= (w \times \text{in-season forecast}) \\ &+ (1 - w) \times \text{pre-season forecast} \end{aligned} \tag{3}$$

where, w, the weight, is calculated based on the 95% confidence intervals of the in-season and pre-season forecasts as:

$$w = \frac{A}{(A + B)} \tag{4}$$

where A is the width of the 95% confidence interval for the pre-season forecast, and B is the width of the 95% confidence interval of the in-season yet to return forecast. Thus, as the

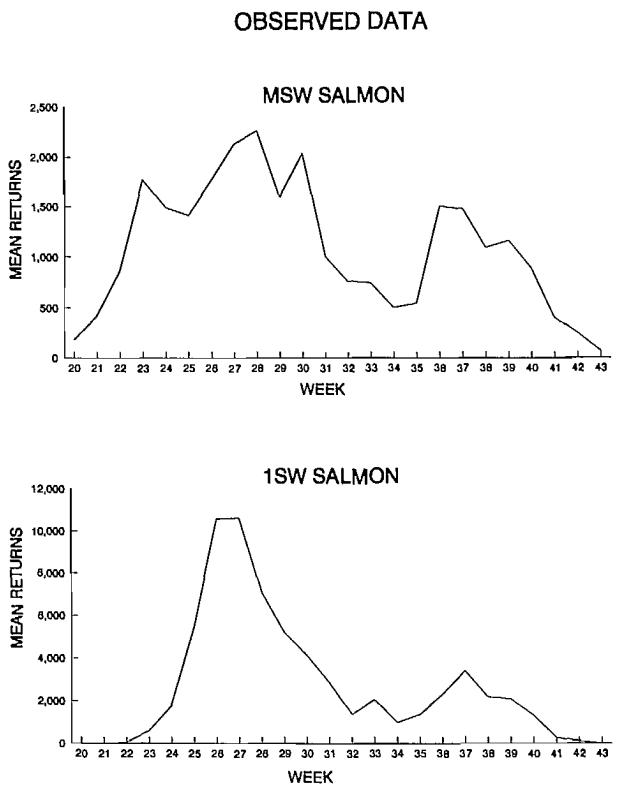


Fig. 3. Average run-timing of MSW and 1SW salmon to the Miramichi River by standardized week.

season progresses, B will become smaller than A and the in-season forecast will have increasingly more weight than the pre-season forecast in determining the final forecast (Noakes 1989).

In-Season Model Comparisons

Observed data — We compared four models for forecasting the probabilities of in-season returns:

1. MEAN, the mean number of salmon returning in a season;
2. FINAL, the final forecast of equation 3;
3. IN-SEASON, the in-season portion of equation 3; and
4. PERCENT, the cumulative returns to date divided by the mean percentage of returns to date.

Jackknife procedures as described previously for the pre-season forecasts were used to calculate all mean values and to evaluate the performance of each of the four models. Jackknife residual sum of squares and pairwise comparisons among the differences between the absolute values of the residuals from these methods measured the relative error of one method versus another. Differences in absolute residuals were plotted against the percentage of the returns in a given week to determine if any of these methods were biased over the range in observed run-timing. For example, we wished to know if some methods were prone to greater error if a larger or smaller than average percentage of total returns was seen up to week 28; that is if returns were earlier or later than expected.

Monte Carlo experiments: In-season simulated data: — To test the in-season methods we conducted two Monte-Carlo simulation experiments. These experiments simulated in-season return patterns that were slightly outside the extremes in run-timing in the observed data. We simulated a 15 week season which corresponded to the length of the early runs examined in the observed data (weeks 20 to 34). Instead of examining the forecasting capabilities of cumulative returns for each week, as was done with the observed data, data were simulated only for week 28 (begins July 9). We chose this week because it was the first one which had an appreciable reduction in residual sum of squares for the in-season early-run forecast model (see results below). We used the same criteria to evaluate the performance of the model as described previously, jackknife forecasts for each year using the four models described above (MEAN, IN-SEASON, FINAL, and PERCENT) at week 28 and comparing them to the absolute differences in residuals between each pair of methods as described above for the observed data.

The first simulation experiment selected percentage returns for each of the 15 weeks from a normal distribution with mean at week 27, the week when on average approximately half the salmon have entered the river, and a standard deviation set so that annual run patterns resembled those in the observed data. This strategy produced a range of percentages for week 28 that were slightly narrower than those in the observed data (see results below).

The second simulation experiment selected percentage returns from a uniform distribution ranging from 0 to 1. These percentages were then applied to the simulated week 28. This strategy produced a range of percentages for week 28 that were somewhat greater than those in the observed data (see results below).

As a result, these two simulation experiments allowed us to examine the behaviour of each of the four potential forecast models over a wide range of possible run-timings and determine which method would in general provide the best in-season forecasts.

Results

Pre-Season Model

Observed data — The observed data indicated a linear relationship between MSW and 1SW salmon returns at lower levels of 1SW salmon, up to about 50,000, but fewer MSW salmon returns per 1SW salmon may be expected at higher levels and overall the relationship was non-linear (Fig. 4). The ratio of 1SW:MSW salmon ranged from 1:1 to 7:1 with a maximum of 120,000 1SW salmon and 50,000 MSW salmon returning to the Miramichi River since 1971 (Fig. 4).

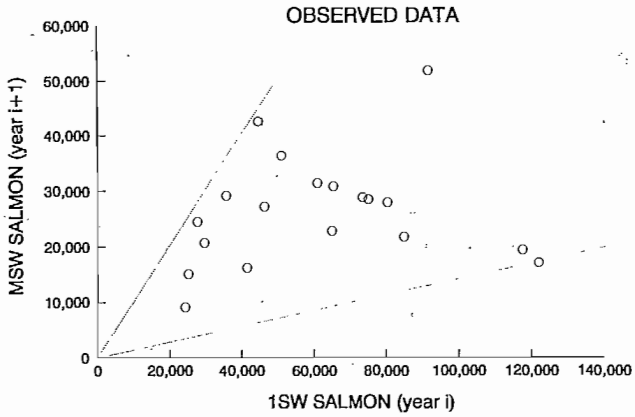


Fig. 4. Scatterplot of observed 1SW and MSW salmon returns to the Miramichi River. Upper dotted line represents 1:1 1SW:MSW salmon ratio and lower dotted line 7:1.

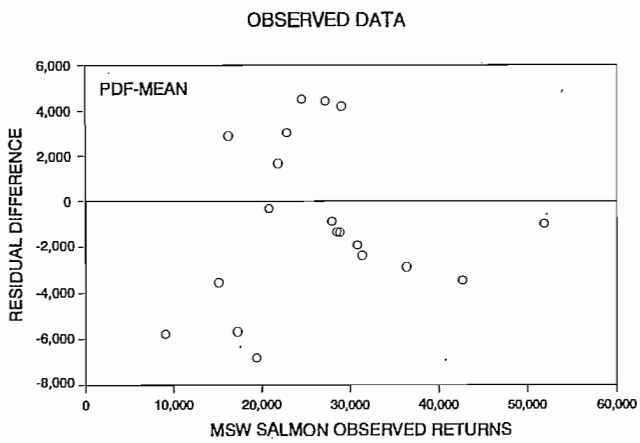


Fig. 5. Absolute residual differences between PDF and MEAN pre-season forecast models using observed data.

PDF residual sum of squares were 20% lower than those using MEAN. The 1977 MSW salmon returns accounted for about 40% of the PDF and MEAN residuals. When this year was removed from both analyses, PDF residuals were 30% lower than MEAN residuals in forecasting returns. Thus, this year was not unduly influencing conclusions regarding the most appropriate model to use and it was decided not to remove outliers from the data sets. There were also no apparent positive or negative residual trends over time with PDF. Jackknife standard deviations were similar for all years. As a result, residual patterns standardized by this statistic were similar to non-standardized residuals, and non-standardized residuals were used throughout this analysis.

Trends in absolute residual differences indicated poorer performance of the MEAN model at low and high observed MSW returns while PDF performed worse than MEAN at intermediate returns (Fig. 5). Positive absolute residuals indicate cases where PDF residuals were further from observed values than MEAN residuals, while negative values indicate worse performance by MEAN.

The reason for the lower residual sum of squares for MEAN compared to PDF are a result of the relatively poorer performance of MEAN at returns <20,000 MSW salmon. At these returns levels, MEAN absolute residual differences were $< -6{,}000$, while for observed returns >20,000 MSW salmon, absolute residual differences were always within $\pm 4,000$ returns for both models (Fig. 5).

Monte Carlo simulation experiments — Primarily linear effects and the least variation with all other parameters held constant were produced by **smort**. All sea mortalities combined (**mort**, **sfa**, **grld**) produced primarily linear effects but with increasing variation at larger 1SW returns with all other parameters held constant. Greater variation at increasing 1SW returns with slightly non-linear effects were produced by **eff** (Fig. 6). Uncertainty in estuarine harvest estimates had little influence on the relative proportions of 1SW and MSW salmon in the population. The 1SW:MSW salmon ratios ranged from 0.5 to 1.5 at a 20% error rate in estuarine harvest estimates, and from 0.5 to 1.8 by increasing the variation to 40%.

The greatest variability in our simulated system, from a single parameter, resulted from the limits we placed on **prop**. It produced the greatest non-linear effects and was the only parameter which produced the complete range of 1SW:MSW salmon ratios seen in the observed data with all other parameters held constant (Fig. 6). Other parameter combinations, however, did produce the 1SW:MSW ratios observed in the data. For example, increased variability in all sea mortalities (**mort**, **sfa**, **grld**) with all other parameters, including **prop**, held constant, also produced the complete range of 1SW:MSW salmon ratios seen in the observed data (Fig. 6). Similarly, with **eff** and **prop** held constant but all other parameters varying as in pre-season simulation 2, the complete range of 1SW:MSW salmon ratios was produced (Fig. 6).

When all parameters were allowed to vary at pre-season simulation 2 levels (Table 1) the pattern most closely resembled that observed with only **prop** varying (Fig. 6) but had greater variation in 1SW:MSW ratios than in the observed data (Figs. 4, 6).

PDF performed better than MEAN and there was very little difference in the success of PDF in forecasting returns under conditions of increasing uncertainty in estimation parameters, pre-season simulations 1–4, 6, and 7 (Table 1). In each of these cases, the greatest difference between the two models was the relatively poorer performance of MEAN, compared to PDF, at the extremes of known simulated returns; particularly those <20,000 (Fig. 7). PDF, however, was slightly worse than MEAN at intermediate return levels (30,000 MSW salmon) (Fig. 7). This pattern was similar to residuals produced by the analysis of observed data (Fig. 5). The exception to this trend was at very high returns (>90,000), where sample sizes were relatively fewer in each return size category, and there was little difference between the two models (Fig. 7). Increasing the variation in **prop** (pre-season simulation 5) was the only experiment in which PDF performed worse than MEAN (Table 1). This decline in performance occurred even though the ranges of the uniform distribution for the proportion of the

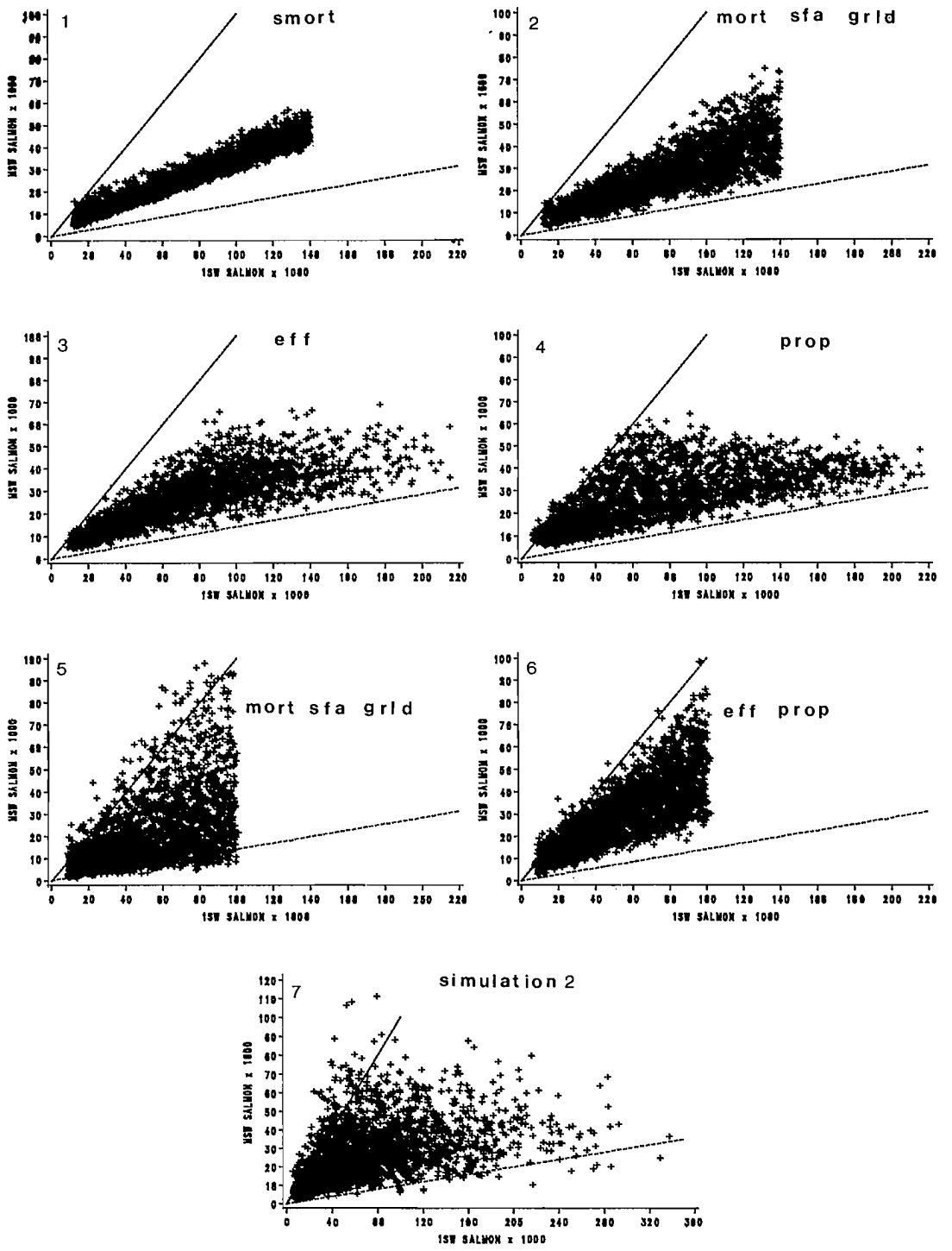


Fig. 6. Variation in simulated data from 100 iterations produced by allowing designated parameters to vary and keeping all other parameters constant (1–4). Variation as in pre-season simulation 6 (Table 1) and keeping all other parameters constant (5) and in maintaining **eff** and **prop** constant but letting all other parameters vary (6). Variation as in pre-season simulation 2 (Table 1) (7). Upper dotted line represents 1:1 1SW:MSW salmon ratio, lower dotted line represents 7:1 1SW:MSW salmon ratio.

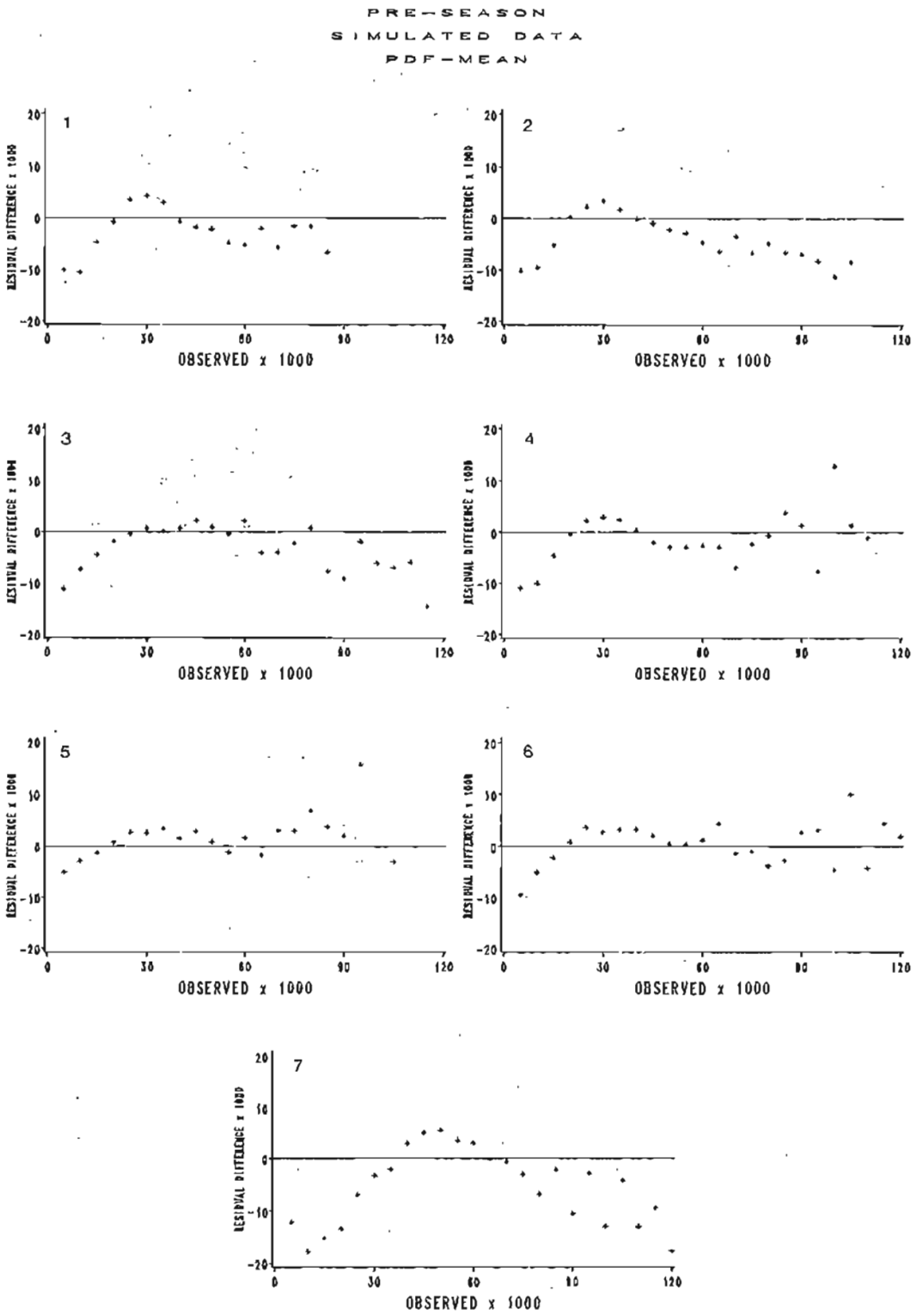


Fig. 7. The mean absolute residual differences by return size group between PDF and MEAN models from each of the pre-simulation experiments in Table 1. Numbers in upper left hand corner refer to pre-season simulation experiments.

older sea-ages did not change from other simulations (Table 1). PDF forecasts were farther from the known simulated returns at almost all levels above 30,000 in pre-season simulation 5 (Fig. 7). Thus, the pre-season PDF model performance was robust to parameter ranges estimated from research data and all expanded ranges except for **prop**.

In-Season

Observed data — IN-SEASON, FINAL, and PERCENT residuals were 40–75% below MEAN residuals by week 28 (begins July 9) for early MSW salmon and total 1SW salmon and by week 37 (begins September 10) for late run MSW salmon. The exception to this trend was for late run MSW salmon, where PERCENT residuals were not below MEAN residuals until week 38 (Fig. 8). We did not consider absolute differences in residuals between MEAN and the other models in further in-season analyses because of its poor performance on the observed data.

We next examined the behaviour of these models at the critical weeks described previously for each season: week 28 for early MSW salmon and total 1SW salmon and week 37 for late MSW salmon. IN-SEASON and FINAL models use cumulative catch to date to forecast fish yet to return. In each of the seasons examined, there was a trend for increased cumulative catch to date to correspond to increased number of fish yet to return in the observed data (Fig. 9). Exceptions to this trend were most numerous in the late MSW salmon returns (Fig. 9). Early MSW salmon run-timing (as percentage of total returns by week 28) had a pattern which most approximated a normal distribution (Fig. 10), while late MSW salmon percentages more closely resembled a uniform distribution and had a much wider range than either early MSW salmon or total 1SW salmon returns (Fig. 10).

Examination of the difference in absolute residuals indicated there was little difference between PERCENT and IN-SEASON. The exception were the residuals for years in which a large portion of the run had returned by week 28, (high percentages) for late MSW salmon. In these cases, PERCENT absolute residuals were much higher than IN-SEASON absolute residuals (Fig. 11). The residuals for 1SW salmon had similar, but less pronounced large values at high percentages (Fig. 11). For early MSW salmon the greatest difference in absolute residuals occurred between PERCENT and FINAL;

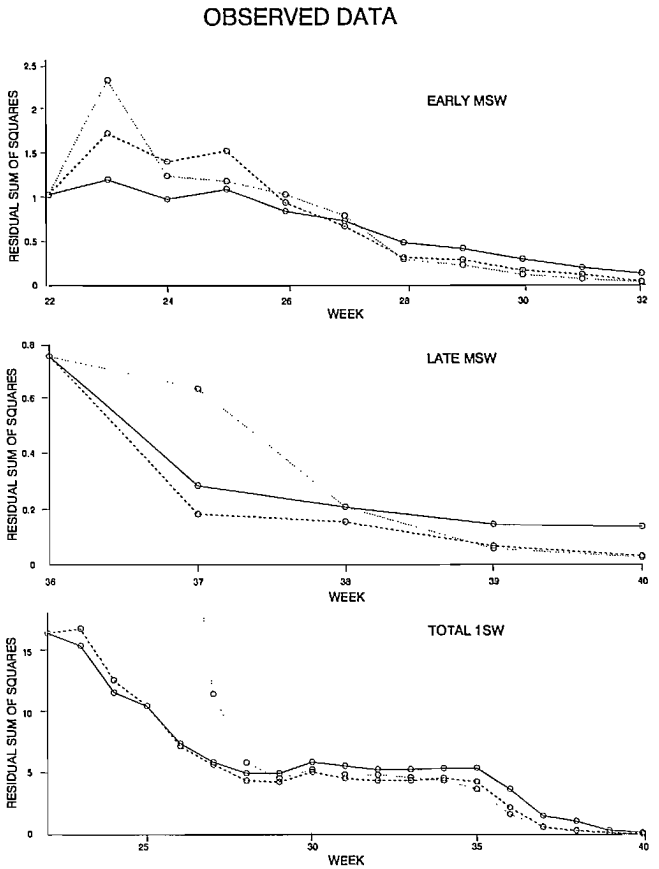


Fig. 8. Decline in pre-season MEAN residual sum of squares using in-season forecast models for early and late MSW salmon runs and the total 1SW salmon run using IN-SEASON, FINAL, and PERCENT models. The residual sum of squares on the Y-axis are times 10^9. The Y-axis intercept represents the MEAN residual sum of squares, the solid line the FINAL, the dashed line the IN-SEASON, and the dotted line the PERCENT residual sum of squares.

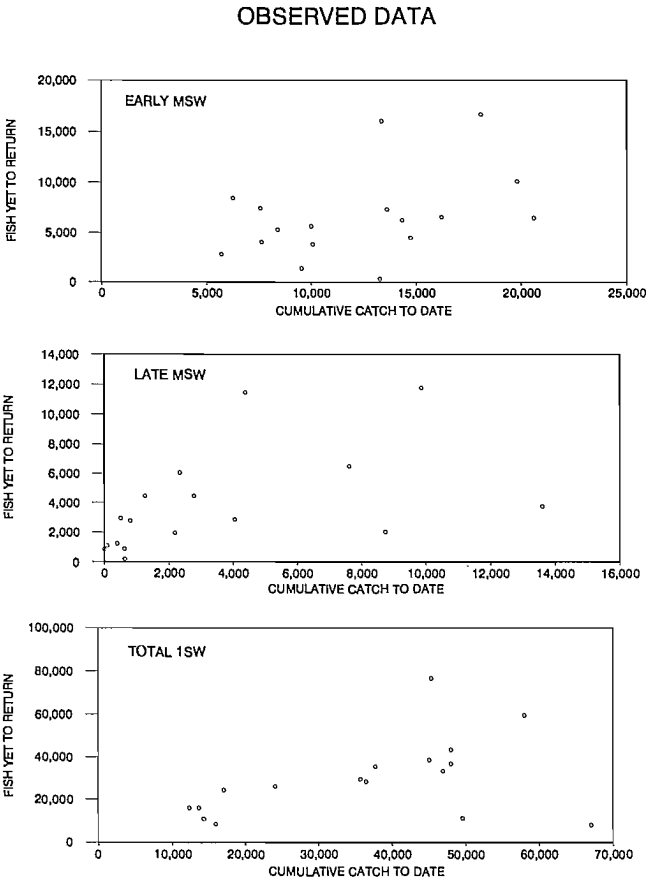


Fig. 9. Observed cumulative catch to date and fish yet to return data used in IN-SEASON and FINAL models for MSW and 1SW salmon runs. Week 28 was used for early MSW and total 1SW and week 37 for late MSW salmon analyses.

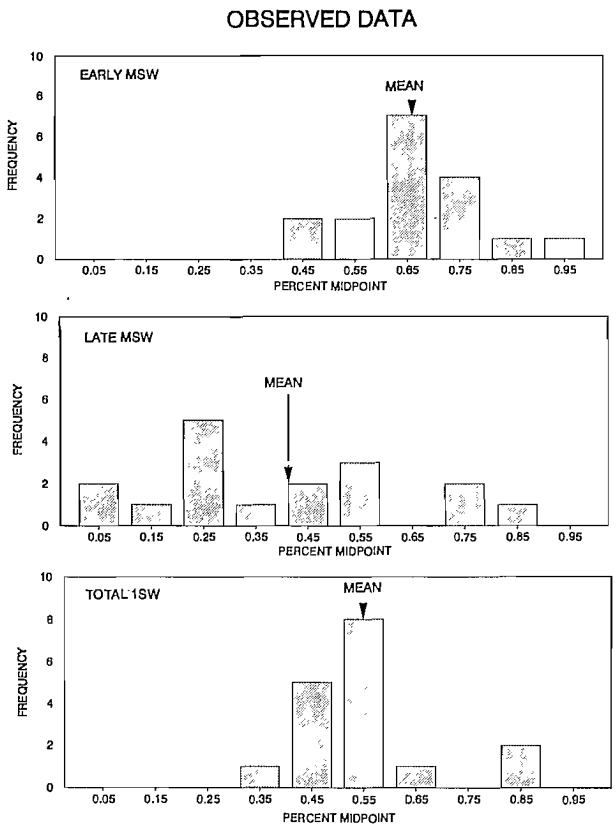


Fig. 10. The distribution of observed annual percentages that the cumulative returns to date were of the total returns for MSW and 1SW salmon runs in the observed data. Week 28 was used for early MSW and total 1SW and week 37 for late MSW salmon analyses.

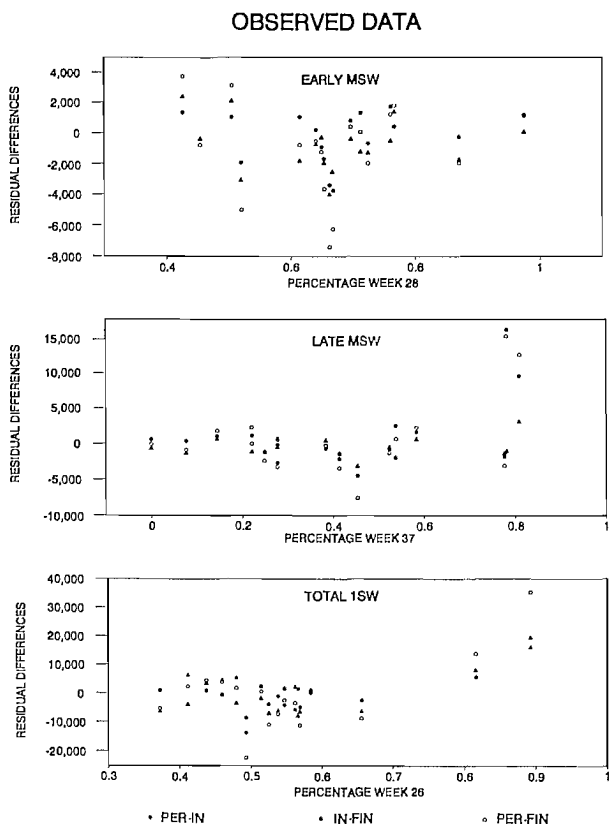


Fig. 11. The differences in absolute residuals for observed data between the PERCENT-IN-SEASON (PER-IN, solid circle), IN-SEASON-FINAL (IN-FIN, solid triangle), and PERCENT-FINAL (PER-FIN, open circle) models for week 28 of the early MSW salmon run, week 37 for the late MSW salmon run, and week 28 for the total 1SW salmon run.

at low percentages, PERCENT had higher absolute residuals but at intermediate percentages FINAL had higher absolute residuals. These results on the observed data indicate that PERCENT and FINAL will tend to be more biased at run-timing extremes than IN-SEASON.

In-Season

Monte-Carlo simulation experiments — The MEAN model, in addition to performing worse than the other three models on observed data, also had the worst model performance in the simulation experiments (Table 2), and, as for the observed data, was not considered in the residual analyses.

The first simulation experiment examined the behaviour of these models with very little variation in run-timing. Although the percentages were selected from a normal distribution, when they were plotted on the same scale as the observed data they more closely resembled a uniform distribution and had a narrower range than that in the observed data (compare Figs. 10, 12). As a result, the relationship between cumulative returns to date and fish yet to return was more linear than in the observed data (compare Figs. 9, 12). IN-SEASON had lower residual sum of squares than PERCENT in 20 of the 100 trials under these conditions (Table 2). Nevertheless, absolute

Table 2. Number of times out of 100 trials that method defined by column had lower residual sum of squares than forecast method defined by row with in-season simulations.

	Method			
	MEAN	IN-SEASON	FINAL	PERCENT
		In-Season Simulation 1		
MEAN	–	87	87	100
IN-SEASON	13	–	13	80
FINAL	13	87	–	100
PERCENT	0	20	0	–
		In-Season Simulation 2		
MEAN	–	75	94	19
IN-SEASON	25	–	25	8
FINAL	6	75	–	19
PERCENT	81	92	19	–

residual differences between these methods were relatively small and IN-SEASON and PERCENT absolute residuals were similar over the entire range of percentages (Fig. 12). In contrast, FINAL did much worse than either IN-SEASON or

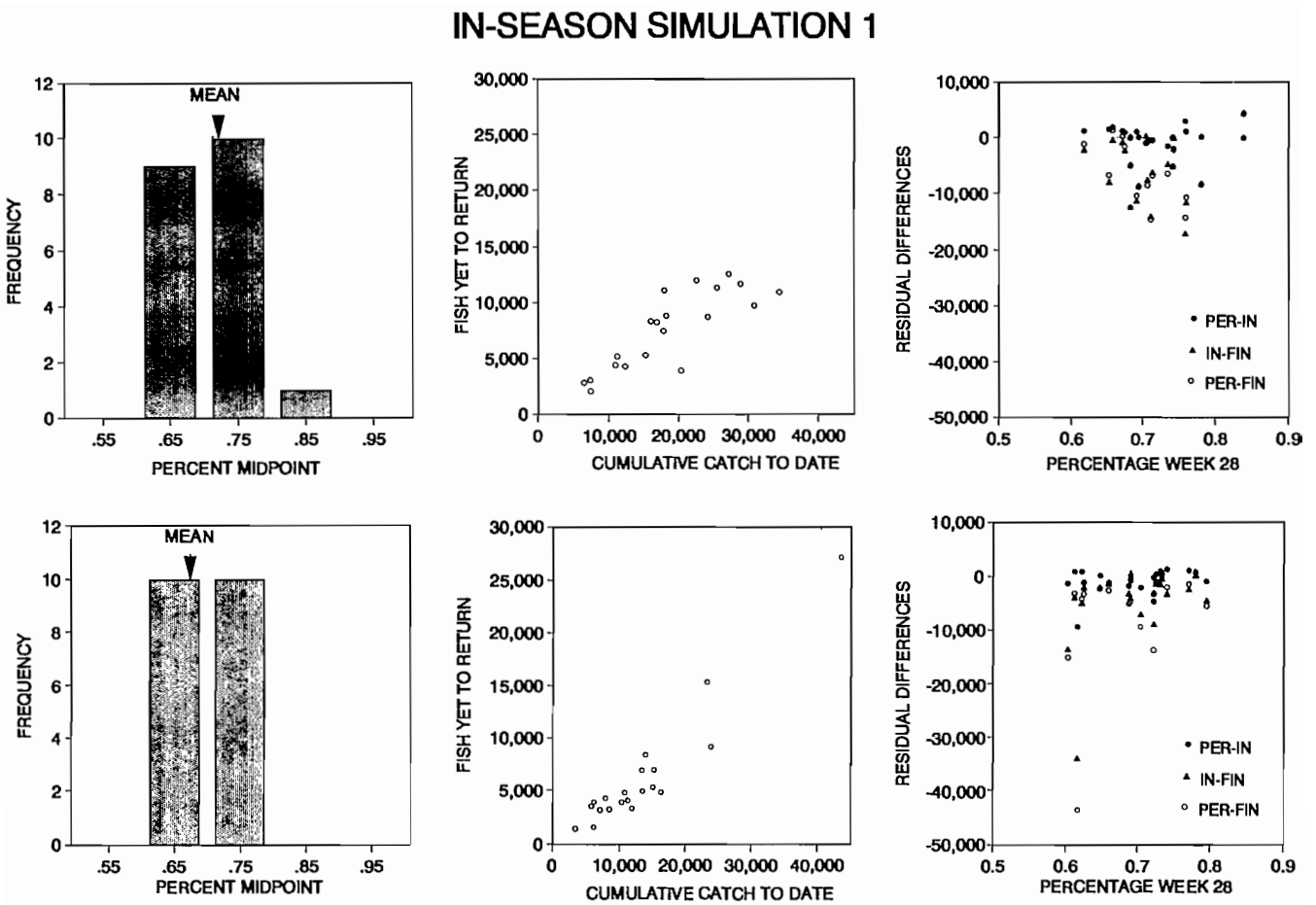


Fig. 12. Examples of results from two iterations, each simulating 20 years of data, from in-season simulation 1. The distribution of annual percentages that the cumulative returns to date were of the total returns, the cumulative catch to date and fish yet to return scatterplots, and the differences in absolute residuals for PERCENT, IN-SEASON, and FINAL models are illustrated.

PERCENT (Table 2) and tended to have higher absolute residuals than both these methods (Fig. 12).

In the second in-season simulation experiment the variation in run-timing was greater than that in the observed data (compare Figs. 10, 13). The wide range in percentages produced a negative relationship between cumulative returns to date and fish yet to return (Fig. 13). PERCENT had higher residuals at extreme percentages than IN-SEASON and FINAL, while FINAL had the highest absolute residuals at intermediate percentages (Fig. 13). Under these conditions, IN-SEASON performed the best of the four methods and MEAN performed better than PERCENT (Table 2).

Discussion

PDF performed better than MEAN in all pre-season simulations with realistic parameter ranges and in all experiments with expanded parameter ranges except **prop** in simulation 5. PDF is therefore robust to considerable uncertainty in marine mortality, exploitation rates, and post-spawning mortality and 1SW:MSW ratios beyond the range of the observed data. Jackknife procedures support this conclusion by removing some of the bias that may result from including observations to be forecasted in the forecast model. We conclude that PDF should continue to be used in pre-season forecasting salmon returns to the Miramichi River.

Simulation experiments identified **prop** as an important parameter with respect to the forecast model and research designed to understand the factors controlling this parameter are likely to be productive in leading to improved advice on the Miramichi River salmon stock. Experimental evidence will be essential in understanding the importance of this parameter in determining 1SW:MSW ratios. The simulation experiments indicate there are a wide variety of potential explanations for the observed range in 1SW:MSW ratios (Figs. 6) and determining the relative importance of natural mortality, fishing mortality, and **prop** is likely to be difficult.

Simulation experiments have identified possible parameters to investigate if the model should fail. For example, changes in the proportion of repeat spawners or fishery exploitation rates are not likely to cause model failure because large changes in these parameters had little effect on model performance (Table 1, Fig. 7). We would not have been able to determine the relative influence of these parameters in any other way. Comparing model results to known populations

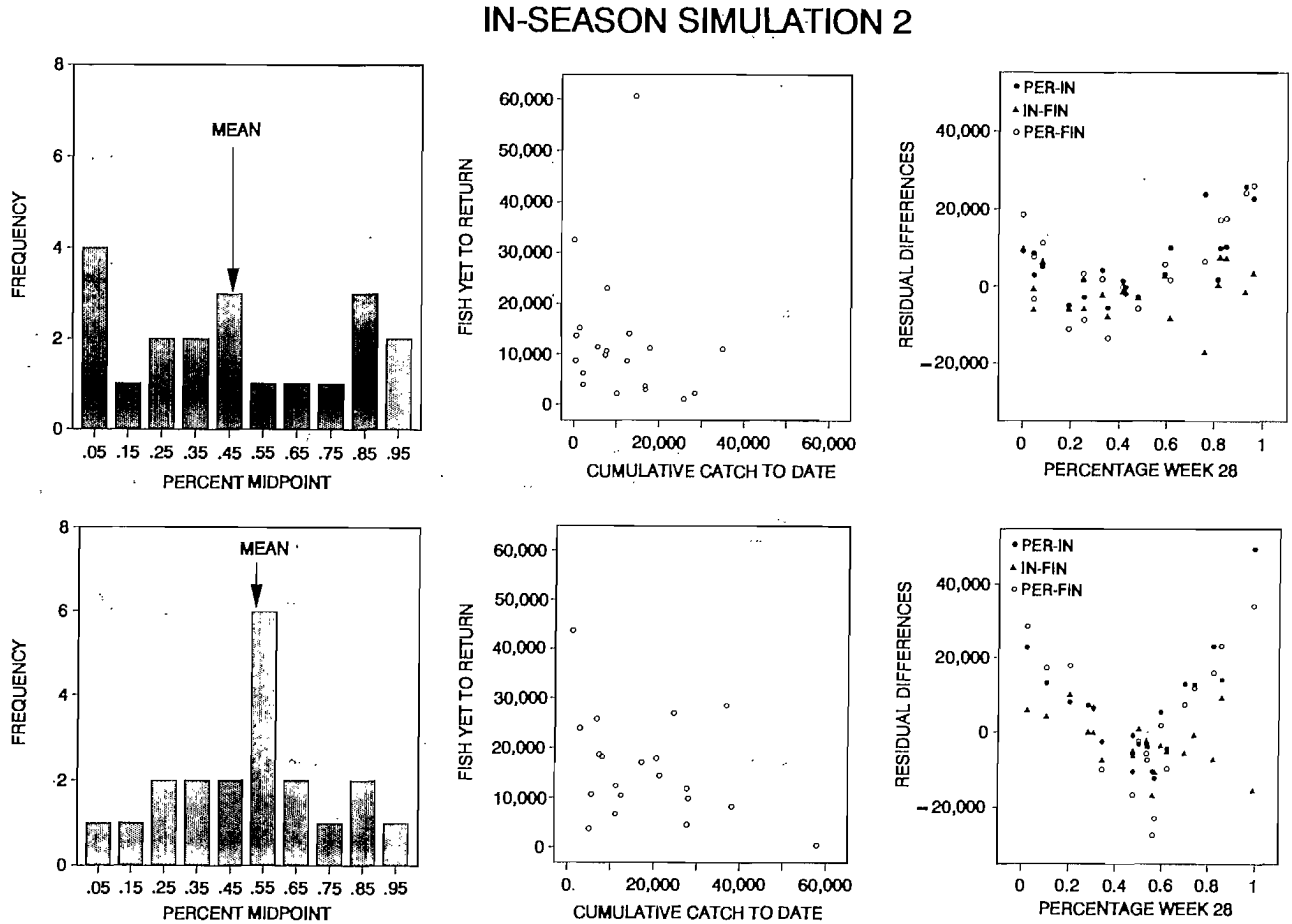


Fig. 13. Examples of results from two iterations, each simulating 20 years of data, from in-season simulation 2. The distribution of annual percentages that the cumulative returns to date were of the total returns, the cumulative catch to date and fish yet to return scatterplots, and the differences in absolute residuals for PERCENT, IN-SEASON, and FINAL models are illustrated.

in a series of simulation experiments is likely to be a useful tool for evaluating stock assessment models before advice is generated.

In-season forecasts are likely to play an important role in future management and in the provision of advice for the Miramichi River salmon stock. In-season forecasts reduce pre-season forecasting error and can be made within sufficient time to allow managers to adjust pre-season regulations if returns well below or above pre-season expectations are indicated (Fig. 8).

We suggest that IN-SEASON is the preferred method for providing in-season forecasts. It performed better than MEAN and FINAL with both observed and simulated data and better than PERCENT with the observed data. PERCENT was very unstable in the early weeks of the season and the IN-SEASON residual sum of squares decreased below those for MEAN before PERCENT, although in later weeks there was very little difference among these models (Fig. 8). In in-season simulation experiment 1, PERCENT had slightly lower residual sum of squares than IN-SEASON (Table 2) but there were few differences in absolute residuals (Fig. 12). IN-SEASON performed much better than PERCENT when run-timing was more variable, as in in-season simulation experiment 2. In this experiment IN-SEASON had lower residual sum of squares (Table 2) and less bias at run-timing extremes (Fig. 13) than PERCENT. Thus, IN-SEASON appears to perform better than the other models over a wider range of conditions than any of the other models tested.

These results suggest a two-step process in the provision of forecasting advice. The first would be to provide pre-season advice on the numbers of MSW salmon expected to return in year $i+1$. The second would be to provide in-season updates on expected MSW and 1SW salmon returns.

Advice on salmon stocks is typically provided with respect to a target spawning stock biomass. For the Miramichi River salmon, spawning targets of 23,600 MSW salmon and 22,600 1SW salmon have been established for the total population, early and late returns combined (Randall 1985). Managers typically are interested in having advice on the probability of exceeding or coming within a certain range of these targets. Probability distributions can easily provide this information in graphical form.

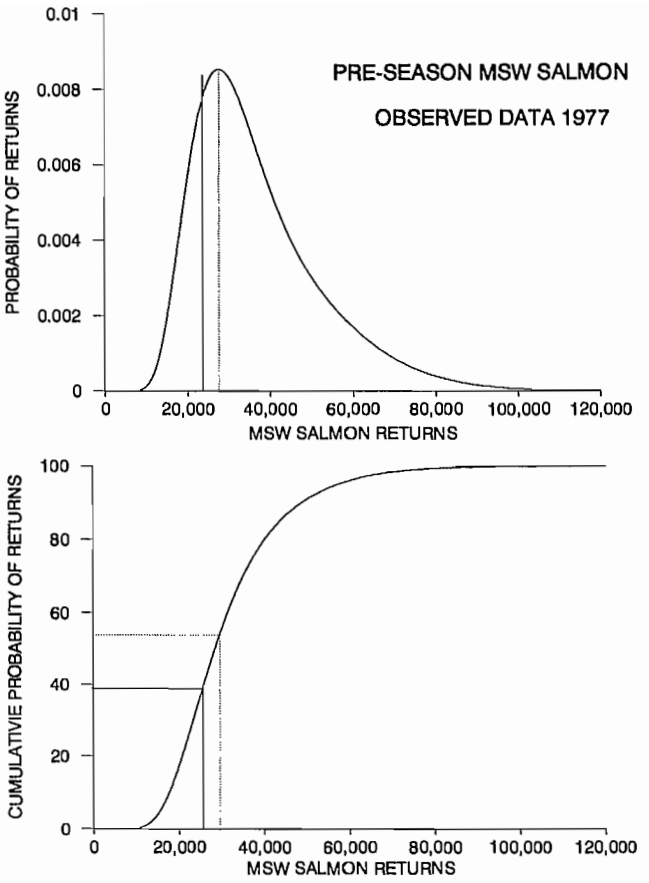


Fig. 14. The probability distribution and cumulative probability curve for the pre-season MSW salmon forecast for 1977. The solid line represents the target spawning biomass and the dotted line the forecast.

We have chosen 1977 as an example to illustrate how advice could be provided with these models and the improvement expected by updating pre-season forecasts with in-season run-timing data because it had the highest residuals for pre-season forecasts. The pre-season forecast for total MSW salmon returns, early and late runs combined, for 1977 would have been about 25,000 MSW salmon with a 61% probability of exceeding target spawning requirements for the total run (Fig. 14).

An early run in-season forecast, at week 28, would have been about 25,000 MSW salmon with a 76% probability of exceeding the target spawning requirements during this run (Fig. 15). The advice would have been that there seems to be a high probability that the pre-season forecast and spawning requirements will be met by the early run. Thus, a manager may choose to manage conservatively and maintain the pre-season management plan, under the assumption that late-run returns will be low or assume that the late-run will be at least average and allow increased harvest.

The situation for management improves as late run MSW salmon forecasts become possible. The late run in-season forecast, at week 37, would have been about 18,000 MSW salmon (Fig. 15). These values combined with the early run returns of 34,000 MSW salmon would have provided a combined forecast of 52,000 MSW salmon for 1977.

Advice on total 1977 1SW salmon returns, based on week 28 would have been 25,280 with a probability of exceeding target spawners (22,600) of 61% (Fig. 15). Pre-season advice for 1SW salmon total returns in 1977 based on MEAN would have been 68,171; considerably higher than observed and PDF forecasted returns. Total 1SW salmon returns for 1977 were 28,000 salmon.

The 1977 example is one in which in-season updates indicated that MSW salmon returns might be well above pre-season expectations but 1SW salmon returns may be well below pre-season expectations. As a result of in-season updates managers would have been able to consider increasing gains in yield from higher than expected MSW returns. Similarly, they would also have been able to consider reducing allocations of 1SW salmon because in-season updates indicated lower than expected returns of this age-class.

PDF forecasts provide an efficient and flexible method of incorporating uncertainty into biological advice. However, providing forecasts is only the first step in helping managers determine the expected gains and losses of various management decisions. The probability associated with achieving a given level of returns could provide input data for formal risk analyses. These probabilities could be used to estimate the expected gains and losses in yield that might occur from a variety of management options. Quantifying these expectations

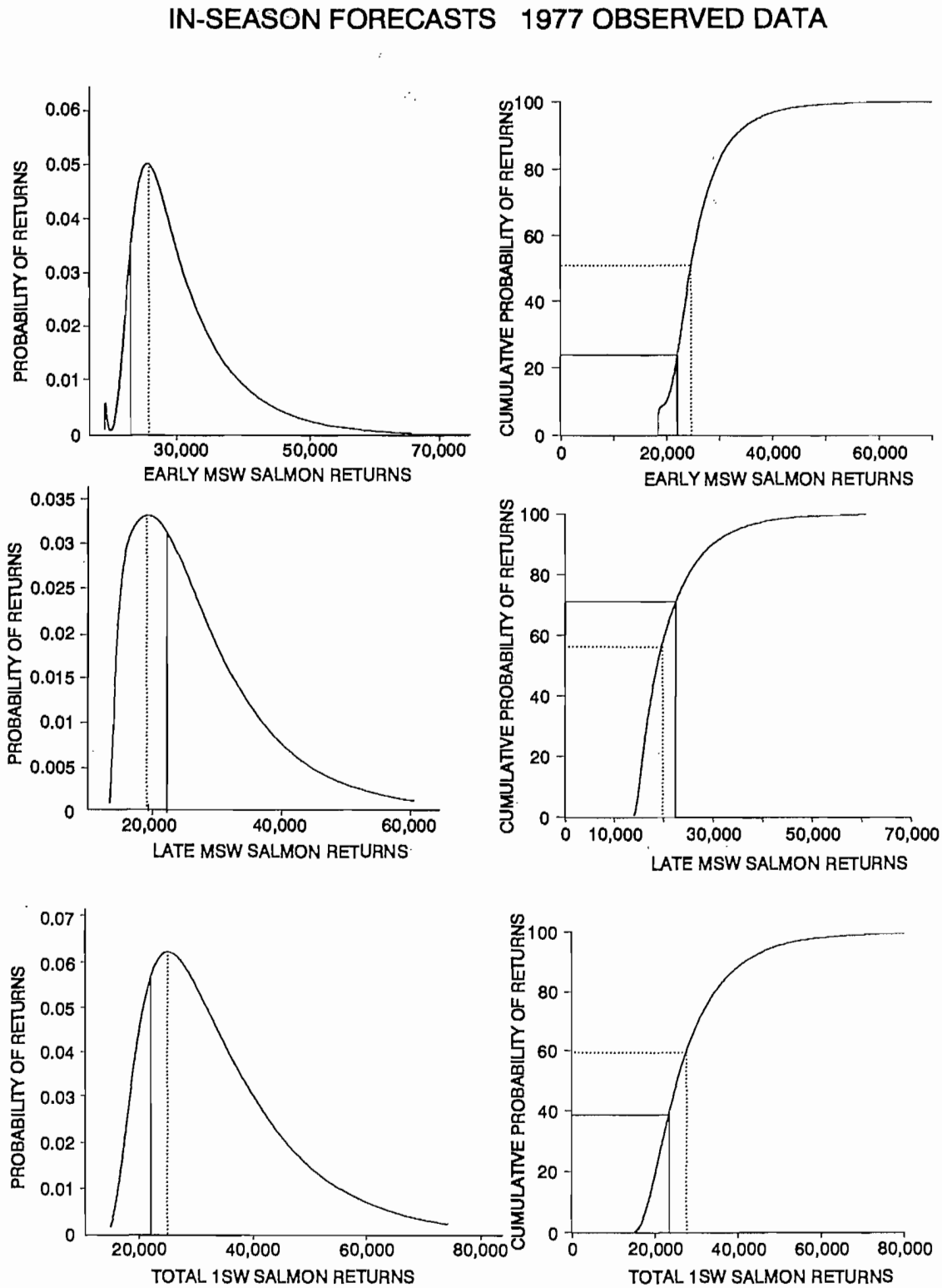


Fig. 15. The probability distributions and the cumulative probability curves for the in-season forecasts of early MSW salmon, late MSW salmon, and total 1SW salmon for 1977. The solid line represents the target spawning biomass and the dotted line the forecast.

would improve the decision making capabilities of fisheries managers. Developing this type of decision making system for Miramichi River salmon assessments is the subject of future research.

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Monte Carlo Evaluation of Risks for Biological Reference Points used in New Zealand Fishery Assessments

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Two biological reference points are routinely calculated in New Zealand fishery assessments: the maximum safe constant catch, and the maximum safe constant fishing mortality. The rules used to calculate these quantities are not based on any formal definition of "safe". I propose the definition that a level of harvesting (either constant catch or constant mortality) should be considered safe if it maintains the spawning stock biomass above 20% of its mean virgin level at least 90% of the time. Monte Carlo procedures are used to evaluate this definition and to determine, for a range of life history parameter values, whether the rules for calculating the biological reference points are safe. With this definition, maximum safe harvesting levels are shown almost always to produce at least 90 % of the maximum mean catch obtainable with either constant catch or constant mortality harvesting. The safety of the rules is found to depend strongly on the extent of compensation in the stock-recruit relationship, less strongly on natural mortality, and least strongly on the extent of recruitment variability. Uncertainty about the true level of natural mortality decreases the acceptability of the rules.

Deux points de référence biologiques sont couramment calculés pour les évaluations de pêche en Nouvelle-Zélande : la prise constante sécuritaire maximale et la mortalité par pêche constante sécuritaire maximale. Les règles utilisées pour calculer ces valeurs ne reposent sur aucune définition officielle de «sécuritaire». Je propose la définition qu'un niveau de pêche (soit de prise constante ou de mortalité constante) soit considéré sécuritaire s'il maintient le niveau de biomasse génitrice au-dessus de 20 % de son niveau vierge moyen, au moins 90 % du temps. Les techniques de Monte-Carlo sont utilisées pour évaluer cette définition et pour déterminer, pour une série de valeurs de paramètre d'histoire de vie, si les règles de calcul des points de référence biologiques sont sécuritaires. À la lumière de cette définition, les niveaux de pêche maximum sécuritaires produisent presque toujours au moins 90 % de la prise moyenne maximale obtenue soit en situation de prise constante ou de mortalité par pêche constante. La sécurité des règles dépend étroitement de la portée de la compensation dans la relation stock-recrutement, moins étroitement de la mortalité naturelle et encore moins étroitement de la portée de la variabilité du recrutement. L'incertitude du niveau réel de mortalité naturelle diminue l'acceptabilité des règles.

In New Zealand stock assessments there are two biological reference points that are routinely calculated whenever there are sufficient data. These are called maximum constant yield (MCY) and current annual yield (CAY). Both are interpretations of the concept of maximum sustainable yield (MSY). The first interpretation stresses the word "sustainable" and is based on the idea of taking the same catch from the fishery year after year. Thus the MCY is defined as

> the maximum constant catch that is estimated to be sustainable, with an acceptable level of risk, at all probable future levels of biomass.

The second interpretation, which stresses the word "maximum", recognizes that fish populations fluctuate in abundance (for environmental and biological, as well as fishery, reasons) so that to maximize the average catch from a population it is necessary to vary the catch from year to year. Thus the CAY is defined as

> the one-year catch calculated by applying a reference fishing mortality, F_{CAY}, to an estimate of the fishable biomass present during the next fishing year; F_{CAY} is the level of (instantaneous) fishing mortality that, if applied every year, would, within an acceptable level of risk, maximise the average catch from the fishery.

(These definitions of MCY and CAY can be found, together with other related definitions, in Annala 1991).

In the current context the key phrase in the above definitions is "an acceptable level of risk". Though a number of rules have been developed for calculating MCY and CAY (see below), these rules have not so far been based on any formal definition of risk. In this paper I describe the rules currently in use, propose a definition of an acceptable level of risk, and address two questions arising from this definition. First, is acceptance of the definition likely to result in suboptimal yields? Second, under this definition are the rules currently used to calculate MCY and CAY safe or unsafe?

These questions were addressed by simulating fishing, using an age-structured model, with constant catch (to evaluate

the MCY rules), or with constant fishing mortality, F, (for the CAY rules). Given a definition of "an acceptable level of risk", these simulations show how safe each rule is. The simulations were repeated for five New Zealand species and a range of stock-recruit relationships.

The Rules

The MCY rules considered were MCY = $0.25MB_0$, MCY = $0.25F_{0.1}B_0$, and MCY = 2/3MSY, where M is the instantaneous rate of natural mortality, $F_{0.1}$ is the instantaneous fishing mortality for which the slope of the yield-per-recruit curve is 0.1 times the slope at the origin (Gulland and Boerema 1973), B_0 is the mean recruited biomass of the virgin population, and MSY is the deterministic maximum sustainable yield.

The CAY rules considered were $F_{CAY} = M$, $F_{CAY} = F_{0.1}$, and $F_{CAY} = F_{MSY}$, where F_{MSY} is the instantaneous fishing mortality associated with the deterministic MSY.

In particular applications, the deterministic MSY may be calculated either from a surplus production model or from combining a yield per recruit analysis with an assumed stock-recruit relationship. In this paper only the latter method was considered.

An Acceptable Level of Risk

The risk we are discussing here is the risk of fishery "collapse". By collapse I mean some change in the ecosystem so that sustainable yield levels after the change are much lower than they were before it. There are numerous examples of such collapses in the fisheries literature (see, e.g., Clark 1985, table 1.1; Thompson 1993, table 1). The reasons for these collapses are not always clear and a number of mechanisms may be responsible. Further, there is evidence that populations can collapse even in the absence of a fishery (Soutar and Isaacs 1974). However, it seems reasonable to assume, as a general principle, that the probability of collapse for a particular fish population will tend to increase as its spawning biomass decreases — i.e., risk increases as spawning biomass decreases. (Note that I distinguish between recruited biomass and spawning biomass. The former is the (natural) basis for the MCY rules of the preceding section; the latter is the natural quantity of concern when considering the viability of an exploited population.)

Thus, there should be, for each population, a threshold biomass level below which the risk of collapse is unacceptably high. It is natural to express this threshold level as a percentage of the mean virgin spawning biomass, S_0, and, for lack of evidence to the contrary, to assume that the threshold is the same (in percentage terms) for all populations. I propose following Beddington and Cooke (1983) in using a threshold of 20% S_0.

In using this threshold in the present context it is not useful to apply it absolutely — i.e., to insist that the level of harvesting should never result in reducing the spawning biomass, S, below 20% S_0. This is because, even with no fishing, there will always be a non-zero probability that a sequence of unusually low recruitments will cause S to drop below the threshold.

Thus I propose the definition that a harvesting regime (with constant catch, or constant F) has an acceptable level of risk if the probability that S falls below 20% S_0 is less than 0.1 (or, equivalently, the percentage of years in which S falls below 20% S_0 is less than 10%).

Model and Simulation Method

The age-structured model used in the simulations was the same as that described by Francis (1992) except for one modification: the parameters S_f and S_m were introduced to describe gradual recruitment and maturity (see Appendix for details).

In this model two parameters describe the stock-recruit relationship: the steepness of the Beverton and Holt relationship, h (= the mean recruitment at $S = 20\%\ S_0$, expressed as a fraction of the virgin recruitment) and recruitment variability, σ_R (= the standard deviation of the natural logarithm of the recruitment at any given stock size).

The aim was to estimate, for each level of harvesting (with either constant catch or constant F), the mean catch and the probability that $S < 20\%\ S_0$. Because no time frame is mentioned in the definitions of MCY and CAY, this probability is taken to be a long-term value — i.e., the probability calculated after the population has reached a (stochastic) equilibrium under the particular harvesting level.

The following initialization procedure was used to obtain approximate equilibrium starting conditions. First, I calculated f_r, the equilibrium recruited biomass (expressed as a fraction of B_0) associated with the given harvest level when recruitment is deterministic. Where the given harvest level was not sustainable with deterministic recruitment, f_r was set equal to the equilibrium recruited biomass (expressed as a fraction of B_0) associated with $F_{MSY}.f_u$ was then calculated as the recruitment (expressed as a fraction of the virgin recruitment) predicted by the (deterministic) stock-recruit relationship when $S = f_r S_0$. Next, a virgin population was generated assuming stochastic recruitment, where the mean (virgin) recruitment was calculated so that B_0, the expected recruited biomass, was always 100 t (note that the results of the simulations depend on relative, rather than absolute population sizes). Then the numbers at age in this population were multiplied by f_r (for recruited fish) or f_u (for unrecruited fish) to reduce the population to the approximate size expected for the given harvest level. Finally, the model was run for A_l years to stabilize, where A_l is the approximate maximum age of the species, defined by $A_l = \log_e(100)/M$.

The model was run for a further A_l years and the whole procedure was repeated 500 times. The mean catch (as a percentage of B_0), and the proportion of years in which $S < 20\%S_0$, (both calculated over the 500 A_l years of the simulations) were recorded.

Table 1. Scientific names, species codes, and life history parameters for the five species used in the simulations. (Full details of the population model are given in Francis (1992) and in an Appendix to the present paper.)

	Orange roughy *Hoplostethus atlanticus*	Tarakihi *Nemadactylus macropterus*		Alfonsino *Beryx splendens*		Southern blue whiting *Micromesistius australis*		Hoki *Macruronus novaezelandiae*	
		Male	Female	Male	Female	Male	Female	Male	Female
Sp. code	s_1	s_2		s_3		s_4		s_5	
L_∞ (cm)	42.5	42.1	44.6	51.1	57.5	47.6	51.5	90.3	97.6
K (yr^{-1})	0.059	0.21	0.20	0.11	0.08	0.35	0.32	0.24	0.21
t_0 (yr)	−0.35	−1.4	−1.1	−3.6	−4.1	−0.9	−1.0	−0.6	−0.9
M (yr^{-1})	0.05	0.10		0.20		0.21		0.30	0.25
a[1]	9.63	4.33	4.00	2.26		0.52	0.41	0.60	
b	2.68	2.77	2.79	3.02		3.09	3.15	2.85	
A_f (yr)	23	3		5		3	4	5	
S_f (yr)	3	1		0		1	1	1	
A_m (yr)	23	4		4		3	4	5	
S_m (yr)	3	0		0		1	1	1	
A_{max} (yr)	70	20		50		25		20	

[1] weight(kg) = a[length(cm)]^b × 10^{-5}.

Estimates of biomass are required for both the MCY and CAY rules. It was assumed that these estimates were unbiased and normally distributed with a coefficient of variation of 20% in all cases.

The MCY rules require an estimate of B_0. It was assumed here that what would actually be estimated for a real fishery would be the initial biomass, i.e., the biomass at the time the fishery started. This may be greater or less than B_0 depending on whether recent recruitment had been above or below average. Thus, in each simulation run with constant catch, the estimate of B_0 was taken as the size of the virgin population generated in the above initialization procedure, plus a random estimation error (as described in the previous paragraph).

Parameter Values

In order to be able to draw conclusions about the general applicability of the above rules for New Zealand stock assessments it was important that the simulations be carried out for an appropriate range of parameter values.

The most important parameters in this respect are those describing the stock-recruit relationship — h and σ_R. Since very little is known of either h or σ_R for New Zealand species, a range of values was used for each: $h = 0.5, 0.75, 0.95$ (describing low, medium, and high compensation); $\sigma_R = 0.4$, 0.6, and 0.8 (describing low, medium, and high recruitment variability). The latter values were chosen with reference to the compilation of estimated σ_R values given by Beddington and Cooke (1983); these values are approximately the quartiles of the values in their table 2 (note that they tabulate the variance, i.e., σ_R^2).

Other life history parameters that are likely to affect how safe the above rules will be are natural mortality, M, and the von Bertalanffy rate parameter, K. From a plot of the M and K values for 11 New Zealand species, five were selected that covered the range (Fig. 1). The life history parameters for these species are given in Table 1.

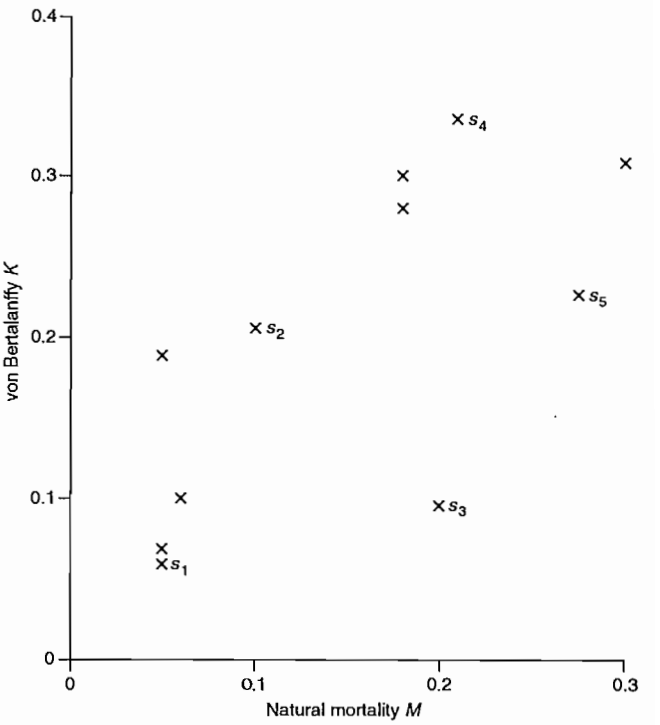


Fig. 1. Natural mortality and von Bertalanffy K for eleven New Zealand fish species. Those species used in the simulations are labelled (see Table 1 for further details).

Another aspect likely to affect the performance of the rules is whether fish recruit to the fishery before or after reaching

Table 2. Harvesting levels for each species for: A. the MCY rules (target catch expressed as a percentage of B_0); and B. the CAY rules (target fishing mortality in units yr^{-1}). (Full species names and life-history parameters are in Table 1. Parameters for s_2' and s_3' are the same as for s_2 and s_3 except that A_f has been set equal to A_m.)

Rule	Stock-recruit steepness, h	Species						
		s_1	s_2	s_2'	s_3	s_3'	s_4	s_5
A. MCY rules								
catch = $0.25MB_0$		1.3	2.5	2.5	5.0	5.0	5.3	6.9
catch = $0.25F_{0.1}B_0$		1.8	2.8	3.3	5.5	5.0	7.5	9.5
	0.95	1.8	2.9	3.2	5.8	5.1	7.6	9.5
catch = 2/3MSY	0.75	1.4	2.3	2.5	4.4	3.9	5.7	6.9
	0.50	0.87	1.5	1.7	2.9	2.5	3.7	4.3
B. CAY rules								
$F_{CAY} = M$		0.05	0.10	0.10	0.20	0.20	0.21	0.28
$F_{CAY} = F_{0.1}$		0.073	0.11	0.13	0.22	0.20	0.30	0.38
	0.95	0.20	0.21	0.27	0.86	0.46	0.78	1.00
$F_{CAY} = F_{MSY}$	0.75	0.082	0.12	0.14	0.29	0.21	0.34	0.40
	0.50	0.038	0.062	0.071	0.13	0.10	0.16	0.18

maturity. For example, if recruitment to the fishery occurs n years before maturity $(A_f = A_m - n)$ then the effect on the spawning biomass of a given level of fishing will increase as n increases. Conversely, if $A_f = A_m + n$ for sufficiently large n then no amount of fishing pressure would drive S below 20% S_0, because of the unfished reservoir of mature fish of ages A_m to $A_m + n - 1$. There are two species in Table 1 for which $A_f \neq A_m$ (s_2, s_3). To investigate the effect of this inequality, analyses for these species were repeated after changing A_f to be equal to A_m.

The safety of each of the MCY and CAY rules was estimated, by simulation, for all combinations of the seven "species" (the five original species plus the two with modified A_f), three values of h, and three values of σ_R. This makes a total of 63 species-parameter combinations. Harvesting levels for each rule are given in Table 2.

Additional simulations were carried out to investigate the effect of uncertainty in natural mortality, M. (This parameter is generally not well estimated, and errors in M will obviously have a marked effect on the safety of a harvesting rule.) For each simulation run a random value of M was generated from a distribution for which the median was equal to the assumed M value (from Table 1), and 95% of the values lay between half and double the assumed value. (Random M = assumed $M \times \exp(X)$, where X is a normal variate with mean 0 and standard deviation $0.5 \log_e 2$). The assumed M was used in calculating the harvesting level; the random M was used in modelling the population. Simulations were carried out with "uncertain M" for the 15 combinations of species and h. For all these simulations σ_R was fixed at its middle value (0.6).

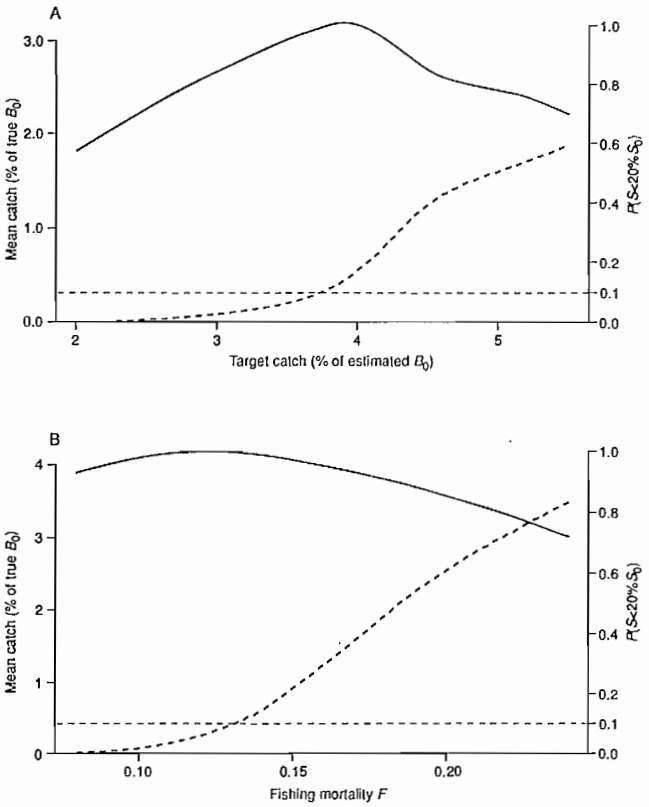


Fig. 2. Typical simulation results. Mean catch (solid line) and $P(S < 20\%S_0)$ (broken line) for species s_3 with $h = 0.5$ and $\sigma_R = 0.8$, harvested at a range of fishing intensities: A. with constant catch; B. with constant fishing mortality, F.

Results

Typical results for simulations of harvesting with constant catch are shown in Fig. 2A. There are two target catches of interest in this figure: that for which $P(S < 20\%S_0) = 0.1$ (the maximum safe target catch, = 3.7% B_0), and that which maximizes the mean catch (= 3.9% B_0). If we accept the above definition of "an acceptable level of risk" then the MCY is the lower of these two target catches (= 3.7% B_0). The maximum

Table 3. Maximum safe harvesting level expressed as a percentage of the harvesting level that maximizes mean catch: A. harvesting with constant catch; B. harvesting with constant fishing mortality. ("safe" means that the probability that the spawning biomass will fall below 20% of its mean virgin level is less than 10%).

Stock-recruit steepness, h	Recruitment variability, σ_R	Species						
		s_1	s_2	s_2'	s_3	s_3'	s_4	s_5
A. constant catch								
0.95	0.4	94	101	97	55	98	98	96
0.95	0.6	92	98	93	51	93	91	91
0.95	0.8	88	93	88	45	87	84	84
0.75	0.4	100	104	104	101	104	98	94
0.75	0.6	98	100	103	95	102	93	91
0.75	0.8	96	95	100	88	96	89	86
0.50	0.4	100	102	104	102	102	103	102
0.50	0.6	101	100	102	101	102	99	103
0.50	0.8	102	98	95	96	97	91	99
B. constant F								
0.95	0.4	57	83	78	49	61	55	45
0.95	0.6	54	77	71	43	55	48	40
0.95	0.8	50	70	64	39	50	42	35
0.75	0.4	110	115	114	106	104	100	93
0.75	0.6	104	108	106	92	94	89	82
0.75	0.8	96	99	95	82	84	78	72
0.50	0.4	140	135	136	134	131	129	127
0.50	0.6	138	128	126	122	118	116	115
0.50	0.8	129	117	115	109	106	102	100

safe target catch tended to be just less than that which maximizes the mean catch for $h = 0.95$, about equal for $h = 0.75$, and slightly greater for $h = 0.5$ (Table 3A).

Similarly, in Fig. 2B there are two target F's of interest: that for which $P(S < 20\%S_0) = 0.1$ (the maximum safe F, = 0.13), and that which maximizes the mean catch (= 0.12). F_{CAY} is the lower of these two target F's (= 0.12). The maximum safe target F was always less than that which maximizes the mean catch for $h = 0.95$, about equal for $h = 0.75$, and always greater for $h = 0.5$ (Table 3B).

For almost all species-parameter combinations the mean catch associated with the MCY (or CAY) was $\geq 90\%$ of the maximum mean catch achievable when fishing with constant catch (or constant F) (Table 4).

The acceptability of the rules for MCY and CAY depended most strongly on the steepness parameter, h. Thus the rules based on M (MCY = $0.25MB_0$, $F_{CAY} = M$) and $F_{0.1}$ (MCY = $0.25F_{0.1}B_0$ and $F_{CAY} = F_{0.1}$) were almost always acceptable for $h = 0.95$ and almost never acceptable for $h = 0.5$ (Figs. 3, 4). For the MSY-based rules (MCY = 2/3MSY, $F_{CAY} = F_{MSY}$) the pattern was reversed, with acceptability decreasing as h increased.

The second most important characteristic for determining acceptability was species. For each value of h, $P(S < 20\%S_0)$ tended to vary more between species for a given σ_R than it did between values of σ_R for a given species. (Figs. 3, 4).

Of the species life-history parameters in Table 1 the most important in determining the acceptability of a rule was natural mortality, M. For each of the 54 combinations of a rule, h, and σ_R, the five species were ranked in increasing order of $P(S < 20\%S_0)$. The average of these rankings over the 54 rule-parameter combinations was (in increasing order): $s_1 = 2.28$, $s_2 = 2.33$, $s_3 = 2.73$, $s_4 = 3.28$, and $s_5 = 4.23$. The same ordering is achieved by ranking the species by increasing M (Table 1). Thus a rule is more likely to be acceptable for species with low M.

Recruitment variability, σ_R, also affects acceptability, with acceptability decreasing as σ_R increases (Figs. 3, 4). However, this effect is much less than that associated with steepness or species.

The two M-based rules performed approximately equally well: both MCY = $0.25MB_0$ and $F_{CAY} = M$ were acceptable for 67% (30/45) of the species-parameter combinations. The $F_{0.1}$-based rules were also about equally acceptable (MCY = $0.25F_{0.1}B_0$ was acceptable for 42%, and $F_{CAY} = F_{0.1}$ for 51% of combinations). However, the MSY-based rules differed markedly: MCY = 2/3MSY was acceptable for 96% of species-parameter combinations, whereas $F_{CAY} = F_{MSY}$ was acceptable for only 40%.

The effect of a mismatch between the maturity and recruitment ogives is complex. For example, the effect, for species s_2, of increasing A_f to equal A_m was to increase the maximum safe target catch and F, and also to increase the target harvesting levels that maximize mean catch. Decreasing A_f for species s_3 had the opposite effect. As a result, the change made both of the M-based rules more acceptable

Table 4. Mean catch as a percentage of the maximum mean catch obtainable. A, at MCY; B, at CAY.

Stock-recruit steepness, h	Recruitment variability, σ_R	Species						
		s_1	s_2	s_2'	s_3	s_3'	s_4	s_5
A. MCY								
0.95	0.4	99	100	99	80	100	99	99
0.95	0.6	98	100	98	77	98	97	98
0.95	0.8	99	99	97	72	96	95	93
0.75	0.4	100	100	100	100	100	100	98
0.75	0.6	100	100	100	99	100	99	98
0.75	0.8	99	99	100	99	99	97	95
0.50	0.4	100	100	100	100	100	100	100
0.50	0.6	100	100	100	100	100	100	100
0.50	0.8	100	100	99	99	99	97	100
B. CAY								
0.95	0.4	96	99	99	97	96	95	92
0.95	0.6	95	99	98	95	95	93	89
0.95	0.8	94	97	97	92	93	90	86
0.75	0.4	100	100	100	100	100	100	100
0.75	0.6	100	100	100	100	100	100	99
0.75	0.8	100	100	100	99	99	98	97
0.50	0.4	100	100	100	100	100	100	100
0.50	0.6	100	100	100	100	100	100	100
0.50	0.8	100	100	100	100	100	100	100

(i.e., $P(S < 20\%S_0)$ decreased) for s_2, and less acceptable for s_3. However, the effect with the other rules was not so easily predictable. This is because the change in A_f caused $F_{0.1}$, F_{MSY}, and MSY to increase for s_2, and decrease for s_3 (Table 2). Thus, for example, the increase in A_f made s_2 capable of sustaining higher fishing pressure but it also made the $F_{0.1}$ and MSY rules require more fishing pressure. The rule MCY = $0.25F_{0.1}B_0$ became less acceptable for s_2, and both $F_{0.1}$-based rules became less acceptable for s_3 . Otherwise, changes in the acceptability of the rules for these two species tended to be small and dependent on the values of h and σ_R.

The effect of uncertainty in M was to decrease the acceptability of the harvesting rules. The probability of S falling below 20% S_0 increased when, for M known precisely, it had been less than 0.5, and decreased when it had been greater than 0.5 (Fig. 5). Of the 90 species-rule-steepness combinations considered, the risk was acceptable ($P(S < 20\%S_0) < 0.1$) for only 35 (39%) when M was uncertain. This compares to 57 acceptable combinations (63%), when M was assumed known exactly.

Discussion

Choice of Threshold

The first use of the 20% S_0 threshold that I am aware of was by Beddington and Cooke (1983). They comment (p. 9) "... this is not a conservative figure, but it represents a lower limit where recruitment declines might be expected to be observable". The following two recent papers support the use of this value.

Thompson (1993) showed that, under certain very general assumptions, maintaining a stock at above 20% of its virgin level could be expected to protect it against collapse. He cites other authors who use the same threshold value or a range that includes 20%. Also, he found that, for a range of stocks (from the Bering Sea, Aleutian Islands and Gulf of Alaska), the equilibrium biomass B_{MSY} associated with F_{MSY} was above 20% of virgin biomass (though he did not consider the effects of stochastic recruitment).

Clark (1991) found that "... with a range of life history parameters typical of demersal fish and a range of realistic spawner-recruit relationships ... yield will be at least 75% of the maximum sustainable yield so long as the spawning biomass is maintained in the range of about 20–60% of the unfished level". He considered both Ricker and Beverton and Holt stock-recruit relationships; for the latter, the curves he considered have steepness between 0.5 and 0.8 (his parameter A is related to steepness, h, by the equation $A = (5h-1)/4h$. Again he did not consider the effects of stochastic recruitment.

Thus both authors support the conclusion of the current study that the 20% S_0 threshold is not so high as to restrict catches much below maximum sustainable levels. In addition, the former offers theoretical evidence that this threshold is high enough to prevent collapse. I conclude that my definition of an acceptable level of risk (i.e., $P(S < 20\%S_0) < 0.1$) is reasonable. Given this, we can address the question of how safe (or acceptable) are the six harvesting rules listed in Table 2.

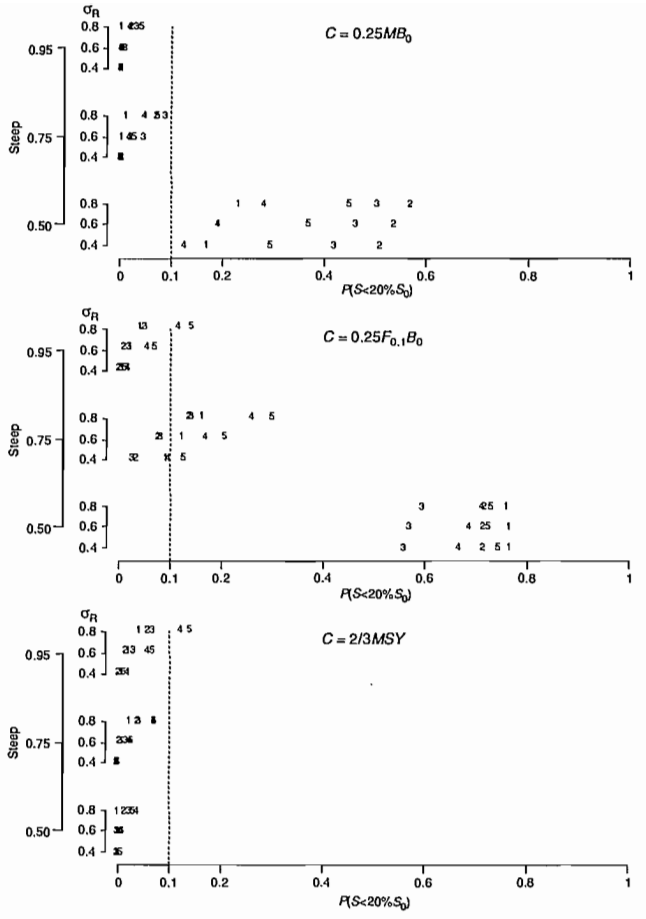


Fig. 3. The safety (or acceptability) of the three MCY rules of Table 2, as measured by the probability that the spawning biomass, S, falls below 20% of its mean virgin level. Levels of this probability below 0.1 are considered acceptable. Results are presented for three levels of stock-recruit steepness, h, three levels of recruitment variability σ_r, and five species. Plotting symbol "1" represents species s_1 , etc. (full species names and life-history parameters are given in Table 1).

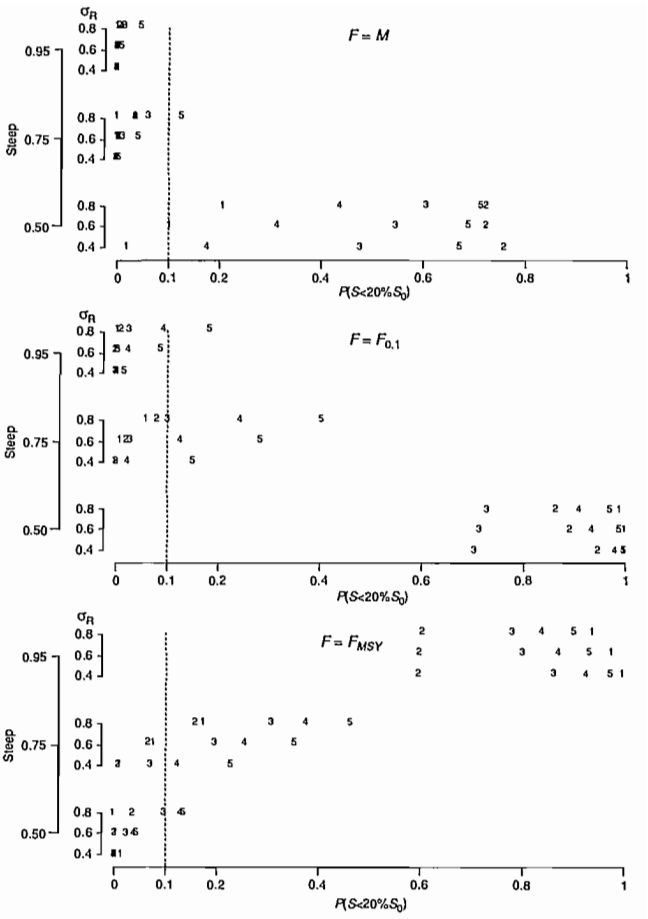


Fig. 4. The safety (or acceptability) of the three CAY rules of Table 2, as measured by the probability that the spawning biomass, S, falls below 20% of its mean virgin level. Levels of this probability below 0.1 are considered acceptable. Results are presented for three levels of stock-recruit steepness, h, three levels of recruitment variability σ_r, and five species. Plotting symbol "1" represents species s_1 , etc. (full species names and life-history parameters are given in Table 1).

Acceptability of Rules

The answer depends strongly on the steepness, h, of the stock-recruit relationship, less strongly on natural mortality, M, and least strongly on recruitment variability, σ_R (Figs. 3, 4). This confirms the finding of Getz et al. (1987) that "the most important source of uncertainty in estimating long-term productivity of a fishery is the degree of density-dependence in the stock-recruitment relationship and not the environmental factors that influence annual recruitment rates". For M and σ_R, acceptability decreases as the parameter increases. This is also true for steepness with the M-based and $F_{0.1}$-based rules, but the opposite holds for the MSY-based rules. Uncertainty in M decreases acceptability (Fig. 5). The effect of a mismatch between the ages of recruitment and maturity depends on the rule.

If my proposed definition of acceptable risk is adopted then the results given here show that it is preferable to calculate the reference points MCY and CAY using a simulation procedure rather than the rules considered above. A difficulty in New Zealand (and in most other places) is deciding what value of the stock-recruit steepness parameter to use for each species.

Choice of Steepness

No data exist to estimate this parameter for New Zealand species. Though there are many published stock-recruit data sets for species in other countries, these do not appear to be useful in providing likely steepness values for New Zealand species. In most cases B_0 is not known and time series are often too short and too noisy, for any meaningful curve-fitting. Also, the estimates of stock size and recruitment that make up the data sets (often derived from VPA) tend to be biased, correlated, and autocorrelated.

It is important to note that the present population model includes only one compensatory mechanism: a convex stock-recruit relationship. Other mechanisms by which a population

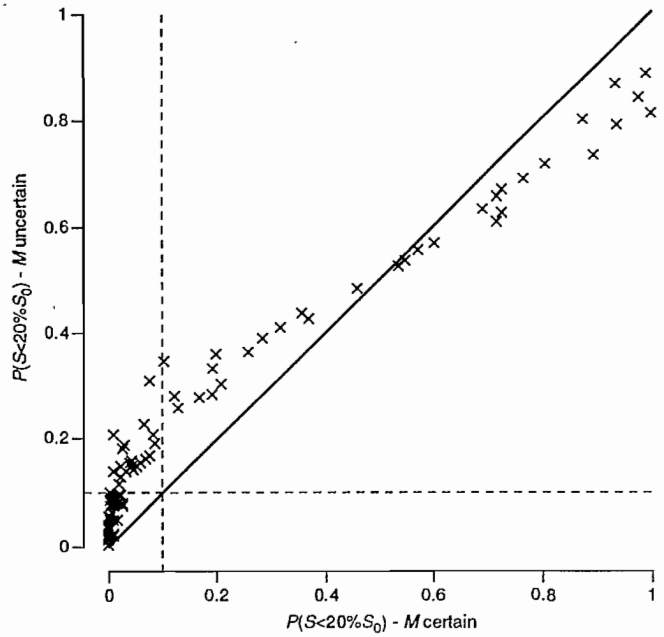


Fig. 5. The effect of uncertainty in natural mortality, M, on the risk associated with the harvesting rules. Each point in the plot represents a species-rule-parameter combination (five species, six rules, three values of stock-recruit steepness). The probability of the spawning biomass, S, falling below 20% of its mean virgin level, S_0, is plotted on both axes: x-axis, when M is assumed known exactly, y-axis, when M is uncertain. The diagonal line is $y = x$; the dotted lines mark the threshold between acceptable and unacceptable risk.

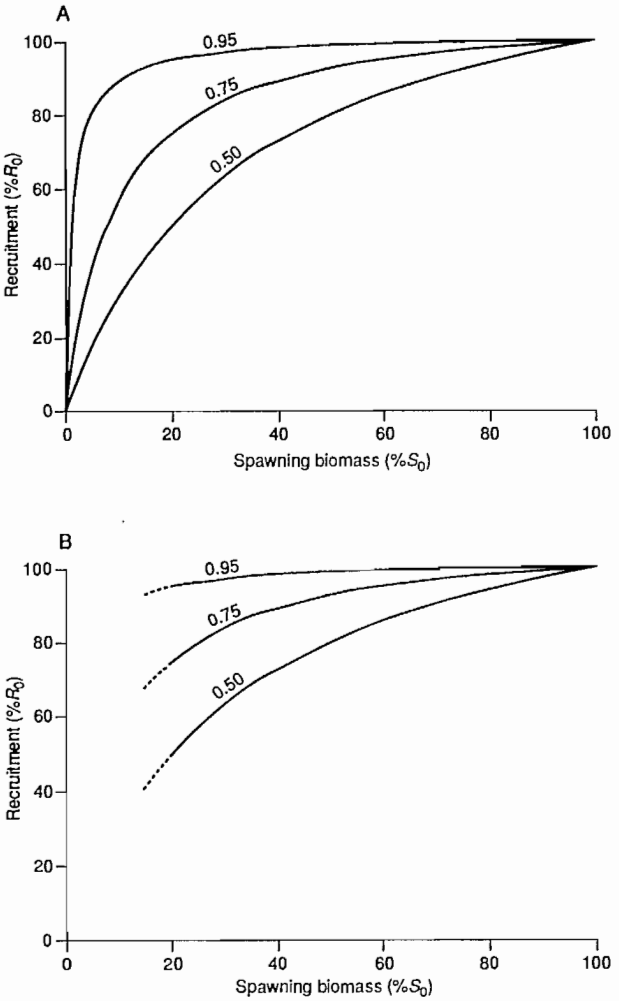


Fig. 6. The Beverton and Holt stock-recruit relationship for three values of steepness (0.5, 0.75, 0.95): A, as described, literally, in the model; B, as effectively used in calculating MCY and CAY.

might compensate for the effect of fishing include: increased growth rate, earlier maturity, increased fecundity, and reduced natural mortality during some life stages. Thus, the steepness parameter should be seen as a simple proxy for a range of compensatory mechanisms. In this sense it may be thought of as representing the resilience of a population under fishing. High steepness implies high resilience.

Given the lack of useful data, a reasonable approach to the problem of choosing a steepness value would be to decide on a plausible range of values and pick the mid-point as a default.

Since $F = M$ is widely considered to be a conservative harvesting policy, it is tempting to use the results of these simulations to conclude that, for most species, steepness is unlikely to be as low as 0.5 (elasmobranchs, with their low fecundity, are possible exceptions). For $h = 0.5$ even $F = 0.8M$, the more conservative rule proposed by Thompson (1993), was unsafe for 52% (11/21) of species-parameter combinations. Clark (1991) used a similar argument to suggest that 0.5 is a lower bound for steepness.

The highest value used here — 0.95 — must be considered an upper limit. (It is almost tantamount to assuming no relationship between stock and recruitment).

Thus a value of 0.75 — about midway between 0.5 and 0.95 — would appear to be a reasonable default steepness. It is also close to the value of 0.69 which, until recently, was a commonly accepted default value on the west coast of the U.S.A. (W. G. Clark, International Pacific Halibut Commission, Box 95009, Seattle, WA 98145-2009, pers. comm.; a steepness of 0.69 means that recruitment is 90% of its virgin level when the stock is at 50% of its virgin level).

It may be that 0.95 is too high as an upper bound. Clark (1991) considered the plausible range to be 0.5 to 0.8 (so the mid-point of his range is 0.65). He argued that, with the life history parameters of his "typical demersal fish", even a steepness of 0.89 (his $A = 0.969$) is too extreme because it implies that a population could sustain fishing at $F = 1$ with "hardly any ill effect". By this he appears to mean that the equilibrium recruitment at this level of fishing (in a deterministic model) is not much lower than the virgin recruitment. However, for all the species-parameter combinations considered here this fishing mortality drives the stock biomass to extremely low levels (and I suspect this would be true for his "typical demersal fish"). At such levels one might expect a stock-recruit relationship to break down so the inference of "hardly any ill effect" may not be tenable.

What is clear is that a value of 0.95, which has been used in a number of New Zealand stock assessments, is at the least conservative extreme of possible values.

Other Comments

The greatest area of uncertainty in stock-recruit relationships is what happens at low stock levels. Thus, predictions based on the above simulations become increasingly uncertain as harvest levels rise (and biomass declines). For this reason the right-most parts of both graphs in Fig. 2 must not be taken too literally. If, as Thompson (1993) suggests, the stock-recruit relationship were to become depensatory (concave upwards) at low stock levels, the right-hand parts of these graphs would be quite different. However, the MCY and CAY results presented here will be approximately correct as long as the Beverton and Holt relationship is reasonably accurate for stock sizes above about 20% S_0. Thus, we might think of the assumed stock-recruit relationships as being like Fig. 6B rather than Fig. 6A.

An obvious extension of the above work would be to see how robust the conclusions are to a change in the stock-recruit relationship. In particular it is of interest to see the effect of using a domed relationship, like that of Ricker (1975), for example. Some preliminary simulations with this relationship have revealed a technical problem: under constant catch (or constant F) harvesting the biomass tends to exhibit a cyclic behaviour. Thus, $P(S < 20\%S_0)$ will depend on the phase and frequency of the cycle and the length of the simulation period. More work is required to resolve this difficulty.

The simulations presented here are supposed to represent an idealized situation, rather than describe what might happen in a real fishery. In a real fishery where a constant F rule is applied, the annual biomass estimates will certainly be serially correlated and probably biased. For most fisheries there will be a limit to the proportion of the recruited biomass that can be caught in any one year (no such limit was assumed here). Also, harvesting rules are never followed exactly for long. Fishing industry viability, improved knowledge of the fish biology, and changes in the perception of the stock status are some of the factors that often cause rules to be modified. (e.g., the change in eastern Canada from $F_{0.1}$ to the "50% rule" (Rivard and Maguire 1993)). This study is aimed at helping managers (and their advisers) to make such modifications by showing what might happen in the hypothetical case where a harvesting rule was applied rigidly for many years.

It is important to note that the values of the probability $P(S < 20\%S_0)$ in Figs. 3 and 4 were obtained by integrating over both the A_l years in each simulation run, and the 500 simulation runs. Thus, to take an extreme example, an estimated probability of 0.1 could arise from the case where $S < 20\%S_0$ exactly 10% of the time for all 500 runs, or from the case where S was always $< 20\%S_0$ for 50 runs and always $> 20\%S_0$ for the other 450 runs. Simulations with constant F and M known exactly tend to produce results closer to the former extreme; those with constant catch or uncertain M gave results closer to the latter.

The simulation procedure and definition of risk used here are superficially similar to those of Beddington and Cooke (1983). However, there are several important differences. They used only a steepness of 1; they simulated only the 20-year period starting from a virgin population; they measured the proportion of runs (rather than the proportion of years) in which S fell below 20% S_0; and they included no biomass estimation error.

Conclusions

I conclude that the definition of "an acceptable level of risk" proposed here is a reasonable starting point. The above simulations show that it is not too conservative; reasons to believe that it is not too liberal are the theoretical work of Thompson (1993) and the growing use of the 20% S_0 threshold. Use of this definition allows the reference points MCY and CAY to be calculated directly, rather than via the ad hoc rules given above. It also highlights the importance in these calculations of the steepness (i.e., degree of compensation) of the stock-recruit relationship.

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Appendix: Gradual Recruitment and Maturity

There are (at least) two ways of modelling gradual recruitment (or maturity) in a fish population. A simple way is to assume, for a given population, that the proportion of fish of a given age that are recruited (or mature) is the same at all levels of biomass and fishing pressure. An alternative approach is to assume that the age at which any particular fish will recruit to the fishery (or become mature) is inherent and unaffected by fishing. With this approach (which is used in the present model) the proportion of recruited (or mature) fish in a given age class at a given time will depend on the history of exploitation. The following equations describe how, in this model, gradual maturity is described. Precisely analogous equations describe gradual recruitment.

I assume that, in an unfished population, the proportion, p_i, of fish of age i that are mature is given by

$$p_i = \begin{cases} 0 & \text{for } i < (A_m - S_m) \\ \left(1 + \exp\left[\dfrac{(\log_e 19)(A_m - i)}{S_m}\right]\right)^{-1} & \text{for } (A_m - S_m) \leq i \leq (A_m + S_m) \\ 1 & \text{for } i > (A_m + S_m) \end{cases}$$

where A_m is the age at 50% maturity and S_m is a parameter describing the extent of gradual maturity. The meaning of S_m may be understood by noting that the percentage of mature fish in a cohort increases from 5% at age $A_m - S_m$ to 95% at age $A_m + S_m$ (in the unfished population). When $S_m = 0$ maturity is assumed to be knife-edge, with 100% maturity at age A_m.

From the above equation it may be calculated that, in the unfished population, the proportion, q_i, of fish that are immature at age i that will mature in the following year is given by

$$q_i = \begin{cases} 0 & \text{for } i < (A_m - S_m - 1) \\ \dfrac{(p_{i+1} - p_i)}{(1 - p_i)} & \text{for } (A_m - S_m - 1) \leq i \leq (A_m + S_m) \end{cases}$$

In modelling a population I assume that the proportions, q_i, are unchanged by harvesting (which implies that the p_i are not).

A Comparison of Event Tree Risk Analysis to Ricker Spawner-Recruit Simulation: An Example with Atlantic Menhaden

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Vaughan, D. S. 1993. A comparison of event tree risk analysis to Ricker spawner-recruit simulation: an example with Atlantic menhaden. p 231–241. *In* S. J. Smith, J. J. Hunt and D. Rivard [ed.] Risk evaluation and biological reference points for fisheries management. Can. Spec. Publ. Fish. Aquat. Sci. 120.

Fishery stock assessments often attempt to provide future projections of population abundance and landings on which management decisions can be based. However, uncertainties in estimating the relationship between spawners and recruits, that drive such projections, are often considerable. The purpose of this study is to compare two relationships for characterizing these uncertainties using simulated projections of future population abundance with an age structure model for Atlantic menhaden data. One relationship is based on a mathematical spawner-recruit relationship (Ricker model). An alternative relationship reduces spawners and recruits to discrete categories, and estimates conditional probabilities to determine subsequent recruitment from spawning biomass (Event Tree risk analysis). Six biological "trigger" variables and activation levels for the Atlantic menhaden population are under consideration by the Atlantic States Marine Fisheries Commission. The first five variables serve as biological reference points for the simulations. The last variable, maximum spawning potential (MSP), defines constant levels of fishing mortality for each simulation. Four of the five biological reference points suggest greater risk associated with the Ricker relationship than that associated with the Event Tree relationship at the 1980's level of fishing mortality (MSP of 4.5%). A large decrease in risk associated with spawning stock biomass from the Ricker relationship was obtained with an MSP of 10%, with little decrease in risk available from further decreasing fishing mortality to obtain an MSP of 20% or higher. Gradual declines in risk associated with spawning stock biomass from the Event Tree relationship was obtained as fishing mortality was decreased to correspond to an MSP level of 10%, 20% and 30%.

Les évaluations de stock de pêche tentent souvent d'offrir des prévisions de l'abondance de population et des quantités débarquées en vue d'étayer les décisions de gestion. Toutefois, les incertitudes d'estimation de la relation entre les géniteurs et les recrutés, sur laquelle reposent ces prévisions, sont souvent considérables. L'objectif de cette étude est de comparer deux relations afin de caractériser ces incertitudes à l'aide de prévisions simulées de l'abondance de la population future selon un modèle de structure d'âge avec les données décrivant l'alose tyran. Une de ces relations s'inspire de la relation mathématique entre les géniteurs et les recrutés (modèle de Ricker). Une autre relation réduit les géniteurs et les recrutés en catégories discrètes et évalue les probabilités conditionnelles de détermination du recrutement futur selon la biomasse génitrice (analyse du risque par arbre d'événements). Six variables de «déclencheurs» biologiques et niveaux d'activation pour la population d'alose tyran sont actuellement en cours d'étude par l'Atlantic States Marine Fisheries Commission. Les cinq premières variables servent de points de référence biologiques des simulations. La dernière variable, le potentiel géniteur maximum (PGM), définit les niveaux constants de mortalité par pêche pour chaque simulation. Quatre des cinq points de référence biologiques laissent croire en un risque plus élevé avec la relation de Ricker qu'avec la relation de l'arbre d'événements, selon le niveau de mortalité par pêche des années 1980 (PGM de 4,5 %). Une importante diminution du risque a été notée, dans le cas de la biomasse génitrice de la relation de Ricker, avec un PGM de 10 % et avec une très faible diminution du risque associée à la réduction additionnelle de mortalité par pêche pour obtenir un PGM de 20 % ou plus. Un déclin graduel du risque associé à la biomasse génitrice à partir de la relation de l'arbre d'événements a été obtenu lorsque la mortalité par pêche était réduite pour correspondre aux niveaux de PGM de 10 %, de 20 % et de 30 %.

To predict future population levels of fish stocks, it is necessary to understand the relationship between spawning stock and subsequent recruits to the fishery. Unfortunately, it is difficult to obtain a reasonable mathematical relationship between spawners and recruits for most fish stocks. Even if "statistically significant", the fit of spawner-recruit data to a mathematical relationship has large error remaining unexplained by the model.

My purpose in this study is to compare two relationships for relating spawning stock biomass and subsequent recruits to age 0 (about six months of age) using Monte Carlo population simulations. The first relationship is based on fitting the historic spawner and recruit data to the Ricker model (Ricker 1975) using nonlinear regression techniques (SAS Institute Inc. 1987), and estimating the parameters for the corresponding normal distribution of the residuals.

The second relationship is based on Event Tree risk analysis (Brown and Patil 1986; Linder et al. 1987), by which spawners and recruits are each divided independently into three categories (low, medium, high) and then the conditional probability is estimated for each recruit category conditioned on the spawner category. Actual values for recruits are then randomly selected from distributions for the three recruit categories.

Parameters of the Ricker spawner-recruit model and conditional probabilities for the Event Tree relationship are estimated from data available from the Atlantic menhaden stock (Vaughan and Smith 1988; Vaughan 1990). Biological reference points based on five of six biological "triggers" adopted by the Atlantic States Marine Fisheries Commission's (ASMFC) Atlantic Menhaden Board (AMB) are used to compare replicate 12-year simulations using these two relationships based on different levels of instantaneous fishing mortality rates (related to the sixth trigger).

In this study, the Atlantic menhaden and six triggers are briefly described. Next, the parameter estimation for the two spawner-recruit relationships and population projections are described. Projections are compared for two spawner-recruit relationships and different levels of fishing mortality rates (as related to the sixth trigger). Finally, the significance of the results are discussed with respect to limitations in conducting the simulations and what management conclusions can be drawn.

Atlantic Menhadden Fishery

Atlantic menhaden *(Brevoortia tyrannus)* is a euryhaline species found in coastal and inland tidal waters from Nova Scotia, Canada, to West Palm Beach, Florida (Reintjes 1969). Adult menhaden are filter feeders, feeding primarily on phytoplankton, and in turn supporting predatory food fishes. A commercial fishery for Atlantic menhaden has existed since colonial times, but purse seines, now the principal gear, were introduced by 1845 (Frye 1978). Large carrier vessels are equipped with a pair of purse boats for setting the seines around the schools of fish. In 1990, five active reduction facilities on the Atlantic coast, including shore based plants in North Carolina (1 plant), Virginia (2), New Brunswick, Canada (1), and an internal waters processing agreement in Maine with the Soviet factory ship Riga, processed the fish into meal, oil, and solubles (Vaughan 1990).

Management of Atlantic menhaden in the United States is by the individual U.S. Atlantic coastal states as coordinated through the ASMFC. The ASMFC's AMB recently approved (August 5, 1992) six biological reference or "trigger" variables devised by the ASMFC's Atlantic Menhaden Advisory Committee (AMAC). When these estimated variables are compared to pre-selected values, and at least one variable is found to lie in a "danger zone", then a meeting of AMAC will be held to determine whether corrective action is needed. In the committee's discussions all six trigger variables and any relevant ancillary information will be considered.

The six biological "trigger" variables are: (1) annual landings in weight below 250,000 t, (2) proportion of age 0 menhaden in the landings by number above 25%, (3) proportion of adult menhaden (age 3 and older) in the landings by number above 25%, (4) recruits to age 1 below 2 billion, (5) spawning stock biomass below 17,000 t, and (6) percent of maximum spawning potential below 3%. Annual data for the first three trigger variables are available soon after the end of the fishing season. The other trigger variables are obtained from output produced in a virtual population analysis (VPA).

The first three triggers are subject to the potential for "false firing", because values in the "danger zone" do not necessarily imply that something is wrong with the Atlantic menhaden stock. Ancillary information is important to judge whether the "firing" of these triggers requires some action by the committee. The final three triggers are also subject to "false firing", but for a different reason than for the first three triggers. Recent estimates from the virtual population analysis are subject to large uncertainty. However, estimates more than 2 or 3 years old are generally more accurate. If the estimates are accurate and precise, then "firing" of these trigger variables reflect a real problem in the Atlantic menhaden stock.

Generally, the interquartile range of the historical data from 1965 through 1990 was used to select the particular trigger value for each variable (Table 1). The value for the third trigger is based on historical data from 1955 through 1990 to include the stock collapse during the 1960's. The first, fourth, fifth, and sixth variables are triggered if the respective values fall below the 25th percentile (or similar value). The second variable is triggered if the respective values fall above the 75th percentile.

Atlantic menhaden landings were selected because they could be an indicator of a change in stock abundance. Estimates are available from 1940 through 1990 (Table 1). They peaked in the late 1950's, collapsed during the 1960's, and improved somewhat during the 1970's. An awareness of whether economic conditions have drastically reduced the fishing activity, as in 1986 when a major plant closed, would be important ancillary information.

The proportion of age 0 menhaden in the landings by numbers was selected because of two concerns. First, very high harvest of the youngest fish may reduce potential yields based on a yield per recruit or "growth overfishing" argument (Vaughan and Smith 1988, 1991). Second, although the catch of age 0 menhaden is highly weather dependent, a large harvest of these fish in years of poor recruitment may greatly reduce subsequent spawning stock biomass. These data are available from 1955 through 1990 (Table 1).

Proportion of adult (age 3+) menhaden in the landings by number was also selected because of two concerns. First, it may result in a short-term reduction of the adult spawning stock. Second, large landings of adults relative to subadults may indicate the potential for recruitment failure as in 1961 and 1962, as the huge 1958 year class attained the age of 3 and 4, respectively, and recruitment for the following years were poor. Hence, a large showing of adult menhaden by numbers

Table 1. Annual estimated values of six Atlantic menhaden triggers, 1940–1990.

Year	Landings[a]	P0[b]	P3+[c]	Recruits[d]	SSB[e]	MSP[f]
1940	179.0	-	-	-	-	-
1941	283.1	-	-	-	-	-
1942	167.4	-	-	-	-	-
1943	215.0	-	-	-	-	-
1944	243.5	-	-	-	-	-
1945	285.6	-	-	-	-	-
1946	351.8	-	-	-	-	-
1947	376.4	-	-	-	-	-
1948	341.3	-	-	-	-	-
1949	363.4	-	-	-	-	-
1950	311.2	-	-	-	-	-
1951	351.3	-	-	-	-	-
1952	423.6	-	-	-	-	-
1953	589.2	-	-	-	-	-
1954	617.9	-	-	-	-	-
1955	644.5	24.4	20.1	3.1	325.4	13.8
1956	715.4	1.0	15.5	5.7	257.3	6.6
1957	605.6	8.5	7.1	7.3	132.6	6.7
1958	512.4	3.9	4.4	3.3	88.3	16.1
1959	662.2	0.2	8.4	15.1	172.9	8.6
1960	532.2	2.6	7.7	2.2	122.7	24.1
1961	578.6	0.0	48.6	3.0	358.6	13.3
1962	541.6	2.5	33.3	2.2	199.0	4.9
1963	348.4	5.5	13.3	2.2	64.9	3.1
1964	270.4	17.5	6.8	1.7	30.7	2.4
1965	274.6	17.1	6.2	1.9	20.7	1.7
1966	220.5	26.1	2.7	1.4	9.0	3.3
1967	194.4	0.7	8.0	1.9	20.8	5.5
1968	235.9	13.4	6.7	1.2	16.7	2.1
1969	162.3	18.2	6.2	1.7	14.0	5.4
1970	259.4	1.5	2.6	2.6	16.1	6.6
1971	250.3	7.5	11.2	1.3	27.9	6.6
1972	365.9	2.9	11.3	3.4	47.7	2.0
1973	346.9	3.0	2.5	2.7	12.5	1.3
1974	292.2	15.9	2.6	3.0	12.0	1.5
1975	250.2	13.8	2.6	3.7	13.6	1.9
1976	340.5	8.4	1.7	6.8	15.5	2.8
1977	341.2	13.2	2.8	5.1	25.5	4.3
1978	344.1	14.8	9.5	4.7	44.3	3.7
1979	375.7	38.6	3.9	4.2	40.2	6.4
1980	401.5	2.6	9.2	6.7	57.7	4.6
1981	381.3	29.8	7.2	4.7	42.2	5.0
1982	382.5	3.6	12.7	6.4	48.5	3.1
1983	418.6	24.5	4.2	2.5	35.6	3.8
1984	326.3	36.5	9.5	3.8	55.0	1.7
1985	306.7	21.1	2.9	5.0	19.2	2.6
1986	238.0	5.2	3.5	4.6	16.5	7.6
1987	326.9	1.9	7.1	3.5	36.7	7.6
1988	309.3	18.6	16.8	3.3	58.2	4.8
1989	322.0	5.7	5.5	6.7	35.4	6.3
1990	401.2	25.5	6.1	2.0	43.9	14.4
Median[g]	324.1	13.6	6.9	3.5	26.7	4.1
25%	250.3	3.6	3.7	2.0	16.1	2.1
75%	365.9	21.1	10.3	4.7	43.9	6.3
Trigger	<250.0	>25.0	>25.0	<2.0	<17.0	<3.0

[a] Landings in thousands of metric tons.
[b] Percent by numbers of age 0's in landings.
[c] Percent by numbers of adults (ages 3 and older) in landings.
[d] Estimated numbers of recruits to age 1 in billions.
[e] Estimated mature female biomass (spawning stock biomass or SSB) in thousands of metric tons.
[f] Estimated equilibrium maximum spawning potential based on egg production (for estimated F vs $F = 0$) in percent (includes F at age 0).
[g] Median, 25th, and 75th percentiles based on fishing years from 1965 through 1990, except for p_3+ which is based on fishing years 1955 through 1990.

in the landings may indicate subsequent poor recruitment, and there are no reliable pre-recruit indices of year class strength.

The remaining trigger variables, based on virtual population analysis (VPA), are indicative of basic stock status. Because there are no fishery independent indices of recruitment available for Atlantic menhaden for use in tuning a VPA (Pope and Shepherd 1985), a separable VPA approach (Pope and Shepherd 1982; Clay 1990) was used to bring the VPA estimates of age-specific population size and fishing mortality rates up to the current year of data. The most recent estimates of age-specific population size and fishing mortality rates, and hence recruits to age 1, spawning stock biomass, and percent maximum spawning potential, should be viewed with some skepticism. Without a credible abundance index with which to tune a VPA, the consensus of AMAC was that values of trigger variables based on VPA should only be considered up to 2 or 3 years prior to the most recent data used in the VPA.

Estimates of recruits to age 1 were selected because they indicate directly what will be available to the fishery one year later as age 2 menhaden (the dominant age in the landings). Estimates are available for fishing years 1955 through 1990 in the VPA (Table 1).

Estimates of female spawning stock biomass were selected to represent the availability of spawners in sufficient quantity to produce adequate recruitment. Estimates are also available from 1955 through 1990 (Table 1). The spawning stock biomass is comprised predominantly of age 3 menhaden (Vaughan 1990).

Estimates of percent maximum spawning potential (%MSP) are used widely by the U.S. Fishery Management Councils and Marine Fisheries Commissions to define recruitment overfishing (Vaughan 1990, 1992, Vaughan et al. 1991, 1992). Estimates are calculated in an equilibrium manner from the fishing mortality rates obtained in the VPA for each fishing year from 1955 through 1990 (Table 1) (Gabriel et al. 1989). The low levels of %MSP are of obvious concern, because values between 20% and 30% have been used in a number of management plans by the South Atlantic and Gulf of Mexico Fishery Management Councils. However, values below 10% have been the norm for Atlantic menhaden since 1962. During this time, the stock collapsed and later rebuilt, with recruitment during the 1980's of comparable level to that of the 1950's (Vaughan 1990).

Methods

Virtual population analysis (VPA) is conducted on Atlantic menhaden catch-at-age data from 1955 through 1990 [as described in Vaughan and Smith (1988) and Vaughan (1990) with the addition of 1989 and 1990 data]. The instantaneous natural mortality rate used in the VPA's and in the simulated projections is 0.45/yr. The mean of the age-specific estimates of instantaneous fishing mortality rates (F) for 1980's are used in the projection simulations (Table 2). Starting population age vector is that estimated for 1989 in the VPA. Growth

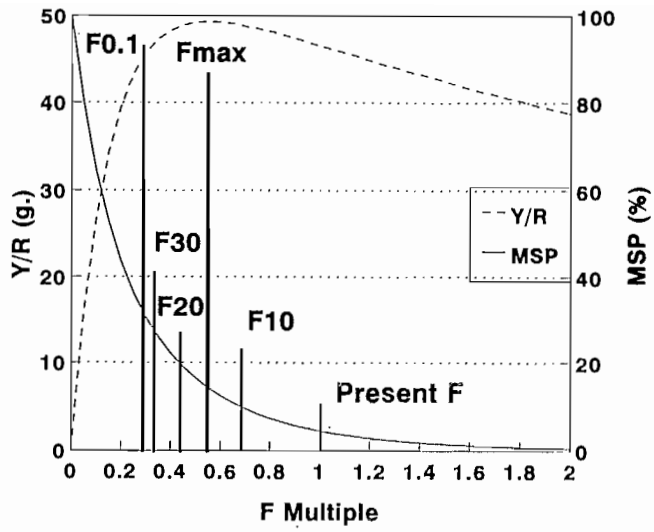


Fig. 1. Critical values of instantaneous fishing mortality rates (F) from yield per recruit ($F_{\max}$ and $F_{0.1}$) and from %maximum spawning potential (F_{10}, F_{20}, and F_{30}) for Atlantic menhaden. The estimate of present F is for an age at entry of 0.5 (6 months) and the mean F for the 1980's (Table 2).

in length for the 1980's is represented by a von Bertalanffy equation relating fork length in millimeters (L) with age in years (t):

$$L_t = 355.2\left(1 - e^{-0.28(t+0.613)}\right), \quad (1)$$

and weight in grams (W) is related to fork length through:

$$W_t = 0.00000556L^{3.2}. \quad (2)$$

Using these estimated F's and growth models, a Ricker-type yield-per-recruit analysis (Y/R) was done, and plotted for the current age at entry of age 0 (Fig. 1). With a sex ratio of 1:1 and female sexual maturity beginning with age 3, a corresponding analysis of maximum spawning potential (%MSP) was done, and also plotted for the current age at entry of 0. The F multiple on the x-axis corresponds to a scalar multiplied by the estimated vector of age- specific F's so that an F multiple of 1 corresponds to current or 1980's mean F (Table 2). Critical values of F are presented from both the Y/R ($F_{\max}$ and $F_{0.1}$) and %MSP (F_{10}, F_{20}, and F_{30}) analyses. The critical values from %MSP are those values of F which result in 10, 20, or 30% MSP. Because the Y/R analysis does not include consideration of maintaining sufficient spawning stock to prevent possible recruitment failure, it is of interest to compare these critical values from two different perspectives; $F_{0.1}$ and F_{30} are very similar in magnitude, while $F_{\max}$ lies between F_{10} and F_{20}.

Table 2. Mean, minimum, and maximum estimates of age-specific instantaneous fishing mortality rates, F, for Atlantic menhaden for the period 1980–89 from virtual population analysis.

Age	Instantaneous Fishing Mortality Rates		
	Mean	Minimum	Maximum
0	0.09	0.01	0.21
1	0.24	0.06	0.40
2	1.61	1.20	2.20
3	1.12	1.63	1.85
4	1.30	0.97	1.78
5	1.93	1.11	2.81
6	1.31	0.52	1.89
7	0.17	0.00	1.42
8	0.16	0.00	1.59

Simulations that follow are concerned with two approaches to capturing the underlying relationship between an index of spawning stock biomass (S) and an index of recruitment (R). The Ricker spawner-recruit model (Ricker 1975) uses the mathematical relationship:

$$R = \alpha S e^{-\beta S}, \tag{3}$$

which suggests that at moderate values of S, R will reach a maximum, but R will be small at low or high values of S. For the simulations that follow, spawners have been calculated in terms of spawning stock biomass by summing the age-specific products of population numbers and weights (females age 3 and older). Recruits are obtained directly from the VPA for age 0 (that is, at an age of about 6 months). Because the recruits in 1958 (19.0 billion) were almost twice that from any other year class (10.0 billion in 1979), the Ricker spawner-recruit model was estimated for the data from 1955 through 1989 (35 years) with and without the 1958 year class (Fig. 2). The two estimated models are:

$$R = 0.221 S e^{-0.0000101S} \text{ (with 1958)}, \tag{4}$$

and

$$R = 0.212 S e^{-0.0000109S} \text{ (without 1958)}, \tag{5}$$

where R is recruits to age 0 in millions and S is spawning stock biomass in metric tons. The use of PROC NLIN (SAS Institute Inc. 1987) with the Marquardt option assumes an additive normal error structure. The residuals based on the Ricker model excluding the 1958 year class tested as not significantly different from a normal distribution ($\mu = 0.5$ billion, $\sigma = 2.8$ billion). Hence, recruitment to age 0.5 is simulated by calculating the spawning stock biomass for a given year, using Eq. (5) to estimate recruitment to age 0.5 for that year, and adding error based on the normal distribution of the residuals estimated in Eq. (5).

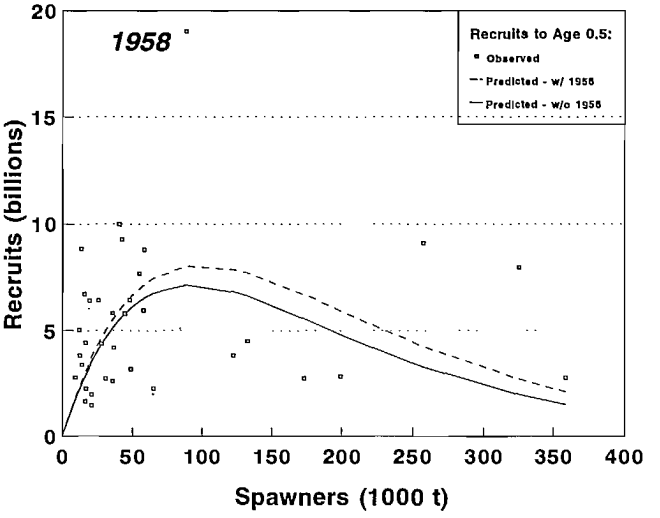


Fig. 2. Observed spawners (weight of females age 3 and older) and resultant recruits to age 0.5 and Ricker spawner-recruit curves for Atlantic menhaden with and without the 1958 year class, 1955–1989.

Table 3. Conditional probabilities from Atlantic menhaden spawners and recruits used in Event Tree relationship. Also given are statistical properties of recruitment categories used to generate specific values for use in simulations.

Spawning Stock Category[a]	Recruits to Age 0.5 Category[b]		
	Low	Medium	High
	Conditional Probabilities		
Low	0.222	0.556	0.222
Medium	0.235	0.530	0.235
High	0.333	0.333	0.334
	Properties of Recruit Categories (billions)		
Mean	2.3	4.5	9.7
Std. Dev.	0.5	1.2	3.6
Minimum	1.5	2.8	6.7
Maximum	2.8	6.4	19.0
Distribution[c]	Gamma	Uniform	Gamma
α	22.34	2.8	7.18
β	0.103	6.5	1.353

[a] The medium category for spawning stock is the interval greater than 16,700 t and less than 64,925 t based on the interquartile range. Low and high categories are below and above these values, respectively.
[b] The medium category for recruits to age 0.5 is the interval greater than 2.8 billion and less than 6.5 billion. Low and high categories are below and above these values, respectively.
[c] Statistical distributions (with parameters α and β) used to simulate actual value of recruitment to age 0.5 given recruit category from Event Tree relationship.

The Event Tree relationship creates discrete categories of spawners and recruits (Table 3). Categories for the spawning stock biomass and recruits to age 0 are based on the interquartile range of the historic data. The spawning stock biomass is

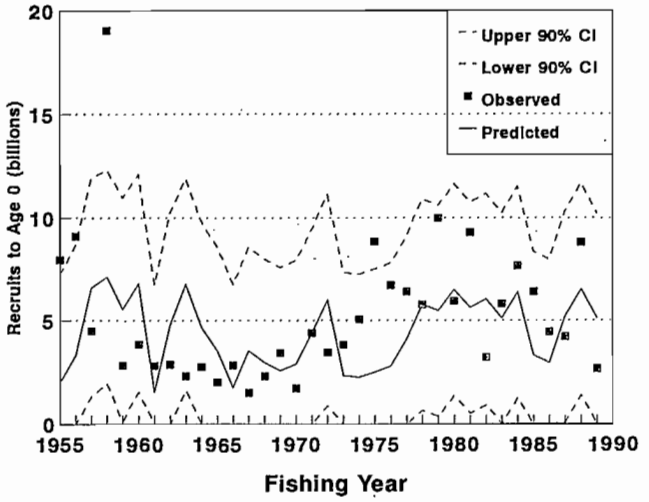


Fig. 3. Observed, predicted and 90% confidence interval about recruits to age 0.5 (6 month) Atlantic menhaden from Ricker spawner-recruit model (estimated without 1958 year class).

considered to be in the "low" category if it is less than or equal to 16,700 t (the 25th percentile), in the "medium" category if it falls between 16,700 t and 64,925 t (the 75th percentile), and in the "high" category if it is greater than or equal to 64,925 t. Similarly, recruits to age 0 are divided into three comparable categories based on its 25th (2.8 billion) and 75th (6.7 billion) percentiles.

The probability of obtaining a particular category of recruit given a particular category of spawning stock biomass forms the basis of the Event Tree relationship. These conditional probabilities are obtained from the historical frequency distribution of these categories for Atlantic menhaden (Table 3). Simulation of the Event Tree process follows two steps. First, given a spawning stock biomass category and the probability of recruitment falling in one of three recruit categories, a uniform distribution is used to assign a recruitment category for a particular year and replicate. Second, statistics calculated on each recruit category are used to generate a quantitative value for use in the population simulations (i.e., uniform distribution for the medium category and gamma distribution for the low and high categories as estimated from the historical data).

Two types of simulations are conducted in this study. The first simulates recruits with error from the historic spawning stock biomass estimated from 1955 through 1989 (100 replicates are made for each relationship). The Ricker model is then estimated (100 times) for each of these data sets generated by the two underlying relationships (Ricker versus Event Tree).

The second set of simulations consists of 12-year population projections (based on the VPA estimates for 1989 from 1990 through 2001). Again, 100 replicates are made for each relationship with the only error introduced where recruits to age 0.5 are estimated from spawning stock biomass. Five sets of parallel simulations are run, based on fishing mortality rates corresponding to five levels of maximum spawning potential. The F multiples and associated MSP are: 1.0 (4.5%), 0.68 (10%), 0.43 (20%), 0.31 (30%), and 0.0 (100%). In addition to the basic population size at age by year, values for the five trigger variables were saved. For comparison the cumulative probability (or risk) of exceeding a particular trigger value are calculated from the projections for each year of the projection. Twelve-year projections were done to represent one full generation (historical data of Atlantic menhaden suggest they lived to about 12 years).

Results

Because error is considerable between the observed and predicted recruits to age 0 based on the Ricker spawner-recruit model, the two relationships (Ricker and Event Tree) were used to recreate or simulate 100 replicates each of recruits from the historic estimates of spawners (Fig. 3). The Ricker model was fit to the 100 replicates of each relationship (200 fits). The observed variability in the Ricker model parameters (α, β in Eq. 3) from the two relationships are compared to the original Ricker model parameters with the 1958 year class excluded (Fig. 4). The Event Tree relationship shows much greater variability in the estimated Ricker parameters than do the estimates from the Ricker relationship. However, there is slightly more bias, based on the difference between the median estimates and the original point estimates, for the Ricker relationship compared to the Event Tree relationship.

Next, the five trigger variables that serve as biological reference points are compared based on simulations from the two relationships with F based on MSP equal to 4.5% in the year 2001. For landings, the trigger is activated when less than or equal to 250,000 t, so risk is defined as the cumulative probability of being less than or equal to a landings value (Fig. 5). Landings in 1990 is fixed by the starting age structure in 1989, with the trigger value of 250,000 t corresponding to 62.5% of the landings estimated for 1990. A value of 0 on the x-axis implies no landings, and a value of 100 implies landings equal to those of 1990. Moving from right to left, the risk curves are initially similar, then the risk associated with the Event Tree relationship is higher, then convergence, and finally the risk associated with the Ricker relationships is higher. At the trigger value of 62.5%, the risk associated with the Event Tree relationship is higher than that associated with the Ricker relationship by about 12%. The risks are almost identical for a decline of about 50% in landings from 1990.

For the percent of age 0 menhaden in the catch in numbers, the trigger is activated if a value greater than 25% is obtained. Risk is defined as the cumulative probability of exceeding a given percent of age 0 menhaden in the catch in numbers (Fig. 6). The risk curves based on both relationships are very similar, with generally slightly higher values from the Ricker relationship, including at the trigger value where the difference is about 5%.

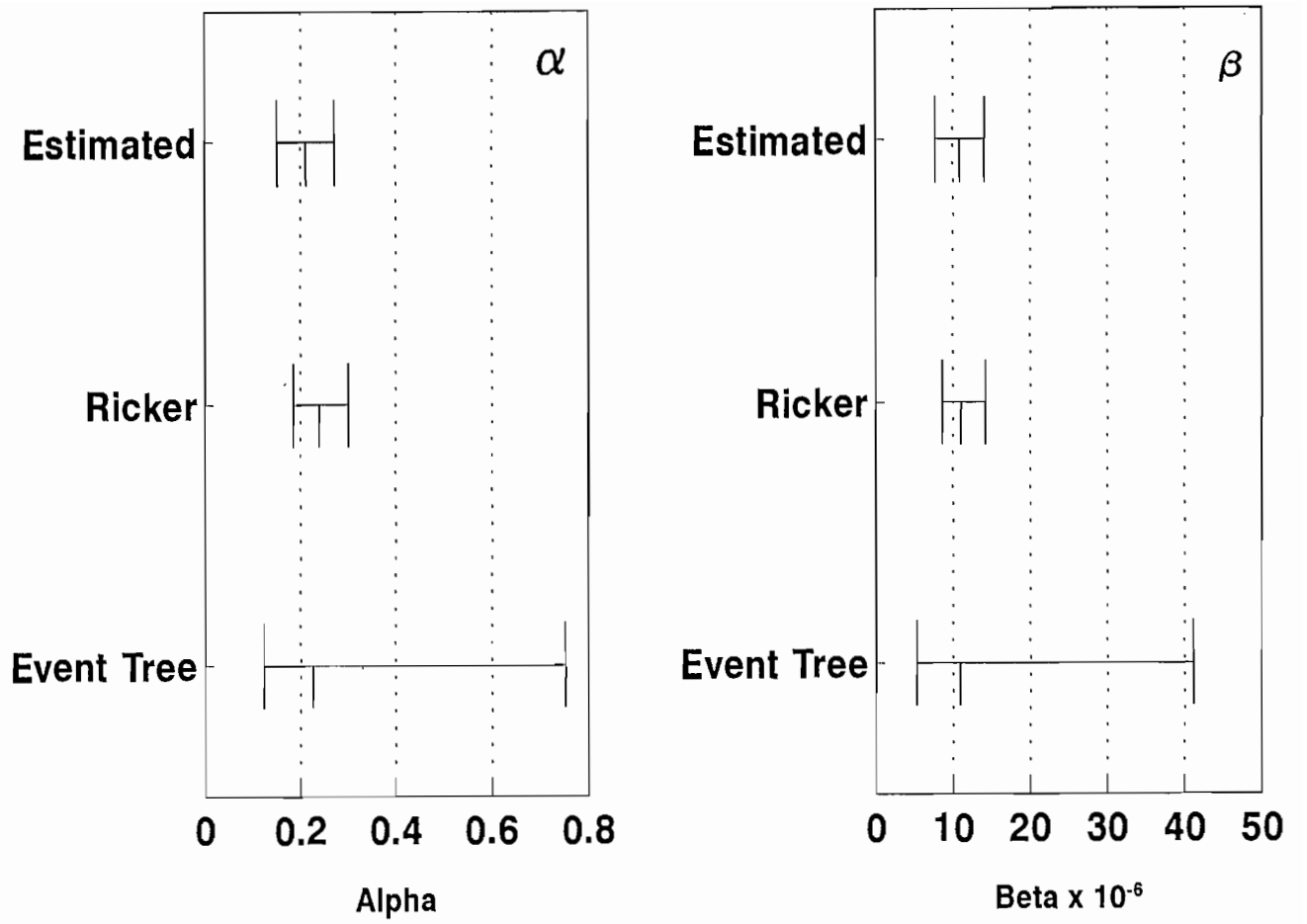


Fig. 4. Estimated Ricker parameters (α and β) with 90% confidence interval for Atlantic menhaden compared for the original estimated parameters, and those from simulated recruits from the Ricker and Event Tree relationships.

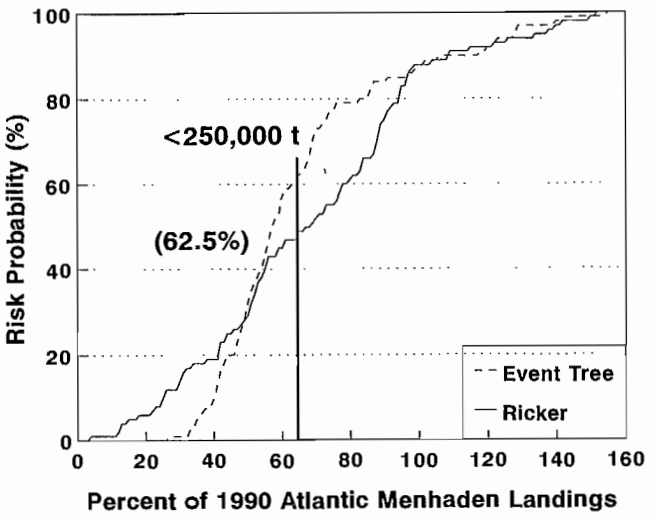


Fig. 5. Cumulative probability (or risk) of Atlantic menhaden landings in 2001 relative to those in 1990 based on Ricker and Event Tree relationships. Trigger value is less than 250,000 t or 62.5% of 1990 predicted landings. Projection instantaneous fishing mortality rate is based on maximum spawning potential of 4.5%.

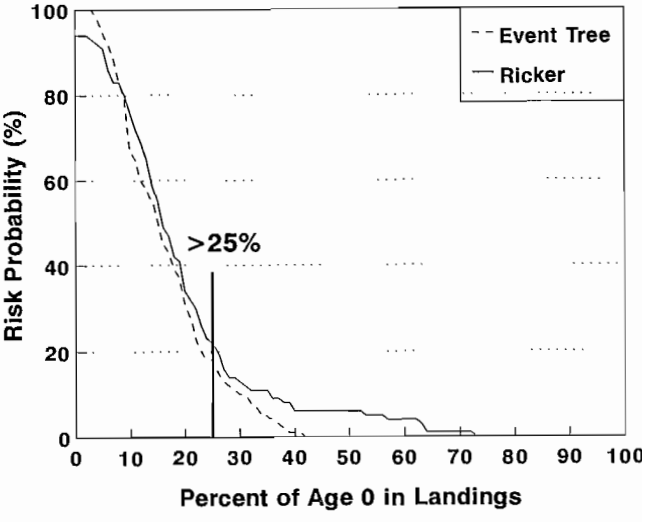


Fig. 6. Cumulative probability (or risk) of proportion of age 0 Atlantic menhaden in the landings by number in 2001 based on Ricker and Event Tree relationships. Trigger value is greater than 25%. Projection instantaneous fishing mortality rate is based on maximum spawning potential of 4.5%.

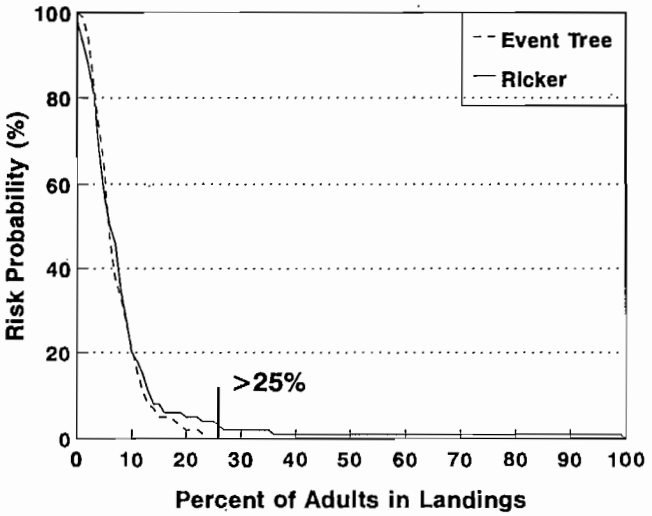


Fig. 7. Cumulative probability (or risk) of proportion of adult (age 3 and older) Atlantic menhaden in the landings by number in 2001 based on Ricker and Event Tree relationships. Trigger value is greater than 25%. Projection instantaneous fishing mortality rate is based on maximum spawning potential of 4.5%.

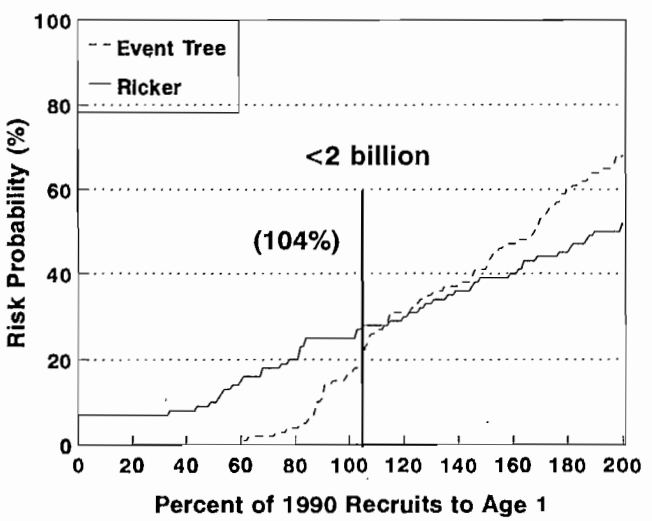


Fig. 8. Cumulative probability (or risk) of decline in recruits to age 1 Atlantic menhaden in 2001 relative to those in 1990 based on Ricker and Event Tree relationships. Trigger value is less than 2 billion recruits or 104% of 1990 recruits to age 1. Projection instantaneous fishing mortality rate is based on maximum spawning potential of 4.5%.

For the percent of adult menhaden (age 3 and older) in the catch in numbers, the trigger is activated if a value greater than 25% is obtained. Risk is defined as the cumulative probability of exceeding a given percent of adults in the catch in numbers (Fig. 7). The risk curves based on both relationships are very similar, with generally slightly higher values from the Ricker relationship, including at the trigger value where the difference is about 4%.

For the recruits to age 1, the trigger is activated when it

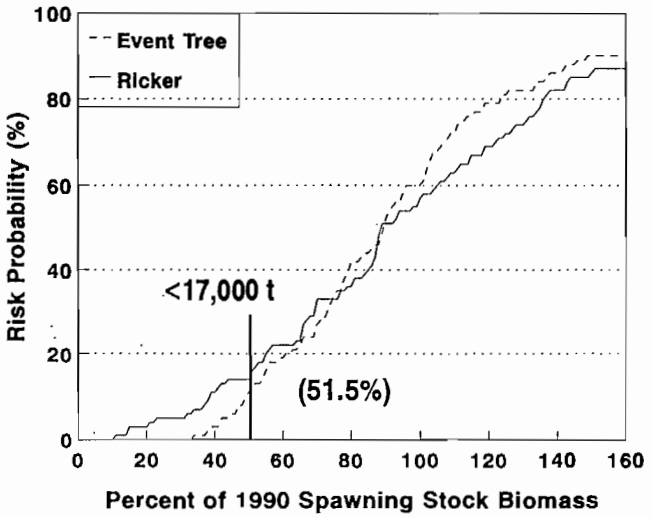


Fig. 9. Cumulative probability (or risk) of decline in Atlantic menhaden spawning stock biomass in 2001 relative to that in 1990 based on Ricker and Event Tree relationships. Trigger value is less than 17,000 t or 51.5% of 1990 spawning stock biomass. Projection instantaneous fishing mortality rate is based on maximum spawning potential of 4.5%.

falls below 2 billion or about 104% of the recruits to age 1 in 1990. The recruits to age 1 in 1990 is obtained deterministically from the recruits to age 0 in 1989 (part of the starting age structure). Risk is defined as the cumulative probability of a value less than or equal to a given value of recruitment (Fig. 8). The risk associated with the Ricker relationship is uniformly larger than that associated with the Event Tree relationship for recruitment values less than the trigger, and the difference is between 7 and 8% at the trigger value.

For the spawning stock biomass, the trigger is activated when it falls below 17,000 t or about 51.5% of the spawning stock biomass in 1990. Spawning stock biomass in 1990 through 1992 is fixed by the initial age structure in 1989. Simulated recruits to age 0 in 1990 do not appear in the spawning stock until 1993 as age 3 menhaden. Risk is defined as the cumulative probability of a value less than or equal to a given value of spawning stock biomass (Fig. 9). The risk associated with the Ricker relationship for low levels of spawning stock biomass is somewhat larger than that associated with the Event Tree relationship for spawning stock biomass, and the difference is about 3% at the trigger value.

Next, annual median spawning stock biomass from the 12-year population projections based on the two relationships are compared for five levels of F (Fig. 10). The Ricker model seemingly prevents the spawning stock biomass from getting very large when F equals 0 (MSP = 100%) (Fig. 10a). Hence, the ratio of the median spawning stock biomass for the first four levels (4.5, 10, 20, and 30) to that for $F = 0$ in 2001 are larger than suggested by the density independent approach used in formulating %MSP. Median spawning stock biomass from the Event Tree relationship is not similarly constrained when $F = 0$ (Fig. 10b), although it is constrained by the

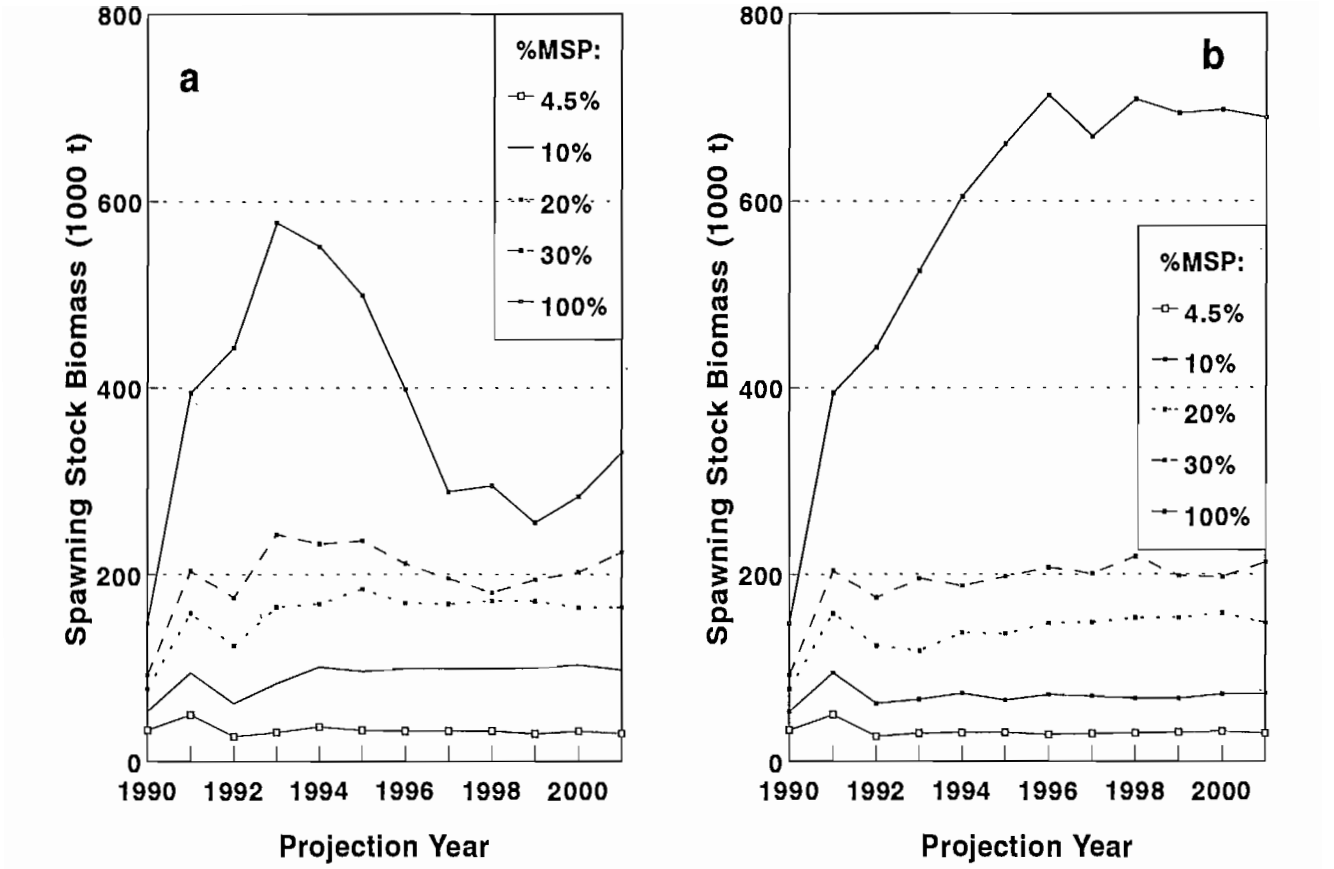


Fig. 10. Median spawning stock biomass of 100 replicate population projections of 12 years (1990–2001) for the Ricker (a) and Event Tree (b) relationships with projection instantaneous fishing mortality rate based on 5 levels of maximum spawning potential (4.5, 10, 20, 30, and 100%).

gamma distribution placed on the high category of recruits to age 0. The ratio of spawning stock biomass in 2001 closely approximates that suggested by %MSP. Similar values of median spawning stock biomass for values of MSP less than or equal to 30% are obtained from the two relationships.

Finally, the risk of spawning stock biomass is compared for the five levels of MSP in the year 2001 between the two simulation relationships (Fig. 11). For the simulations based on the Ricker relationship, there is a large drop in risk of a decline in spawning stock biomass between F based on an MSP equal to 4.5% and MSP equal to 10% and considerable overlap of the risk curves for MSP of 10%, 20%, and 30% (Fig. 11a). However, for the simulations based on the Event Tree relationship, there is a more gradual decline in risk with decreasing F's associated with increasing MSP's (Fig. 11b).

Discussion and Conclusions

In this section, I begin by discussing several questions that arose in the course of this study. This is followed by a discussion of what conclusions relevant to management of the Atlantic menhaden might be drawn from these simulations.

While setting up the simulations, the question arose as to how many variables should be simulated with error. Because the purpose of this study is to compare the relative risks associated with the two relationships for spawning stock biomass with subsequent recruits, error was only simulated in obtaining recruits to age 0 from spawning stock biomass. The simulation of additional error in other population model parameters would only tend to obscure the differences between the two relationships.

The sensitivity of the estimates of risk to underlying probability distributions was not explored. In this analysis, error for the Ricker and Event Tree relationships was simulated using the normal, uniform, and gamma distributions. How sensitive the final results might be to these assumptions has not been explored and is open to further study by simulations.

Is there an underlying mechanistic model (i.e., the Ricker model) that relates spawners and subsequent recruits for predicting future Atlantic menhaden populations? Is this relationship simply obscured by large environmental variability as is usually suggested? Or is the use of conditional probabilities more appropriate? As noted in the foregoing simulations, there are differences in the risks, but simulations such as those described here are not able to address the appropriateness of the two relationships.

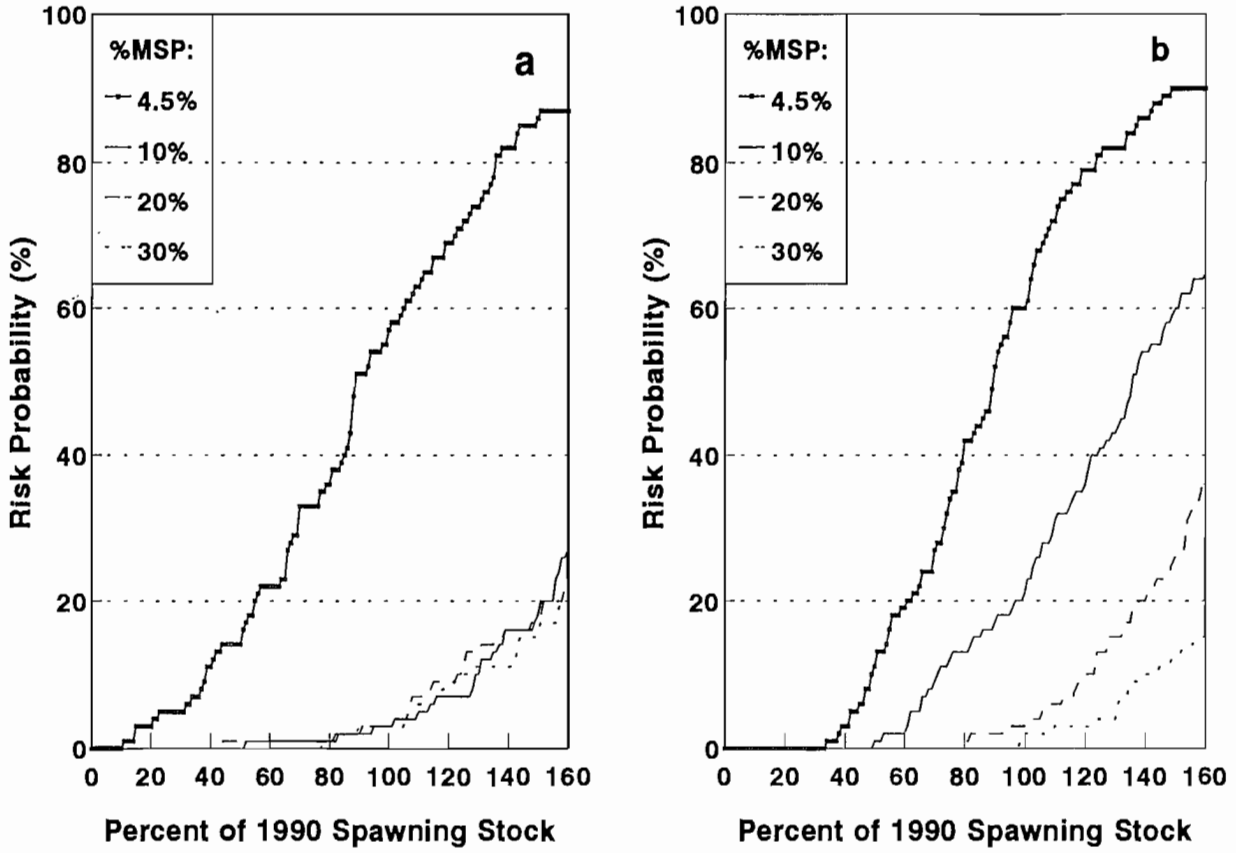


Fig. 11. Cumulative probability (or risk) of decline in Atlantic menhaden spawning stock biomass in 2001 (relative to 1990) for the Ricker (a) and Event Tree (b) relationships with projection instantaneous fishing mortality rate based on 4 levels of maximum spawning potential (4.5, 10, 20, and 30%).

It is interesting to note that simply by applying the conditional probabilities of the Event Tree relationship to the historical set of spawning stock biomass to obtain corresponding recruits to age 0, relatively unbiased estimates of the Ricker model are obtained when compared to the original data (Fig. 4). However, as should be expected the variability in estimates of the Ricker parameters is much larger when simulated from the Event Tree relationship as compared with the variability when simulated from the Ricker relationship. The latter variability is very similar to that found in the original Ricker nonlinear regression.

The greatest differences in risk at the trigger values were found for landings in weight and recruits to age 0 (Figs. 5 and 8). The landings in weight is the only trigger variable for which the risk is greater for the Event Tree relationship at the trigger value (12% at 250,000 t or 62.5% of landings in 1990). For reductions in landings below about 50% from the 1990 level, risk associated with the Ricker relationhip was higher; while reductions in landings between about 60% and 100% from the 1990 levels, risk associated with the Event Tree relationship was higher. For the remaining trigger variables, the risk associated with the Ricker relationship is uniformly greater for more excessive levels of the trigger value (i.e., larger percents of age 0 and adults in landings by number, and smaller values of recruits and spawners).

It is not surprising that there is a large difference in risk for the recruits to age 1 (7 to 8% at the trigger value of 2 billion or 104% of recruits in 1990), because that is the level at which basic uncertainty is being simulated. Greater differences in risk would be expected in landings compared with spawning stock biomass (only about 3% at 17,000 t or 51.5% of 1990 level), because the latter trigger variable contains fewer ages (3–8 compared to 1–8). Small differences in risk are associated with the two triggers comparing age 0 and adult menhaden in the landings in numbers; 5% versus 4%.

With respect to the risk of a particular trigger value being exceeded (i.e., activated), the risk associated with the Ricker relationship appears to be activated more often. Activation of the recruit and spawning stock trigger variables are by the nature of the variables indicative of poor stock health, whereas activation of the first three triggers may give "false firings". But what should be of particular interest to managers is the difference in advice that might be suggested when comparing the risk curves based on different levels of F associated with maximum spawning potential (Fig. 11).

For the Ricker relationship, a risk of a decline in landings or spawning stock biomass drops precipitously from an MSP of 4.5% to 10%. For greater improvements in MSP, there is no apparent gain in reduction of risk remaining to be accrued. A

quite different picture arises from the Event Tree relationship: a more gradual decline in risk is observed with the decrease in F associated with increasing MSP from 4.5% to 10% to 20% and to 30%. The mathematical nature of the Ricker model is undoubtedly the cause for this difference (compare Figs. 10a and 10b for median values of spawners at the MSP of 100%).

In conclusion, if the Ricker relationship best represents the underlying relation for spawners with recruits, then a goal for MSP higher then 10% may not be needed. However, if the Event Tree relationship better represents the underlying spawner-recruit relationship, then a goal greater than 10% MSP may be needed. Unfortunately, there is no absolute way to determine which relationship best represents the future course of the Atlantic menhaden stock. Regardless of which relationship is more realistic, there is substantial reductions in risk by adoption of a 10% MSP as the management goal for Atlantic menhaden.

Acknowledgements

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Choosing A Management Strategy for Stock Rebuilding When Control is Uncertain

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For many marine resources which are currently overexploited, the task of management is to devise a strategy for rebuilding the stock over a given period to restore productivity and stability to the fishery. Most studies have accounted for the inherent uncertainty in recruitment and some, for the measurement error in the assessment process. Another important source of variation is the ability of management to achieve a particular harvest rate in any one year. In this study we explore the effects of this source of uncertainty, which we refer to as implementation uncertainty, on stock rebuilding strategies for southern New England yellowtail flounder. Four families of simulations are presented. The first case serves as a "control", where the achieved harvest rate is fixed, with no implementation uncertainty. In the second case, random variation in the achieved harvest rate is included to examine its effects on the rebuilding program. In the third experiment, the distribution of the harvest rate, including implementation uncertainty, is a positive function of the size of the fully recruited stock, to mimic the effects of an effort-controlled multispecies fishery where vessels could switch between target stocks. Finally, in a fourth experiment the harvest rate distribution is a negative function of the size of the fully recruited stock, approximating the effects of a catch quota. Simple random variation in fishing mortality rates does not substantially affect the expected yield and spawning biomass although it does modify the distribution of these quantities. However, when there is a relation between implementation uncertainty and stock abundance, the form of this relation is relevant. Regulation of effort that leads to switching between stocks causes an increased risk to the spawning biomass for similar yield patterns. Conversely, regulation through quotas produces lower expected yields but a higher expected spawning biomass.

Dans le cas de nombreuses ressources marines actuellement surexploitées, le défi de la gestion est d'arriver à une stratégie de reconstitution du stock dans une période donnée afin de restaurer la productivité et la stabilité de la pêche. La plupart des études ont tenu compte de l'incertitude inhérente au recrutement et, quelques-unes, de l'erreur de mesure lors de l'évaluation. L'aptitude de la gestion à obtenir un taux de prise particulier en une année est une autre importante source de variation. Dans cette étude, nous explorons l'incidence de cette source d'incertitude, que nous appelons «incertitude de mise en oeuvre», sur les stratégies de reconstitution du stock de limande à queue jaune du sud de la Nouvelle-Angleterre. Quatre cas de simulations sont présentés. Le premier cas sert de «témoin«: le taux de prise obtenu est fixe, sans incertitude de mise en oeuvre. Dans le deuxième cas, la variation aléatoire du taux de prise obtenu est prise en considération afin d'examiner son incidence sur le programme de reconstitution. Dans le troisième cas, la distribution du taux de prise, y compris l'incertitude de mise en oeuvre, est reliée positivement à la taille du stock complètement recruté, afin d'imiter les conséquences d'une pêche polyvalente contrôlée dans le contexte de laquelle les bateaux peuvent passer d'un stock cible à un autre. Enfin, dans le quatrième cas, la distribution du taux de prise est reliée négativement à la taille du stock complètement recruté, imitant ainsi les conséquences des contingentements de pêche. La simple variation aléatoire des taux de mortalité par pêche n'affecte pas significativement le rendement prévu ni la biomasse génitrice, quoiqu'elle influe sur la distribution de ces quantités. Toutefois, lorsqu'il y a détermination de relation entre l'incertitude de mise en oeuvre et l'abondance du stock, la forme de cette relation est significative. La réglementation qui encourage le passage d'un stock à un autre augmente le risque pour la biomasse génitrice, en situation de rendement identique. Inversement, la réglementation par contingentements entraîne une diminution du rendement prévu, mais une augmentation de la biomasse génitrice prévue.

Projections in time of resource characteristics such as yield, spawning biomass or population age structure under different regulatory scenarios form the basis of scientific advice for managers. Such projections are often made using different assumptions concerning future recruitment (e.g., NEFSC 1991), a major source of uncertainty for resource status in the future. Monte Carlo simulations with some assumptions about recruitment variability have been

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used to make such projections and develop probability statements concerning future yields and risk to the resource (e.g., Overholtz et al. 1986; Rosenberg and Brault 1991).

More extensive simulation studies developed by Restrepo et al. (1991) have incorporated crucial aspects of the uncertainty in stock assessments to evaluate the risk of management strategies. A third source of uncertainty is in the degree of control over the harvest accompanying a particular management decision, which may be termed implementation uncertainty. Management advice is usually made assuming that a target harvest rate, say, will be perfectly achieved, although this is widely recognized not to be the case in practice.

In this paper, we have carried out simulation studies similar to those in our earlier work on stock rebuilding strategies (Rosenberg and Brault 1991), but including implementation uncertainty. The simulations are designed to examine potential stock rebuilding strategies for the southern New England yellowtail flounder stock as an illustration of the effects of implementation uncertainty on management advice. The reader should not take these studies as exhaustive with respect to advising on yellowtail flounder management. More extensive work, with input from managers to guide the analysis, is needed on this overexploited stock to develop a rebuilding strategy.

Here, we evaluate risk to yield and to the resource by comparing two arbitrarily chosen regulatory schemes to the case where no action is taken. Regulation is focused on the fully recruited harvest rate. Rebuilding schemes are to reduce the harvest rate from its recently estimated instantaneous fishing mortality rate of 1.6 down to 0.5, the fishing mortality rate corresponding to 20% of the maximum spawning biomass per recruit for this stock (NEFSC 1991). The two rebuilding options are to stage this reduction over 5 years or 10 years (Table 1). These time scales were chosen because they are in the range under consideration by fishery managers in New England for groundfish rebuilding programs.

The Southern New England stock of yellowtail flounder was recently estimated to be severely overexploited and at low stock abundance levels (NEFSC 1991). The fishing mortality rate on fully recruited flounder is more than three times the target level for this stock set by the New England Fishery Management Council. The catch in 1990 increased as a result of the recruitment of the large 1987 year class. Subsequent recruitment is estimated to be low.

Methods

The simulation model uses standard fishery equations for the population dynamics and catch, but includes gamma distributed random variation in the fishing mortality rate on each age group. The model is

$$
\begin{aligned}
N_{1,t} &= R_t \sim \Gamma\left(a', b', c'\right), \\
N_{i,t+1} &= N_{i,t} e^{-Z_{i,t}}, \\
Z_{i,t} &= X_t F_{i,t} + M, \\
X_t &\sim \Gamma\left(a, b, c\right), \\
C_{i,t} &= \frac{X_t F_{i,t}}{Z_{i,t}}\left[1 - e^{-Z_{i,t}}\right] N_{i,t}, \\
Y_t &= \sum_{i=1}^{\infty} C_{i,t} W_i
\end{aligned}
$$

where Γ represents the gamma density function with parameters a, b, c, $N_{i,t}$ is the number in the population and $C_{i,t}$ is the catch in numbers of fish age i in year t. $Z_{i,t}$ is the total instantaneous mortality rate on fish age i in year t and X_t is a multiplier in year t on the fishing mortality rate, $F_{i,t}$, varying between 0 and 2. M is the constant instantaneous rate of natural mortality set at 0.2. Y_t is the yield in weight, W_i. R_t is the number of recruiting fish at age 1 in year t.

Starting values for the simulations were taken from the most recent assessment of the stock (NEFSC 1991). The stock sizes and fishing mortality rates were for 1990. The gamma distribution for recruitment was fitted to the existing recruitment data for 1973–1990, with estimated parameters $a' = 2.0$, $b' = 17.5$ and $c' = 0$, resulting in a distribution skewed to the left. Although the recruitment is not a function of stock size in these simulations, there is the assumption made in interpreting the results that both increased stock size and an increase in the number of age groups contributing to the population will benefit recruitment and long term stability of the population.

Table 1. Summary of the simulated cases discussed in the paper.

a)

Strategies	
status quo	No reduction in fishing mortality from the 1990 level.
5 year rebuilding	Reduction of the fully recruited fishing mortality rate from the 1990 level to 0.5 over five years.
10 year rebuilding	Reduction of the fully recruited fishing mortality rate from the 1990 level to 0.5 over ten years.

b)

Case	Distribution of X_t	Gamma Parameters[1]	Effect
0	1	—	deterministic F_{it}
A	Gamma	$a = 2$, $b = 0.2$, $c = 0.7$	randomly varying F_{it}
B	Gamma		effort control, switching between target stocks
C	Gamma		target quota

[1] for the fishing mortality rate multiplier X_t.

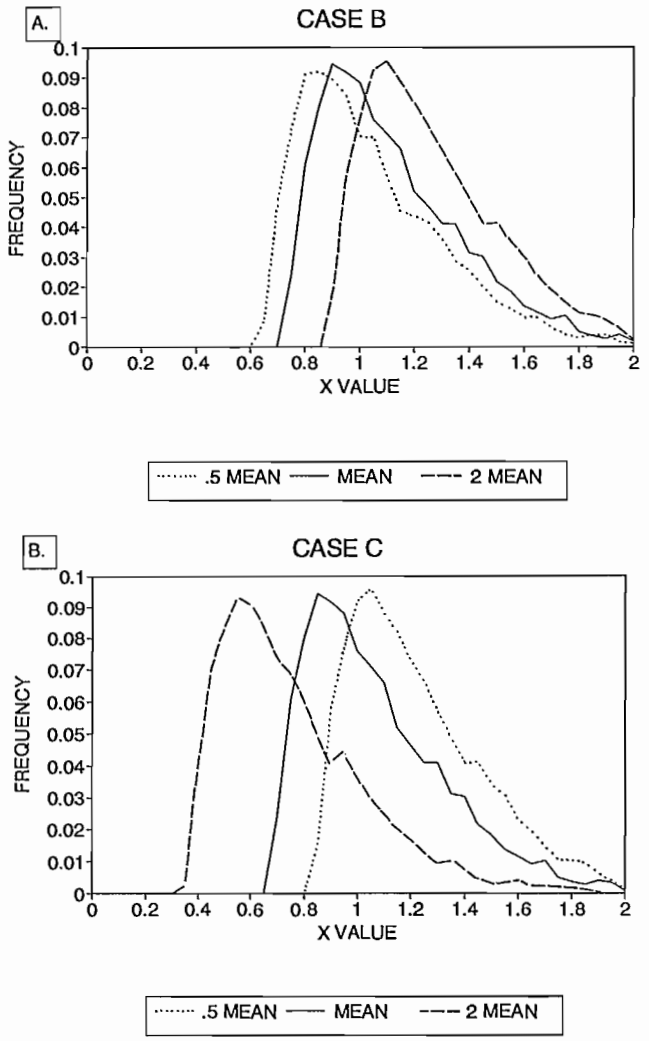


Fig. 1. Distribution of fishing mortality rate multipliers, X_t, at different stock levels. The dashed line is at stock abundance one half the mean, the solid line is at mean abundance and the dotted line is for twice the mean abundance. A. case B where fishing mortality rate increases on average with stock abundance. B. case C where fishing mortality rate decreases on average with stock abundance.

We compared four sets of simulations, summarized in Table 1. The first set (Case 0) had fixed harvest rates ($X = 1$) in each year corresponding to each management strategy. This was the base case for comparison of the effects of implementation uncertainty.

In the second set (Case A), the fishing mortality rate varied around the deterministic rate given by each management strategy, as a result of the gamma distributed variation in the value of the multiplier X_t, which was applied to all age groups maintaining the partial recruitment pattern. The use of a gamma distributed X_t allows a distribution of fishing mortality skewed to the left, i.e., with a larger tail at higher values of $F_{i,t}$. This corresponds to a situation where fishing mortality is more likely to be much higher rather than much lower than the mean. The parameter values chosen for the distribution of X_t (Table 1) give a mean of 1.1 (median of 1) and a coefficient of variation of 26%. We also made the assumption that fishing mortality rate should not increase by more than a factor of two in any one year, therefore the distribution of X_t was truncated at two.

In the third and fourth sets (Cases B and C) fishing mortality rate on the fully recruited ages varied with a gamma distributed multiplier X_t where the scaling parameter c varied as a linear function of fully recruited stock abundance (Table 1; Fig. 1). Case B, with the scaling parameter a positive function of abundance (Table 1), was designed to approximate the effects of an effort controlled multispecies fishery where vessels could switch between target stocks. In this case, a stock at above average abundance will attract additional fishing effort (Fig. 1a).

In Case C the scaling parameter was a negative function of abundance (Table 1), to approximate the effect of a regulated fishery where harvest is capped by a catch quota. Here, the fishing mortality rate on fully recruited ages decreases with high abundance (Fig. 1b). In both Cases B and C, when abundance is at the average level from the historical time series (NEFSC 1991), the scaling parameter, c, equals 0.7, as in Case A.

For each case, we calculated the expected (average across all 1000 runs) yield, spawning stock biomass and skewness of the age composition in each year of the 13 years simulated.

Each set contains 1000 runs for each of the three rebuilding strategies; no harvest rate reduction, reduction in harvest rate over five years (from 1.6 to 0.5 in steps of 0.22 and then constant at 0.5) or reduction over 10 years (from 1.6 to 0.5 in steps of 0.11 and then constant at 0.5) to the target level of the fishing mortality rate on the fully recruited age groups equal to 0.5. The partial recruitment pattern at age was kept constant across all runs. The stream of random recruitment numbers was common across sets of runs to enable comparison. The random number streams for the implementation uncertainty were also common across each set of runs, although because of the distribution of fishing mortality rate is affected by stock abundance in some of the cases, the X_t are not the same across cases.

We then calculated the risk to average (across all years) yield and spawning biomass for each strategy. Risk here was defined as the probability of the yield or biomass being less than a given level. We also calculated, for each of the rebuilding programs, the probability that the yield and biomass was less than would have been obtained if the fishing mortality rate was not reduced. Since the random number sequences were the same in the runs under each management strategy, we could make pairwise comparisons between rebuilding strategies for each simulation run to calculate this probability.

Results

In examining the results we are concerned with differences in the prognosis for the stock for different underlying

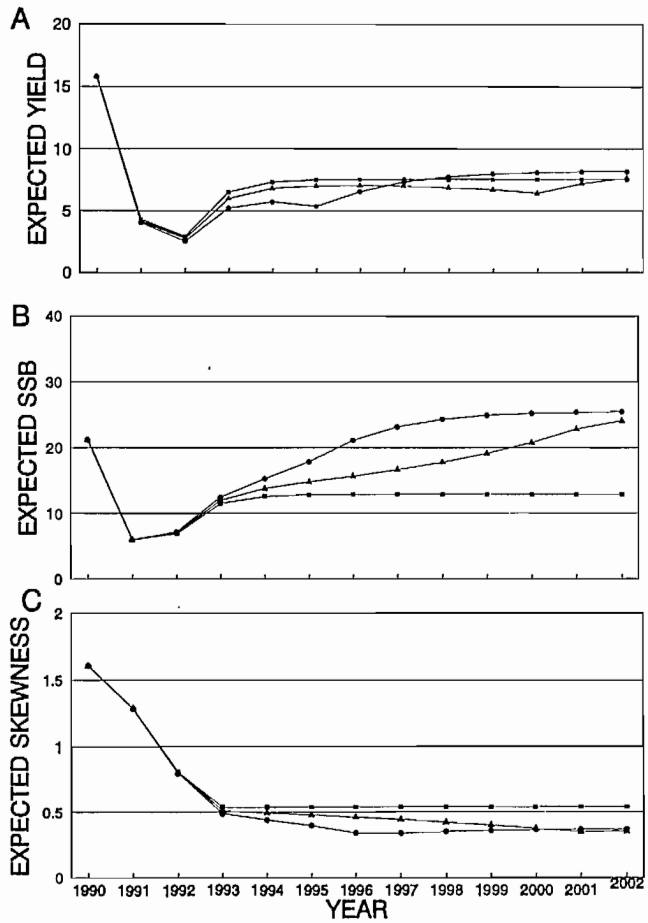


Fig. 2. Case 0. a) expected yield in each year for each management scenario obtained by averaging over all runs; b) expected spawning biomass in each year for each scenario; c) expected skewness of the age composition in each year for each scenario, no reduction (squares), five year rebuilding program (circles) and ten year rebuilding program (triangles).

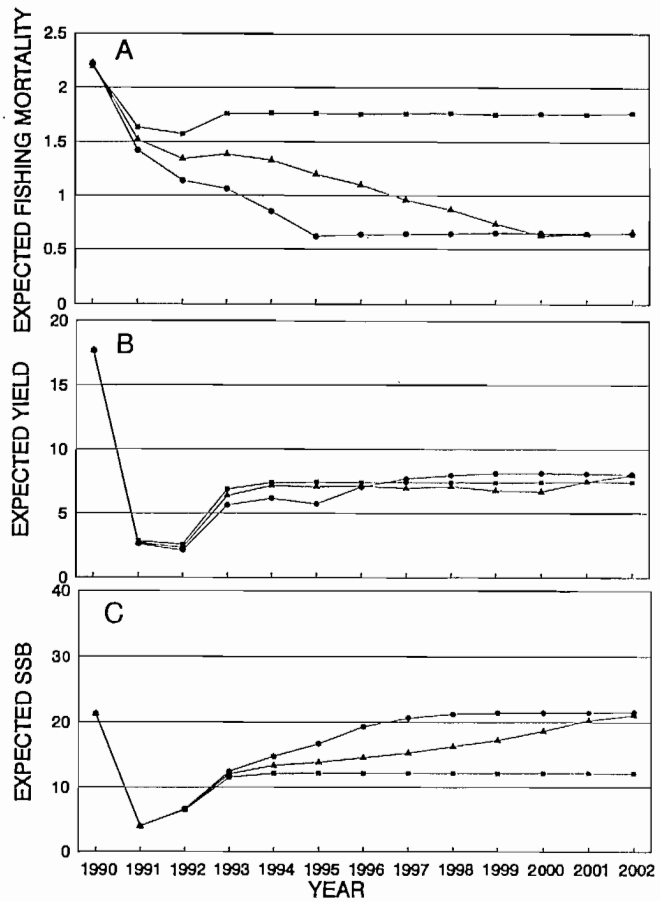


Fig. 3. Case B. a) expected fishing mortality rates over time for each management scenario obtained by averaging over all runs, b) expected yield in each year for each management scenario, c) expected spawning biomass in each year for each scenario. Plotting symbols as defined in Fig. 2.

model structures (Case 0–C) and between the two rebuilding strategies.

Patterns of Expected Yield and Spawning Biomass

For the base case (0) fishing mortality rate pattern all management strategies result in very similar levels of expected yield in each year of the simulation (Fig. 2a). An increased spawning biomass (Fig. 2b) compensates for the decreases in fishing mortality rate for each of the rebuilding strategies, so that the expected yield remains relatively constant. The implication is that the same level of yield can be taken with much reduced effort, i.e., catch rates increase inversely with fishing mortality rate reductions. The skewness of the age distribution of the population also decreases as the harvest rate is reduced (Fig. 2c). If more age groups are beneficial to spawning stock productivity, by, for example, providing a buffer against one or two years of poor recruitment, then a reduced harvest rate should benefit the resource in the long term. In Case A (not shown), where the fishing mortality rate randomly varied, the expected fishing mortality pattern was very similar to the base case. The resultant patterns of expected yield, spawning biomass and skewness were also similar to the base case.

In Case B, the distribution of the multiplier X_t is shifted to produce higher mortality rates at higher stock abundances (Figures 1a and 3a). The expected yield patterns (Fig. 3b) are very similar to the previous cases because, while fishing mortality rate is often higher, biomass is lower (Fig. 3c). Expected rebuilding of the stock is slower because of the fleet's ability to focus on high abundance and increase fishing mortality from the regulated level.

In Case C, the fishing mortality rate multiplier, X_t, is a decreasing function of stock abundance, as would occur if there was a fixed quota cap on the harvest (Fig. 1b). As a result, the expected fishing mortality rate is higher in low abundance years such as 1991 and 1992 and low when abundance is high, such as in the first year in these simulations (Fig. 4a). The payoff is to greatly increase expected spawning biomass when fishing mortality is reduced through the rebuilding programs (Fig. 4c) while yield remains stable (Fig. 4b).

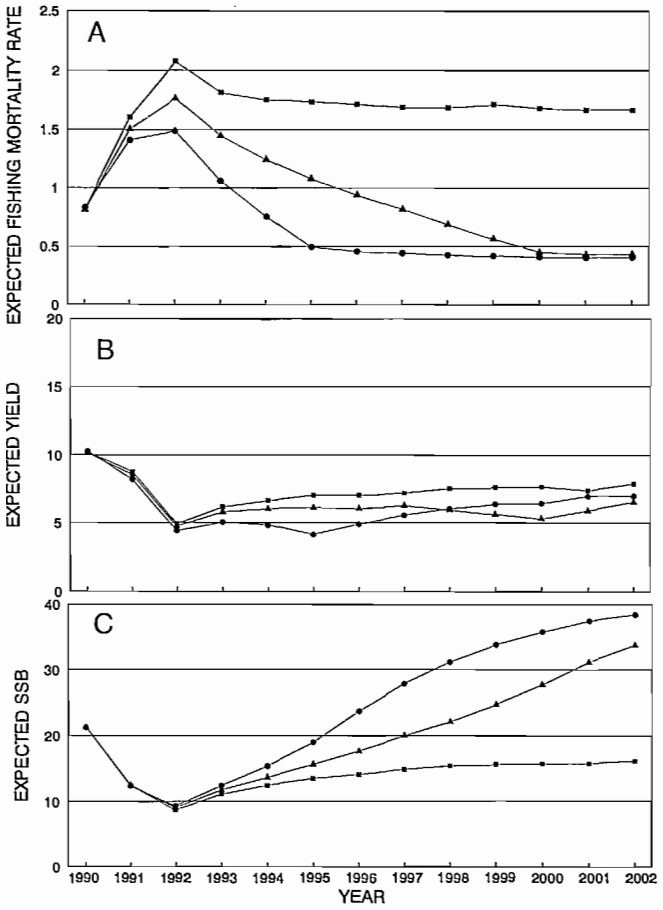


Fig. 4. Case C. a) expected fishing mortality rates over time for each management scenario obtained by averaging over all runs, b) expected yield in each year for each management scenario, c) expected spawning biomass in each year for each scenario. Plotting symbols as defined in Fig. 2.

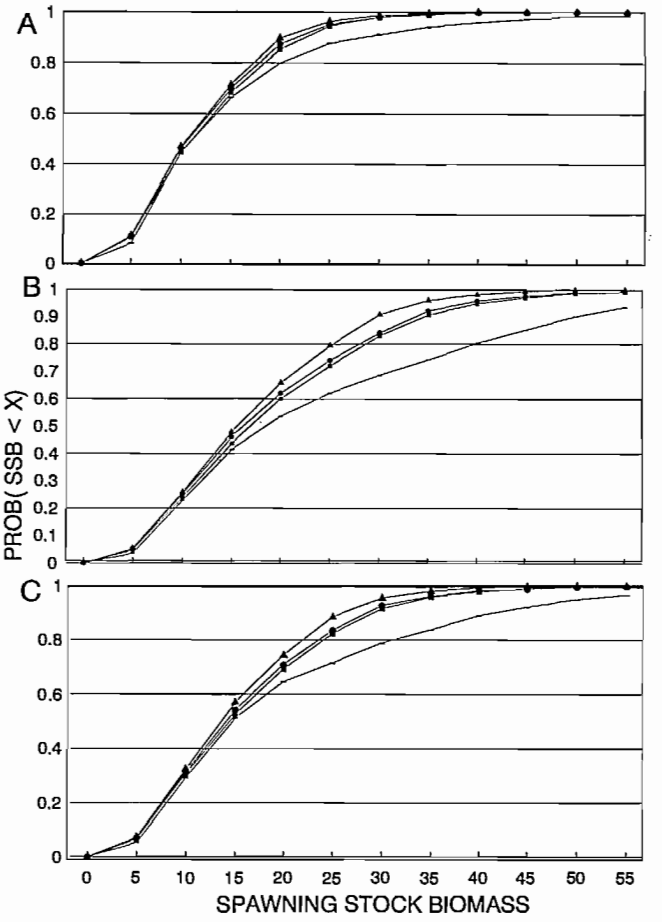


Fig. 5. Comparison of risk curves for each management scenario for Case 0 (squares), Case A (circles), Case B (triangles) and Case C (dashes). The ordinate is the probability that the spawning biomass is less than the given value on the abscissa. a) no reduction strategy, b) five year rebuilding program, c) ten year rebuilding program.

Probability Profiles for Yield and Spawning Biomass

The distribution of spawning biomass across runs in Case 0 and A is very similar (Fig. 5). These probability profile curves are shifted to the left for the strategies which reduce the harvest rate more slowly or not at all. This means there is a higher probability of low biomass by the end of the time period if the mortality rate is not reduced, but with little difference with or without simple implementation uncertainty. However, case B, mimicking an effort control program, results is an increased chance of lower spawning stock biomass than case A particularly for the 5 year rebuilding program (Fig. 5 b). For case C, mimicking a quota cap on the harvest, on the other hand the prognoses for the spawning stock is improved, with a lower chance of low stock abundance under any of the management scenarios (Fig. 5).

Case 0 and A give very similar patterns of expected yield and spawning biomass, but do differ in the distribution of these quantities across the simulation runs. This is shown by comparing the probability of each of the rebuilding strategies giving average yield less than would be obtained by maintaining the status quo (Fig. 6). For case 0 this is nearly a step function, where yield from the rebuilding program is always less than the status quo until 1998 for the five year program and 2002 for the 10 year program. On the other hand, without implementation uncertainty (Case 0) the rebuilding programs always result in higher spawning biomass after 1992 (Fig. 7). When simple implementation uncertainty is included (Case A) then these step functions are modified so that there is a small probability of yield being greater than the status quo strategy by 1996, but there is also a chance, toward the end of the runs, that the rebuilding program may not result in improved yield (Fig. 6).

When the implementation uncertainty is a function of stock abundance then there is a substantial reduction in the probability that yield will be less than status quo during the beginning of the time series (Fig. 6). There is only a slightly greater than 50% probability that yield will be less than the status quo strategy for cases B and C in the second simulation year. This reflects an increased variance of yield resulting from the increased exploitation of good year classes. The distribution of spawning stock biomass for the rebuilding strategies in this case also overlaps the status quo (no reduction)

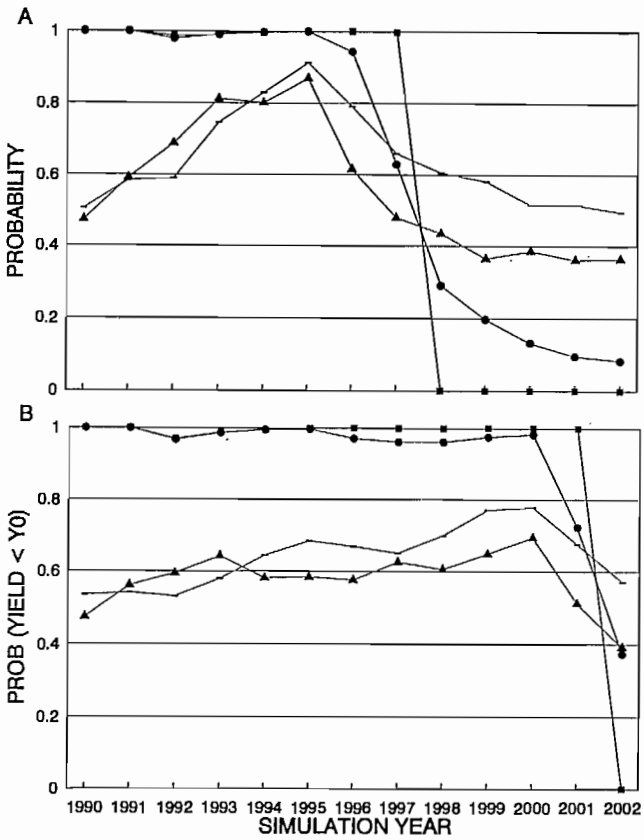


Fig. 6. Comparison of the yield from the rebuilding programs compared to the status quo (no reduction) management yield. The ordinate is the probability that the yield was less than that obtained from the status quo in the year given on the abscissa. a) five year rebuilding program, b) ten year rebuilding program.

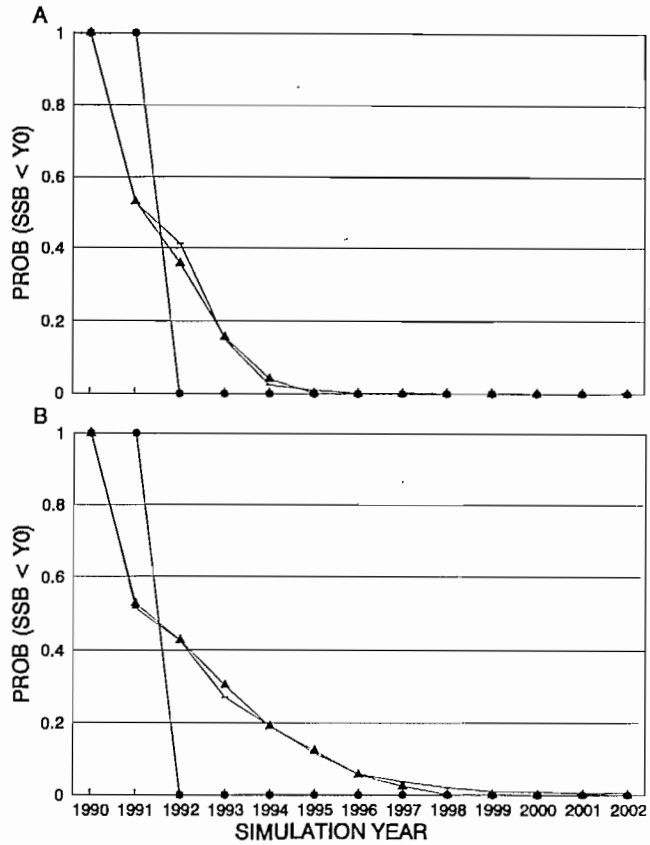


Fig. 7. Comparison of spawning biomass from rebuilding programs compared to the status quo. The ordinate is the probability that the yield was less than that obtained from the status quo in the year given on the abscissa. a) five year rebuilding program, b) ten year rebuilding program. Plotting symbols as in Fig. 5.

distribution. In effect this means that the rebuilding programs may not necessarily result in increased spawning biomass when the harvest rate varies in response to stock abundance (Fig. 7).

Rebuilding Strategies

We can compare the efficacy of the two rebuilding strategies by considering the probability that the yield and spawning biomass in the ten year strategy are less than in the five year strategy in any year (Fig. 8). For Cases 0 and A, the ten year strategy always gives a higher yield in the first half of the series and a lower yield in the second half; spawning biomass is always higher for the 5 year strategy. A similar, but not so extreme pattern results in Cases B and C. Yield is likely to be lower in the five year strategy in the early years, and higher for the later years. Spawning biomass is likely to be higher for the five year strategy; however, there is a substantial chance that the ten year program would do just as well in these two cases.

Discussion

The risk to both yield and the resource base can be strongly affected when the ability of managers to achieve a regulated harvest rate varies with the abundance of the stock. Simple random variation in fishing mortality rates does not substantially affect the patterns of risk or expected yields and biomasses.

When the harvest rate is affected by stock abundance it is important to know, at least, the sign of the slope of the relationship. In a multispecies fishery, such as in southern New England, it is sometimes possible for fishing effort to shift target onto more abundant species. In such cases, stock rebuilding programs which rely on regulation of total effort could be seriously undermined, although yields may be maintained. Note again the implicit assumption in the interpretation of our work that high stock biomass and fuller age structure will engender more stable good recruitment to the stock in the long term.

If higher stock abundance cannot be capitalized upon, due to market conditions or a target catch quota, such that the rate of fishing mortality is likely to decrease with high stock, there will probably be a loss of yield unless the quota is continually readjusted. Rebuilding will likely be more rapid and the risk

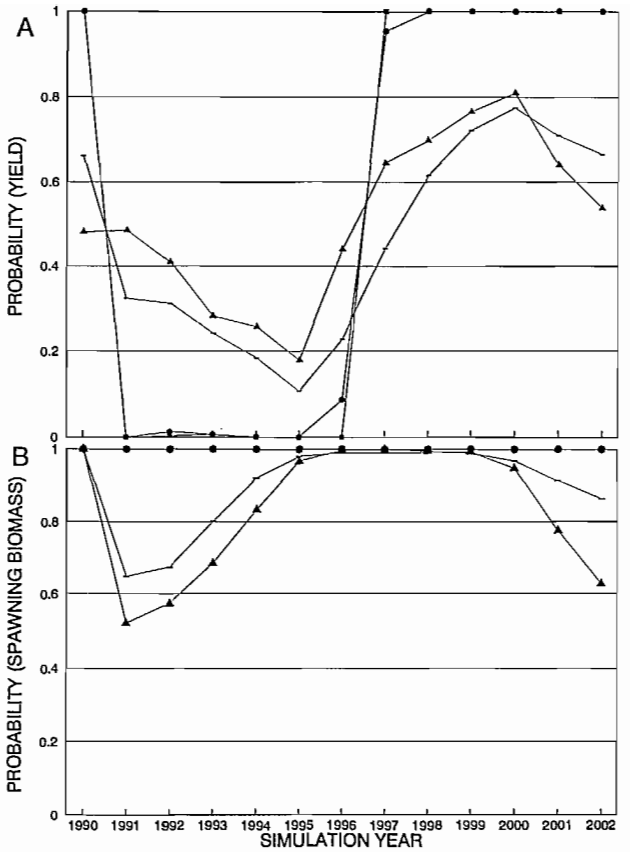


Fig. 8. Comparison of the five year and ten year rebuilding programs. a) probability that the yield from the ten year program is less than from the five year program for each case, b) probability that the spawning biomass from the ten year program is less than from the five year program. Plotting symbols as in Fig. 5.

to the stock will be less (of the order of 15–20% for a five year rebuilding program in this example; Fig. 4b) at the cost increased risk to yield (of the order of 5–10% for the five year program here; Fig. 5b).

The advice on the appropriate time scale for the rebuilding program will also depend on the form of implementation uncertainty. When the fishing mortality rate varies randomly around the regulated level, this decision will be based on tradeoffs between short term losses in yield versus longer term benefits in stock biomass and increased yields. However, the issue is less clear cut when the distribution of fishing mortality varies as a function of stock abundance. In Case B, where targeting to take advantage of increased abundance is possible, the yield from the ten year strategy may be nearly risk neutral (50% probability that the yield will be less than the status quo strategy) in our example (Fig. 6). This is an attractive property for managers since it implies less disruption of the industry. The five year strategy will more than likely result in reduced yields in the short term. However, the five year strategy is always likely to give a higher stock size than the ten year strategy over the whole period (Fig. 8).

The changes in prognosis and risk can occur with relatively small levels of implementation uncertainty. In our simulations, the coefficient of variation for the fishing mortality rate multiplier is of 26%. This uncertainty results in changes in risk of the order of 10–20%.

The choice between rebuilding time scales will ultimately depend upon the initial condition of the stock. In this paper, the yellowtail flounder stock is projected from a severely overexploited condition because of the lack of strong year classes after the 1987 cohort and the extremely high fishing mortality rate. Although maintaining yield near the status quo level may be possible with a slow reduction in fishing mortality rate, the actual yield levels are very small compared to the potential once the stock is rebuilt. This may indicate that rapid increases in stock biomass should take precedence over short term maintenance of yield. In other cases, the reverse may be true. Our earlier work on Georges Bank cod comparing two year and five year rebuilding programs illustrates this point (Rosenberg and Brault 1991). Of course, this tradeoff must be determined by managers, but it must be placed in the proper context of long term potential yield to allow an informed decision.

Argue et al. (1983) have presented an approach for comparing various management strategies with multiple objectives. They do not explicitly incorporate uncertainty in their analysis of management tradeoffs, but show that choosing management constraints and contouring objective functions in the plane of regulatory measures (e.g., quota and effort regulation) may quickly delineate feasible management scenarios. Here we have tried to incorporate uncertainty and risk in the analysis of multiple objectives. Future analyses must seek to determine acceptable levels of risk, possibly through combining our approach with that taken by Argue et al. (1983).

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Session III
Alternative Strategies and Reference Points

Risks and Uncertainties in the Management of a Single-Cohort Squid Fishery: the Falkland Islands *Illex* Fishery as an Example

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Basson, M. and J. R. Beddington. 1993. Risks and uncertainties in the management of a single-cohort squid fishery: the Falkland Islands *Illex* fishery as an example. p. 253–259. *In* S. J. Smith, J. J. Hunt and D. Rivard [ed.] Risk evaluation and biological reference points for fisheries management. Can. Spec. Publ. Fish. Aquat. Sci. 120.

This study focuses on a fishery managed by controlling effort in terms of the number of licences issued and the length of the fishing season to ensure a target level of escapement of an annual squid stock. Methodology for assessing the probability of having to shorten the fishing season in order to achieve the conservation goal is presented. The fishery for *Illex argentinus* around the Falkland Islands is used as an example for the methods. The probabilities of closure for different levels of target escapement and assumptions about the parameters describing the statistical distribution of recruitment are quantified. The implications of results for management decisions are discussed.

L'étude porte sur une pêche contrôlée grâce à la limitation du nombre de licences et de la durée de la saison de pêche en vue d'arriver à un niveau cible d'échappée d'une population annuelle de calmars. On fait état de la méthodologie utilisée pour évaluer la nécessité de raccourcir la saison de pêche à la lumière de l'objectif de conservation établi. La pêche de *Illex argentinus* aux environs des îles Falkland servira d'exemple des méthodes mises en oeuvre. L'étude quantifie les probabilités de fermeture selon différents niveaux cibles d'échappée et les diverses suppositions quant aux paramètres de distribution statistiques du recrutement. On y discute ensuite de l'importance des résultats obtenus pour la gestion.

The short fin squid, *Illex argentinus*, is exploited in a major fishery in the waters along the Patagonian shelf and around the Falkland Islands. The fishery started in the late 1970's and a rapid increase in effort between 1984 and 1986 led to the declaration of a conservation zone around the islands and the imposition of regulations in 1987. The Renewable Resources Assessment Group at Imperial College have advised the government on the development of the management system and have assessed the squid stocks on a regular basis since 1987.

The management problems for squid stocks and procedures for conservation and development of a sustainable fishery are discussed in Beddington et al. (1990) and details of assessment methods appropriate for squid stocks are discussed in Rosenberg et al. (1990). In this paper, we consider some additional problems of management. The probability of having to close the fishery early to satisfy conservation goals, is quantified. The implications of closure are considered and the sensitivity of results to assumptions and input parameters are investigated.

Distribution and Biology

The short fin squid, *Illex argentinus,* is distributed along the Patagonian shelf (from 42°S latitude) to the waters around the Falkland Islands (at 49–54°S). Harvesting occurs on three main fishing grounds: at 42°S, around 45–46°S and within 150 nm of the islands (Fig. 1).

The stock structure is not well known. All current knowledge from age, growth and genetic studies, however, indicate that a single stock is exploited from the two southernmost grounds (Thorpe et al. 1986; Rodhouse and Hatfield 1990). Current knowledge also indicates that the basic pattern of migration is as follows. In January small squid (<300g in body weight) are found in concentrations at 45–46°S in international waters. As the season progresses, the squid migrate south toward the Falkland Islands. During this period they also grow at a rapid rate and by the time they reach the waters around the islands, they are around 500g in body weight. Very dense concentrations are found in these waters during April and May while less dense concentrations are still found in international waters. Toward the end of the season (end of May, to early June) the squid migrate northward, out of the waters around the islands, to spawn and die in July or August (Fig. 2).

The fact that *Illex* is an annual species is clearly crucial for the development of management procedures for this species.

The Fishery

The squid stock is exploited by distant water fleets from 12 countries. Most of the vessels are jiggers but trawlers are also involved. The fishery operates on a single cohort each year and there is a single fishing season between January and June.

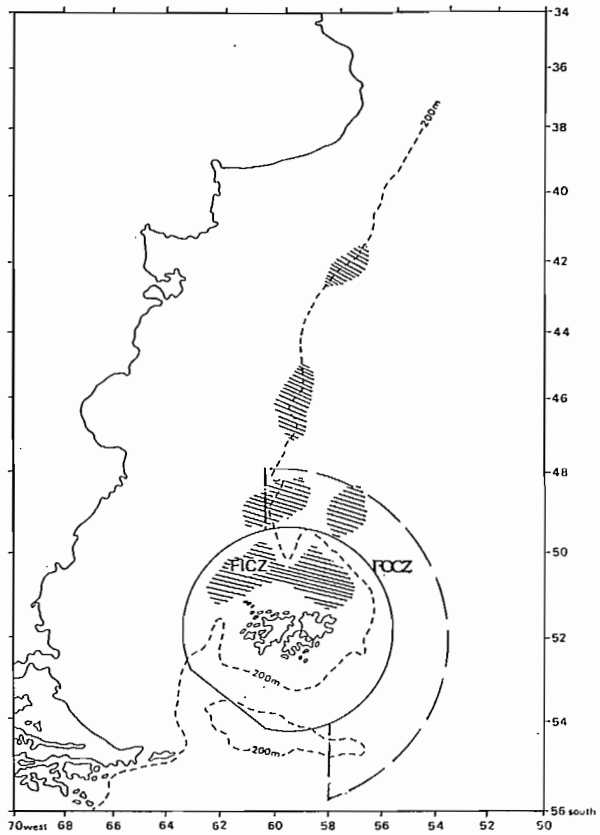


Fig. 1. A schematic map of the south-west Atlantic indicating the approximate locations of the Falkland Interim Conservation Zone (FICZ) and the Falkland Outer Conservation Zone (FOCZ). The shaded areas indicate the major fishing grounds and the 200 metre depth contour is indicated by the dashed line.

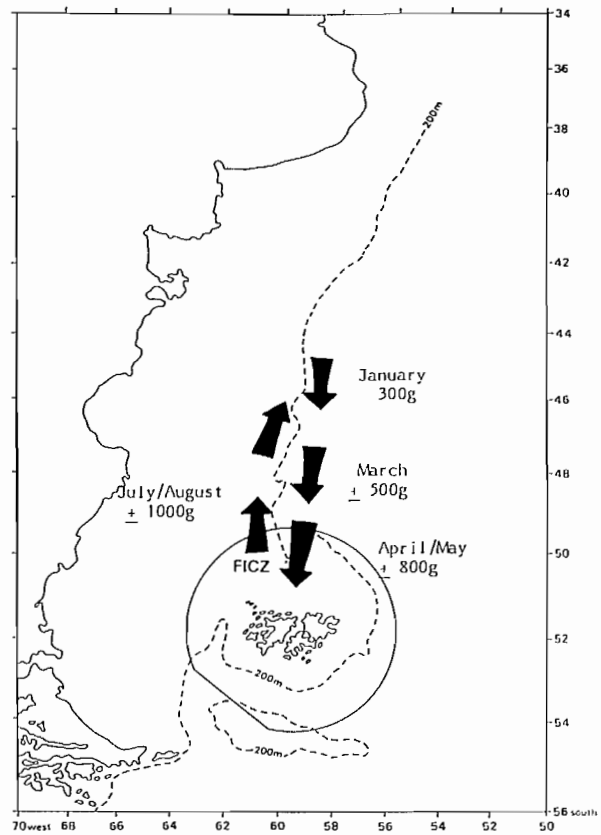


Fig. 2. A schematic map of the south-west Atlantic with arrows suggesting the migration pattern of *Illex* as currently understood. The approximate average weight of individuals (in grams) at different locations and times through the season are shown.

Table 1. Catches of the squid *Illex argentinus* taken in the FICZ during the 1989 to 1991 fishing seasons and the number of vessels involved in the FICZ and under the voluntary restraint agreement (VRA - see text for details).

	YEAR		
	1989	1990	1991
Catch in FICZ (mt)	221,000	98,000	169,000
Vessels in FICZ/VRA	166[1]	192	195
Season length (months)	3	2.3[2]	3

[1] The number of vessels in 1989 refers only to the number of licensed vessels. In 1990 and 1991, the number of vessels include vessels fishing outside the FICZ, but under the voluntary restraint agreement (VRA).
[2] In 1990 a closure came into effect before the official end of the fishing season.

Since the declaration, in 1987, of the conservation zone (Falkland Islands interim conservation zone, FICZ) which extends for 150 nm from the centrepoint of the islands, some vessels fish under the management and regulation system set up by the Falkland Island Government. Other vessels fish exclusively in international waters. Fishing within the FICZ is regulated by licensing vessels to fish during a restricted season, from 1 March to 31 May. In most years the catch rates are higher within the FICZ than in international waters, particularly during the peak of the fishing season in April and May. Basic details of the fishery are summarised in Table 1.

It is clear that, because of the migratory nature of the stock, it is only partially regulated by management in the FICZ. Fishing by unlicensed vessels in international waters may partially undermine management gains within the FICZ. As highlighted in Beddington et al. (1990), however, the bulk of the stock occurs within the FICZ during much of the season. Since 1989 attempts have been made to create incentives for major fishing companies to voluntarily reduce their fishing effort outside the FICZ in exchange for preference for licences in the regulated zone. The effect of these arrangements, referred to as voluntary restraint agreements (VRA's), is that the level of effort is not set simply by the management authorities but is a compromise based on the commercial negotiation.

In December 1990, the Falkland Islands Outer Conservation Zone (FOCZ) was declared. Fishing was banned in this zone during the 1991 fishing season with an aim to further improve escapement of the *Illex* stock (Fig. 1).

A Basis for Management

Management within the FICZ is based on two main characteristics of the stock. Firstly, a single cohort is exploited each year, with the population size of that cohort unknown until after fishing has commenced. Secondly, the hypothesis about the nature of the stock recruitment relationship is such that very low levels of spawners are unlikely to produce high levels of recruitment. Beddington et al. (1990) present a graphical representation of the likely ranges of recruitment (in January) as a function of the number of spawners in the previous year. A similar graphical representation is shown in Fig. 3 and it suggests that, for levels of spawning stock above some threshold value, recruitment in the following year is basically a random variable. At levels of spawning stock below the threshold level, the probability of low recruitment in the following year is, however, relatively high. The management objective is therefore to maintain 'safe' levels of spawning stock biomass (above some threshold) and to try and avoid high probabilities of very low recruitment.

Two approaches are considered to achieve this objective. Effort, which is directly related to the number of licences allocated in each fishing season, is limited to ensure some target level of proportional escapement. Proportional escapement is defined as the ratio between the number of spawners that survive to the end of the fishing season under a given level of fishing effort and the number of spawners that would survive were there no fishing. The second approach is to consider absolute escapement, in other words, the number of animals that survive to spawn. This is necessary since, in a situation where recruitment is very low, even relatively high proportional escapement may not allow a sufficiently large number of individuals to survive and spawn.

During the fishing season, the stock status is monitored on a weekly basis and if indications are that the stock size is very low (compared to previous years), or that proportional escapement may be very low, there may be a need to close the FICZ and prohibit fishing before the official end of the season. In such a situation, there may also be a need to refund a proportion of the licence fee, depending on the length of the closure.

The management strategy is therefore to ensure some target level of escapement and the basic management tool is the control of effort. There are two types of effort control, each relating to a different aspect of the management process:

1) Control Mechanism: limiting the number of licences.

   Management Process: determining allowable levels of effort (number of licences) prior to the start of the fishing season (the licensing phase).

2) Control Mechanism: limiting the length of the fishing season.

   Management Process: monitoring the stock during the fishing season (the stock assessment phase) and determining whether the fishing season needs to be shortened.

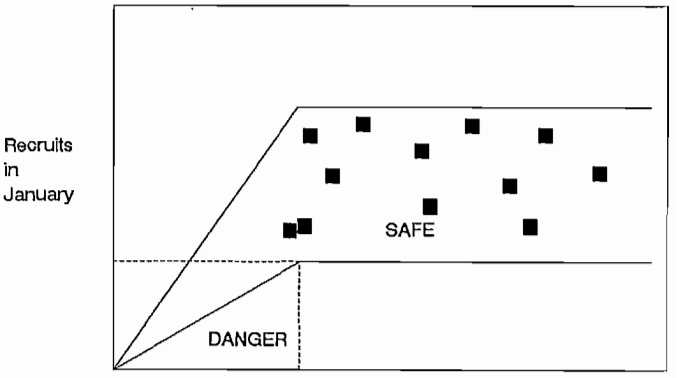


Fig. 3. Schematic relationship between stock and recruitment for squid to illustrate the basis for management. (After Beddington et al. 1990).

These two aspects are described in detail in Beddington et al. (1990) (licensing phase) and Rosenberg et al. (1990) (stock assessment phase). Here we consider the licensing phase and, in particular, the issue of closure in relation to the level of effort that is licensed.

It is important to recall that the size of the incoming cohort is unknown at the time of licensing and it is also highly variable from year to year. At the licensing stage, it is therefore only possible to aim at a target level of proportional escapement. The number of licences that are allocated in accordance with the target escapement is dependent on information on effort patterns (for example average hours fished per vessel per week) and vessel efficiency (estimates of catchability coefficients) from previous seasons. It is at the assessment stage, during the fishing season, that the need for a closure can be detected and corrective action taken if required.

The need for a closure can be caused by one, or both of the following factors:

1. a very low level of recruitment.

2. changes in efficiency or operational practices of vessels.

In this study we are only considering the need for a closure due to low levels of recruitment. The question that is asked at the licensing stage is: what is the probability that a closure would be required for a given target of proportional escapement?

Methods

As outlined above, the main conservation goal is to keep spawning stock levels above some threshold level to avoid high probabilities of very low recruitment in the following year. An early closure of the fishery may sometimes be required to attain this goal.

The probability of closure in year j is therefore defined as the probability that the spawning stock level at the end of

the season, S_t, will fall below the threshold level, S^*, in other words:

$$(1) \qquad p_t = P\left[\left(S_t = R_t e^{-M-F}\right) < S^*\right]$$

where R_t is the population size at the start of the fishing season (recruitment in year t), M is total natural mortality and F is total fishing mortality. Fishing mortality is assumed to be related to total effort by:

$$(2) \qquad F = qE$$

where q is the catchability coefficient. In reality, a catchability coefficient would be estimated for each fleet or each vessel and total fishing mortality would be calculated as the products of effort and catchability summed over all fleets or all vessels. The total fishing mortality is determined by choosing a certain target level of proportional escapement, K, which is given by:

$$(3) \qquad K = e^{-F}$$

In order to calculate the probability in (1), it is necessary to make some assumptions about recruitment and we assume that:

1. there is a threshold level of spawning stock such that recruitment (in year $t + 1$) is random for a spawning stock level (in year t) above the threshold;
2. random recruitment is log-normally distributed.

In more formal notation, therefore:

$$(4) \qquad \ln(R_{t+1}) \sim \mathrm{N}(\mu, \sigma^2), \text{ for } S_t > S^*$$

where S_t is the spawning stock at the end of the fishing season in year t, S^* is the threshold spawning stock, R_{t+1} is the initial population size (or recruitment) in year $t + 1$. The parameters of the normal distribution are μ, the mean, and σ^2, the variance. We also assume, for calculations presented here, that management succeeds in keeping spawning stock levels above the threshold.

The main parameters required for calculation of the probability of closure are the mean and variance of log-recruitment and the threshold level, S^*. As indicated above, the fishery in the FICZ has been monitored on an annual basis since 1987 and, although there have been changes in the assessment methods and improvements in data quality and quantity, we basically have 5 years worth of recruitment estimates and spawning stock estimates. There is also information in Csirke (1987) although these data are not directly comparable since they relate to a larger area and therefore include the stock at around 42°S. Only the data set from the FICZ was therefore used to estimate a mean and variance of log-recruitment. Results from a joint 95% confidence interval for the mean and variance suggest that a range of around 0.05 to 1.00 should be considered for the variance. In calculations presented here, values of 0.1, 0.5 and 1.0 are used.

Obtaining an estimates of a threshold spawning stock level is of course far more problematic. The approach that has been taken so far is to consider the lowest estimate of spawning stock that has led to viable recruitment in the following year. By viable recruitment we mean a level that is not lower than any previous recruitment level. In addition to the estimate obtained in this manner, we also consider two other levels, half the value of the estimate and double the value of the estimate, to investigate the sensitivity of results to assumptions about this parameter. It is useful to express the threshold as a proportion of the spawning stock that would result from mean recruitment with no fishing. The three threshold levels correspond to 'threshold' ratios of 7%, 14% and 27% (at a variance of 0.5).

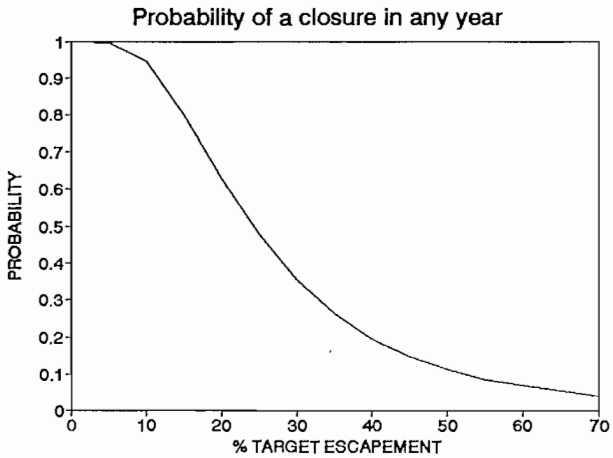


Fig. 4. The probability of a closure of the fishery for various levels of target escapement. (The variance of log-recruitment is 0.5 and the 'threshold' ratio is 14%).

Results

Figure 4 illustrates the probability of closure as a function of the level of target escapement when the variance is in the middle of the range (0.5). It is clear that the relationship is not linear and, as expected, low escapement has a high associated probability of closure whereas high escapement has a low associated probability of closure.

It is also possible to consider probabilities of at least one closure over a period of, say, 2 or 5 years. The probability of closure in any one year is assumed to be independent of events in previous years. This is valid under the assumption that management succeeds in detecting and effectively implementing a closure, thus maintaining spawning stock levels above the threshold. In this case the probability of at least one closure over a period of T years is easily calculated from the binomial distribution.

Figure 5 illustrates these calculations for periods of 1, 2 and 5 years. It is worth noting that even if the probability of closure in any one year is not particularly high (e.g., around 0.35 for 20% escapement), the probability of at least one closure over a period of 5 years, is close to 1.00.

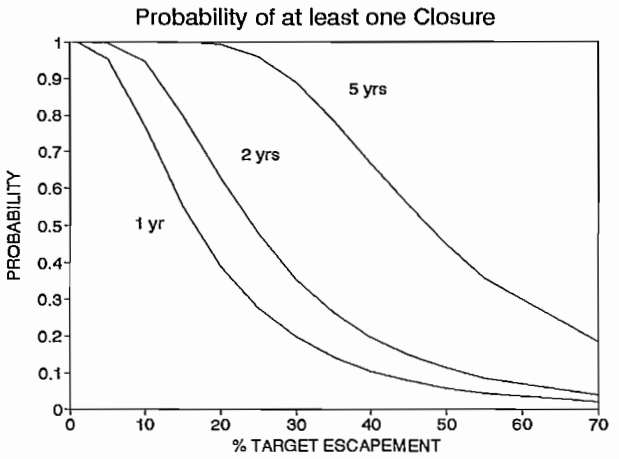


Fig. 5. Probabilities of at least one closure being required over periods of 2 and 5 years shown with the probability of closure in any one year. Parameters are as in Figure 4.

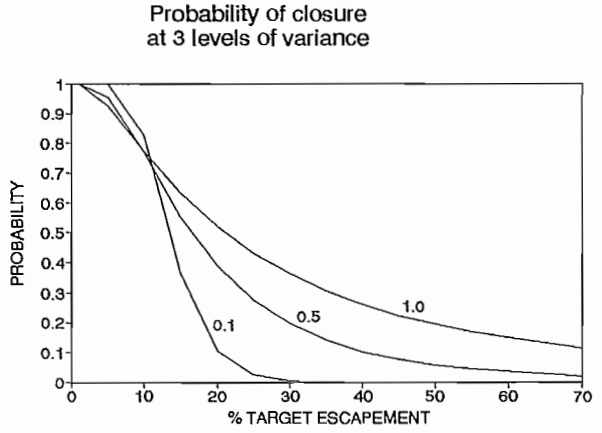


Fig. 6. The probability of a closure at three levels of variance, as a function of the level of target escapement. The threshold level is the same for all curves.

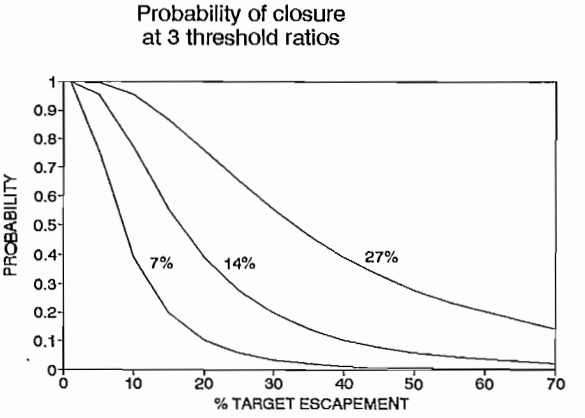


Fig. 7. The probability of a closure for three 'threshold' ratios. The variance of log-recruitment is 0.5 in all cases.

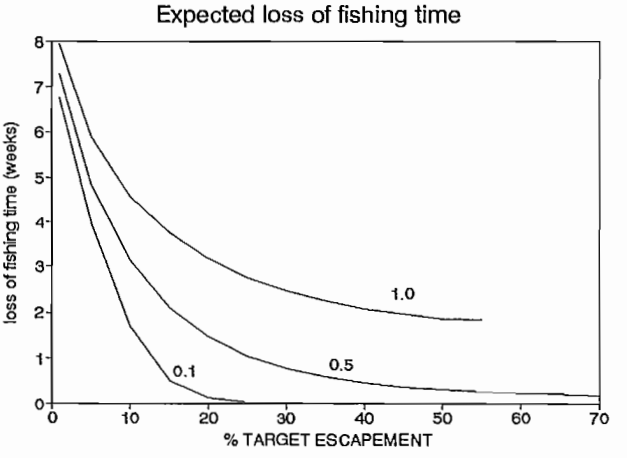


Fig. 8. The expected duration of a closure at various levels of escapement and three values of the variance of log-recruitment. The curve for a variance of 1.0 has been truncated, because of model break-down. The expected duration of a closure becomes longer than 12 weeks when escapement is high. This is clearly unrealistic since the total duration of the fishing season is assumed to be 12 weeks.

One of the key issues underlying these figures is the level of uncertainty in the input parameters. It is therefore important to investigate the sensitivity of results to assumptions about the input parameters. Figure 6 illustrates the probability of closure in any one year for three levels of variance in recruitment. At high levels of variance in recruitment the probability curve is relatively flat, particularly at high levels of escapement. This implies that, for example, a 10% change in escapement would not make a large difference to the probability of closure. In contrast, a small variance in recruitment implies that over a certain range of escapement levels, a relatively small change in escapement can make quite a large difference to the probability of closure.

The shapes of these curves are of course also affected by the threshold level and, in particular the threshold level relative to the mean. Figure 7 shows probability graphs for a variance of 0.5 and three threshold levels. It is common sense that the closer the threshold is to the mean, the higher the probability of closure at a fixed proportional escapement.

Figures 6 and 7 also illustrate that if the variance or threshold level has been underestimated, the 'true' probability of a closure would be underestimated and vice versa. With respect to the threshold level, this is of course only really of academic interest since it is highly unlikely that one would ever be able to estimate the 'true' threshold level. With respect to the variance the figures suggest that, in order to be conservative, it is preferable to over- rather than underestimate the variance.

The above analyses have only focused on the probability of a closure being required or not. The expected duration of a closure (given that a closure is required) is also affected by the input-parameters. Figure 8 illustrates the expected loss of fishing time, given by the product of the probability of a closure and the expected duration of a closure (given that a closure is necessary) as a function of target escapement. The expected loss of fishing time increases as the target escapement decreases. At a given target escapement, the expected

loss in fishing time is longer for a high variance of recruitment than for a low variance.

Discussion

Uncertainties of Input Parameters

There is a large degree of uncertainty associated with the input parameters used to generate these probability levels, but it is possible to identify likely ranges of values. It is also possible to narrow the range of values by focusing on the more conservative scenarios. It is furthermore not the exact value of the probability of closure that is of interest but the relationship between the probability and the level of proportional escapement.

The simple calculations presented above, illustrate two important points which, even if obvious to the fisheries scientist, may not be to the manager. The two points are:

1. there is an inverse relationship between the probability of closure and escapement — in other words, the lower the level of escapement, the higher the probability of closure and vice versa;

2. this relationship is not linear and there are parameter values for which a small change in escapement would imply a relatively large change in probability of closure. Conversely, there are other parameter values for which even a large change in escapement would only imply a small change in the probability of closure.

Although the basic pictures are similar for different parameter values, management strategies may differ greatly depending on where in parameter space the system really is. For example, if the system is in a region where a very small increase in the level of escapement would imply a large reduction in the probability of a closure, it would be worth trying to achieve an increase in target escapement. On the other hand, if the system is in a region where even a large increase in escapement would hardly affect the probability of closure, there may be no need to try and achieve an increase in target escapement.

The two main parameters that affect the probability curves are the variance of recruitment and the relative magnitudes of the mean and the threshold of recruitment. An estimate of the degree of variability in recruitment should improve with more information but this does assume that there are no trends in the dynamics. The same is true for mean recruitment. It is unlikely that one would ever have sufficient information to assess the level of 'threshold' spawning stock without risking a collapse in the population. In the case of an annual species, where there is no 'buffering' effect of pre-recruits and multiple age classes, the consequences of allowing spawning stock levels to drop to very low levels can be very serious indeed. It is therefore common sense that very low levels of spawning stock should be avoided. Defining what is 'very low' is of course problematic, particularly in the absence of any pre-exploitation data on stock and recruitment levels.

In this situation a 'common sense' approach is a sensible option. As indicated in the Methods section, the lowest level of spawning stock that has led to viable recruitment (that has not been the lowest observed) has been used as a rule of thumb in the Falkland Islands fishery. Given only 5 years worth of data, there is no way of knowing whether a level set in this way is too high or too low. In practice one would of course closely monitor the series of spawning and recruitment levels for any evidence of a pattern of decline in recruitment from year to year, as well as any other indications or evidence suggesting that absolute escapement may be too low.

Evaluation of Risks Associated With Licensing Decisions

The types of calculations presented here have been used to aid the Falkland Island government in making decisions about levels of effort to licence. A large number of licences would imply a low level of escapement and therefore a high probability of having to close the fishery early. In the case of a closure there may also be a need to refund a proportion of the licence fee, depending on the duration of the closure. The expected cost of a closure, in terms of loss of fishing time, is higher at high levels of effort than at lower levels of effort and this clearly has implications for the expected magnitude of any refund (see Fig. 8).

Closure of the fishery is therefore not without cost and should really be seen as a fail-safe to ensure that conservation targets are met. The above calculations can easily be extended to quantify the expected financial cost of a closure and the trade-off between the expected income and cost at different levels of target escapement.

In addition to direct costs associated with high probabilities of closure (which are associated with low levels of escapement), the choice of escapement level also has implications for assessment and the ability to detect the need for a closure. It is always possible to deem a closure necessary when it is in fact not. More serious, however, particularly in a fishery on an annual species, is the possibility of not detecting the need for a closure at all or in time to take the necessary action. The two most important aspects are the following. Firstly, the need for a closure must be detected at an earlier stage and action must be taken more promptly when effort levels are high than when they are low. This is because the rate of decline in population size, caused by the fishery is faster at high levels of effort. Secondly, the chances that insufficient data (for assessment purposes) would have been collected by the time a closure is required is much higher at high levels of effort than at low levels.

Although we are not concerned with this aspect of 'risk' in this paper, it is a very important factor which needs to be considered when management decisions are made.

Further Aspects of the Management System

A management target of 40% proportional escapement for the fishery in Falkland Islands waters has been advised in the past (Beddington et al. 1990), following conventions in

other major squid fisheries. If the entire stock were found in Falkland Islands waters, the situation would have been relatively simple. Managers could have been advised to aim at a target level of escapement of 40% and licence accordingly. Results presented here suggest that the associated probability of closure is relatively low.

Unfortunately, the situation is far more complicated in reality, mainly because of the migratory nature of the stock. It is still possible to aim at a target level of proportional escapement in the FICZ, but the number of licences that are allocated affect the level of effort outside the FICZ and fishing in the whole region affect overall escapement and hence the probability of closure. If, for example, far fewer licences are issued, many vessels will still fish in international waters and for as long a period as they wish. This may, under extreme circumstances, lead to an overall increase in effort rather than a decrease.

The implications of different levels of escapement for the entire stock and the associated probabilities of closure have therefore become far more important. Since 1989 the approach to management has been to consider the stock and fishing effort in the whole area, south of 45°S. The fishery over this whole region can of course only be partly controlled, but account has to be taken of overall effort in the region and overall escapement of the stock. In this context one of the most serious problems is the uncertainty in the actual level of effort outside the FICZ. This uncertainty can be incorporated into the management system by using 'worst' and 'best' case scenarios in addition to an 'average' scenario with respect to the level of effort outside the FICZ.

The annual nature of squid stocks means that most advice is revised on an annual basis. It is, however, possible to examine the medium term implications of management decisions by considering probabilities of closure over periods of, say, 5 years. This is of particular relevance in situations where domestic vessels become involved in the fishery.

Advice to the Falkland Islands Government has always been to aim for a relatively high level of target escapement. At the same time, the increased probability of having to shorten the fishing season, for any decrease in target escapement has been emphasised. In addition, the expected period of closure increases when the target escapement decreases. These general 'messages' can easily be translated into practical terms, such as:

- at 40% target escapement, a closure would, on average, be required once every five years (for example, see Fig. 4) or,
- at 20% target escapement, the expected cost of a closure would be 'so many' thousands (or millions) of pounds (sterling)

It is essential that the fisheries managers understand what the implications of a decision may be and if these can be expressed in simple terms it is more likely that the scientists' advice would be taken seriously.

There are, of course, other sources of 'risk' involved in the management process. In the above calculations we have assumed that a closure, if required, is detected and action is taken in time. We have alluded to the probabilities of errors occurring at the assessment stage, either by not detecting a closure when it is required or by detecting the need for a closure when it is, in fact, not necessary. We have also indicated that a closure may be required because of changes in operational patterns or the efficiencies of vessels. Although these factors are more difficult to quantify, they should, ideally, be incorporated in calculations of the probability of closure.

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Risks of Over- and Under-Fishing New Resources

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New fishery resources are invariably developed before basic assessment information on the structure, size and productivity of the stocks is available. While this lack of information may be remedied over time, exploitation and management will proceed on the basis of the information which is available at any given point in time. In some cases the rate of fishing down may exceed the rate of learning, resulting in overfishing of the resource. In others, a more cautious approach to management may result in underexploitation through a failure to "test" the productive potential of the stock. Both outcomes represent the risks of managing an uncertain resource. This paper explores some of the issues involved in this form of "management under uncertainty", using the development and exploitation of orange roughy stocks (*Hoplostethus atlanticus*) as an example. Using Monte Carlo simulation, management strategies are assessed against the risks of over- and under-exploitation during the development phase of a fishery, illustrating the importance of the feedback between rate of development and learning about stock dynamics.

Invariablement, on commence à exploiter de nouvelles ressources halieutiques avant de disposer de l'information de base sur la structure, la taille et la productivité des stocks. S'il est possible de remédier avec le temps à ce manque d'information, l'exploitation et la gestion se poursuivent à partir de l'information disponible en un point donné du temps. Dans certains cas, le taux d'exploitation peut dépasser le rythme d'accumulation des connaissances, ce qui cause une surpêche. Dans d'autres cas, une approche plus prudente de la gestion peut causer une sous-exploitation du fait qu'on n'a pas testé le potentiel reproducteur d'un stock. Ces deux résultats sont les risques que posent la gestion d'une ressource mal connue. Notre étude explore certains des problèmes posés par cette forme de «gestion dans l'incertitude», à partir de l'exemple de l'exploitation des stocks d'hoplostète orange (*Hoplostethus atlanticus*). À l'aide d'une simulation de Monte-Carlo, nous évaluons les stratégies de gestion par rapport aux risques de surexploitation et de sous-exploitation pendant la phase de développement d'une pêche, en illustrant l'importance de la rétroaction entre le rythme de mise en valeur et l'accumulation de connaissances sur la dynamique d'un stock.

The development of a new fishery, or the discovery of a new resource within an existing fishery, is a striking example of a general class of problems of "management under uncertainty". In such cases, neither industry nor managers may know the initial size of the resource, even to an order of magnitude. If the species is previously unexploited, basic biological information such as growth and longevity, which provide some indication of productivity, may also be unavailable. If initial exploitation demonstrates the economic viability of fishing, there will be a natural tendency for effort to increase. This effort will be drawn in at a time when catch rates, reflecting stock abundances, are high, and may result in overcapitalization in the new fishery as either the stock becomes overexploited or initial catch rates reduce to some sustainable level. Assuming the fishery is managed more or less from its inception, managers are faced with making a choice between hindering justifiable development of the fishery, and preventing overcapitalization and overexploitation.

The concepts and techniques of adaptive management have been developed as a framework for addressing the general problem of management under uncertainty (Walters and Hilborn 1976; Smith and Walters 1981; Walters 1986). The adaptive management approach recognizes that for most fisheries, information on the size and productivity of the resource comes from monitoring its response to exploitation, that is, we learn about the productive potential of a resource by exploiting it. In some cases management actions should be chosen which deliberately probe the response of the resource over a range of stock sizes, even at the expense of short term gains in yield. Hilborn and Sibert (1988) have examined the particular case of developing fisheries within the adaptive management framework.

This paper addresses the problem of managing a developing fishery where the main source of uncertainty is the size of the resource, and where only relative indices of stock size are available for stock assessment purposes. Catch rates in the commercial fishery may provide such indices in some cases, while for others specially commissioned resource surveys may be required. To provide a focus for the formulation of the problem, the development of the orange roughy fisheries in New Zealand and Australia is used as an example.

Orange Roughy

Orange roughy is a relatively recently exploited species of cosmopolitan distribution which is fished at depths of 700 to 1500 metres. The major fisheries so far are in New Zealand and Australia, but exploratory fishing has begun in the North

Atlantic. The species forms dense aggregations, often around irregularities on the ocean floor along the mid-continental slope. Apart from being one of the deepest commercially exploited fishes, it is also one of the longest lived. Age at maturity is thought to be about 20 years (Mace et al. 1990), and a maximum age in excess of 100 years is now considered likely (Fenton et al. 1991). Basic life-history parameters are given in Francis (1992).

Several stocks of orange roughy have been fished by New Zealand fleets since about 1979, with a peak in catches in 1989 of 54,000 tonnes. At least two of the larger stocks have been reduced to low levels within about 10 years (Robertson 1991), and Francis (1992) presents a risk analysis for a range of catch reduction strategies for the heavily exploited Chatham Rise stock to the east of New Zealand. However, new stocks have been discovered as recently as 1990 to the south of New Zealand (Francis et al. 1992).

The Australian orange roughy fishery did not develop until 1986, and catches peaked in 1990 at about 42,000 tonnes (Smith 1991). The main areas of fishing have been to the east and south of Tasmania, and at least one major stock may have been reduced to below half its initial size within four years (Smith 1992).

Campbell et al. (1993) have developed a bioeconomic model for orange roughy which they apply to the eastern Tasmanian stock and use to evaluate the economic benefits of management, and the economic costs of a conservative ($F_{0.1}$) strategy. Provided the discount rate is low, there are clear economic benefits to managing orange roughy stocks.

Model Structure and Evaluation

The aim is to evaluate alternative management strategies for developing fisheries. The approach is to use Monte Carlo simulation to evaluate the performance of different "decision rules" in managing a simulated stock whose dynamics are described by an "operational" model. Each decision rule specifies a catch to be taken in the current year, based on an assessment of the size of the stock, and on a prescribed "fishdown" rate. The term "fishdown" is used to describe the reduction in stock size from virgin levels brought about by fishing during the initial development of the fishery. The fishdown rate is the exploitation rate during this period.

Operational model

The model used to simulate resource dynamics is modified from the Deriso-Schnute type (Schnute, 1987), specifically

$$B_{t+1} = SB_t - C_t + R_t \tag{1}$$

where t = time, B = stock size, C = catch, R = recruitment and S = survival (actually "biomass survival", a function of natural mortality and growth). B, C and R are in units of biomass (weight).

As noted above, age at recruitment for orange roughy is on the order of 20 years, so no effects of fishing on recruitment would be expected for the first 20 years of the fishery. Virgin levels of recruitment are given by:

$$R_0 = (1 - S)B_0 \tag{2}$$

where B_0 is virgin (unexploited) biomass.

For orange roughy, natural mortality M is thought to be about 0.05 (Mace et al. 1990). Growth is very slow after age at recruitment, so its effect in the biomass survival term S is ignored in this analysis. S is therefore set to 0.95.

Observation process

It is assumed that a relative index of stock size can be obtained at any time step. For an aggregating species like orange roughy, catch rates are unlikely to reflect stock size but an annual trawl survey might provide such an index. This index is subject to observational error, so the observation equation is given by

$$Y_t = qB_t + \varepsilon_t \tag{3}$$

where Y_t is the index at time t, q is a (catchability) parameter to be estimated, and $\varepsilon_t \sim \mathrm{N}(0, \sigma_t^2)$ is the error term. It is assumed that σ_t^2 is known or measurable. In these simulations, σ_t^2 is set such that the coefficient of variation of Y for a given B is 0.2. The catch C_t is assumed to be observable without error.

Parameter estimates

It is assumed that the productivity parameter S is known. Estimates of the remaining two parameters B_0 and q can be obtained from a time series of catches and indices, using a maximum likelihood estimator similar to that of Francis (1992). This estimator requires at least three observations of Y and a complete catch history, so the initial estimate can not be revised until at least three indices have been obtained. The derivation of the maximum likelihood estimates for B_0 and q is given in the Appendix.

For each simulation, an initial estimate $\widehat{B}_0$ is drawn from a distribution about the "true" B_0 for the simulation, with a coefficient of variation of 0.5. However $\widehat{B}_0$ is constrained to be not less than 10% of the true value. Given $\widehat{B}_0$ and the catch history, an estimate of current stock size $\widehat{B}_{\mathrm{cur}}$ can also be derived using Equations 1 and 2. The true B_0 is arbitrarily set to 100 units and the true q to 1.

Decision rule

To simulate management of the modelled stock, an initial "fishdown" exploitation rate E_F and a target stock size B_τ are specified. The initial fishdown rate is expressed as a fixed percentage of the estimated initial stock size $\widehat{B}_0$ such that

$$C_{\mathrm{cur}} = E_F\widehat{B}_0/100 \tag{4}$$

where C_{cur} is the catch chosen for the current time period. However since $\widehat{B}_0$ will change over time, so will the catch during the fishdown phase.

The target stock size is also calculated as a percentage p of the estimated B_0 such that

$$B_\tau = p\widehat{B}_0/100$$

Once this target is reached, the catch is chosen to stabilize the stock at the target size, i.e.,

$$C_{cur} = S\widehat{B}_{cur} - B_\tau + \widehat{R}_0 \quad (5)$$

where $\widehat{R}_0$ is estimated virgin recruitment, calculated by substituting $\widehat{B}_0$ in equation 2.

During the fishdown phase, a constraint is placed on the change in catch from one time period to the next. The catch at any time must be less than 1.5 times the previous maximum catch, and greater than 0.5 times the previous minimum, unless the coefficient of variation on the estimate of B_0 is less than 0.1.

Evaluation criteria

The performance of each management strategy or decision rule (as specified by an $\{E_F, B_\tau\}$ combination) is based on criteria evaluated over 100 simulations. Calculations for each decision rule use the same set of random numbers to calculate initial $\widehat{B}_0$ values and observed indices. The time frame chosen for evaluation is 15 years, as this should equal or exceed the development phase for most new fisheries.

Three general sets of criteria are used for evaluation. Total catch over the 15 year development period is used as an indicator of the short term economic performance of each strategy. Biological risk is measured by the frequency with which the stock size falls below 20% of B_0. Finally, several measures of estimation performance (i.e., how well the true B_0 is estimated) are used as a proxy for longer term economic performance.

The specific evaluation criteria, averaged over 100 simulations, are:

1. total catch over the 15 years;
2. the percentage of simulations in which biomass fell below 20% of B_0;
3. biomass after 15 years, as a percent of B_0;
4. the final (year 15) biomass estimate $\widehat{B}_0$;
5. the standard error in the final $\widehat{B}_0$ estimates;
6. the percentage of simulations which failed to estimate B_0.

Failed estimation in criterion 6 is invoked whenever the final $\widehat{B}_0$ estimate is negative. Any such simulations were disregarded in calculating criteria 4 and 5, but all simulations were used for criteria 1 to 3. Negative estimates of B_0 can arise when there is an overall increasing trend in biomass indices over time.

Table 1. Performance indicators in relation to fishdown rate (catch as a percentage of estimated B_0). Target $B = 30\%\ B_0$. For an explanation of the performance indicators, see text on evaluation criteria.

	FISHDOWN RATE E_F		
INDICATOR	5%	10%	15%
Catch	38	66	70
$\%B < 20\%B_0$	0	12	19
B_{15}/B_0	59	34	33
Final $\widehat{B}_0$	96	97	97
SE Final $\widehat{B}_0$	31	7	6
% failed estimation	18	4	2

Results

Table 1 shows the effects of the initial fishdown rate E_F. The target stock size in each case is 30% of B_0. A catch of 3.5% of B_0 would stabilize the stock at this target level in the model, given continuing (deterministic) recruitment at virgin levels. Thus all the fishdown rates in Table 1 (5%, 10% and 15% of B_0) exceed the "sustainable" rate.

The catch increases with the fishdown rate, as would be expected given the relatively short time horizon of 15 years. The biological risk to the stock also increases as the fishdown rate increases. Stock sizes below 20% of virgin biomass levels occurred in one in five simulations at the 15% fishdown rate. However stocks were never reduced to low levels at the lowest fishdown rate of 5%, and the mean stock size after 15 years was still only about 60% of virgin levels at this rate of fishdown.

The last three rows of Table 1 provide three measures of the "estimation performance" of each strategy, that is, the extent to which the true B_0 was correctly identified given the range of initial estimates. Final estimates averaged over simulations were close to the true value for all strategies, but individual estimates were highly variable for the 5% fishdown rate. A comparison of the distribution of final B_0 estimates for the 5% and 10% rates is shown in Fig. 1. Parameter estimation also tended to fail much more commonly for the lowest fishdown rate. In summary, estimation performance was clearly worst for the 5% strategy, and much improved for both the higher exploitation rate strategies.

Table 2 shows the effect of choosing a more conservative (i.e., higher) target stock size. As one would expect, this reduces both the short term catch and the probability of low stock sizes. It also has a minor adverse effect on estimation performance.

The effects of the management strategy on the performance in estimating B_0 (and q) can be explained as follows. The ability to turn the relative indices of biomass (Y) into an absolute estimate of biomass B_0 relies on using the catch history (and the known dynamics) to put an absolute scale on the change in relative index from one sample period to the

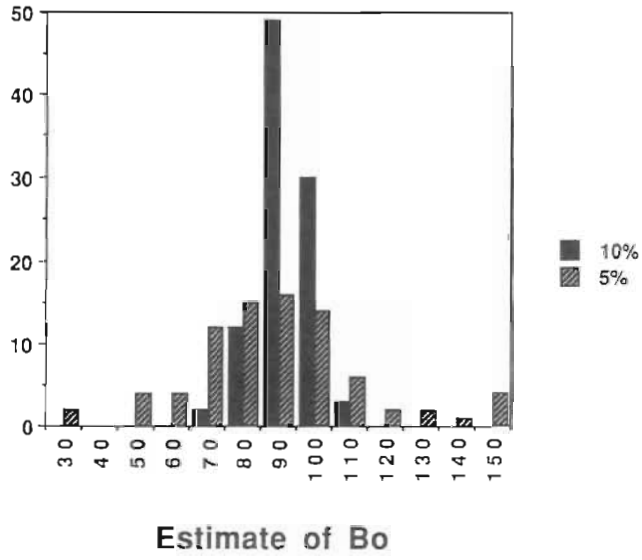


Fig. 1. Frequency distribution of final estimates of B_0 for fishdown rates of 5 and 10%. The true B_0 is 100. Negative estimates are not included. The 150 category is 150 and above.

Table 2. Performance indicators in relation to target B as a percent of B_0. Fishdown rate = 10%. For an explanation of the performance indicators, see text on evaluation criteria.

	(TARGET B)/B_0	
INDICATOR	30%	50%
Catch	66	48
$\%B < 20\%B_0$	12	1
B_{15}/B_0	34	55
Final $\widehat{B}_0$	97	92
SE Final $\widehat{B}_0$	7	14
% failed estimation	4	4

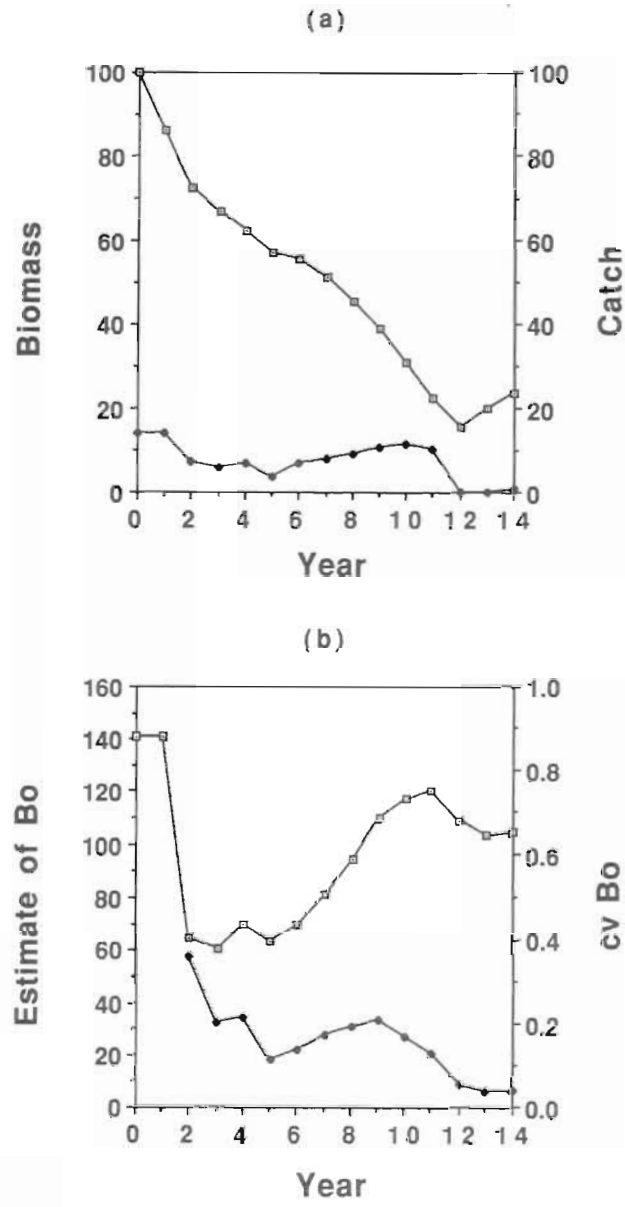


Fig. 2. Detailed time trajectories for a simulation using the 10% fishdown rate and a target biomass of 30% B_0. (a) Biomass (open squares) and catch (closed diamonds). (b) Estimate of B_0 (open squares) and coefficient of variation in estimate of B_0 (closed diamonds).

next. If there were no observation error (and no process error), this would be straightforward. Given the error in measuring the relative index, the estimator relies on strong contrasts in the relative indices over time in order to distinguish signal from noise. Hilborn (1979) has pointed to a similar need for contrasts in fitting production models to catch and effort data. This explains why a low fishdown rate, in particular, can lead to poor estimates of B_0, especially where the initial estimate of B_0 is low relative to the true value. Initial catches will be very low, and will fail to generate sufficient contrasts in stock size to over-ride observation error. The management strategy becomes locked into a scenario of low B_0 estimates and correspondingly low catches, from which it never escapes.

Some of these effects are shown in the detailed simulations in Figs. 2 and 3. Figure 2 shows a simulation where the initial estimate of stock size is 40% higher than the true value, and where the 10% fishdown rate is applied. Because the initial estimate is high, initial catches are quite high and the stock is rapidly depleted. The estimate of B_0 eventually converges to close to the true value, but not before the stock is reduced to below 20% of B_0. Notice that a zero catch, allowing stock recovery, only occurs when the CV on the B_0 estimate drops below 0.1, as specified in the constraints on the management strategy.

Figure 3 shows a comparison between the 5% and 10% fishdown strategies for identical simulations (i.e., using the same initial B_0 estimate and random number sequence). The initial estimate is only 43% of the true value, so the initial true exploitation rate is low, particularly for the 5% strategy. Figure 3b shows that for the 5% strategy the stock is still at 80% of its virgin level after 15 years of fishing. In consequence, the estimation performance is very poor (Fig. 3d) in comparison to the performance for the 10% strategy (Fig. 3c) where the final B_0 estimate is much closer to the true value.

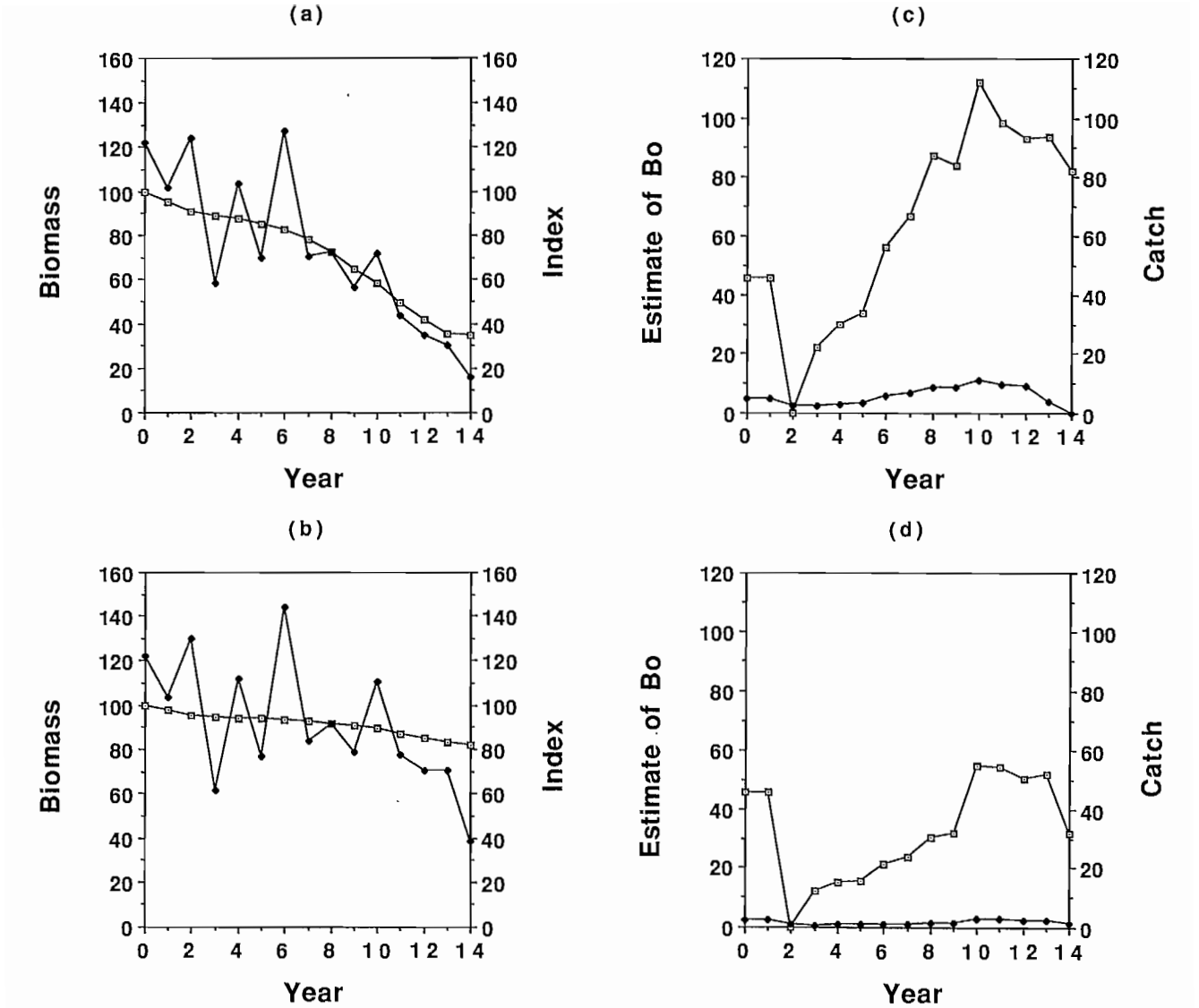


Fig. 3. Detailed time trajectories for simulations using the 5% and 10% fishdown rates and a target biomass of 30% B_0. The same initial estimates and observation errors are used for both rates. Results for the 10% fishdown rate are shown in panels (a) and (c), and for the 5% rate in panels (b) and (d). Panels (a) and (b) show the biomass (open squares) and the index of biomass (closed diamonds). Panels (c) and (d) show the estimate of B_0 (open squares) and the catch (closed diamonds).

Discussion

The results of these simulations suggest that there are likely to be trade-offs between risks (of low stock size), rewards (short term catch) and estimation performance (which will affect longer term catches) in choosing strategies for managing new resources. In particular, in cases where only a relative index of stock size is available, there may be risks associated with a very cautious approach to development of a fishery, as well as risks of too rapid development. In other words, there may be economic risks of underfishing as well as biological risks of overfishing a developing resource.

The robustness of this conclusion warrants further research, as the problem addressed in this paper has been simplified in several respects. For example, the stock dynamics have been deliberately simplified, many of the biological parameters have been assumed known, and no "process noise" (e.g., recruitment variability) has been allowed. The variance in the observation error is not particularly high, nor is the variation in the initial estimate of B_0 from the true value. The modelling of the management process (via the decision rule) could also be made more realistic, for example by specifying more severe constraints on reductions in catch or effort. Some of these additional uncertainties may weigh the conclusions more on the side of caution, but this should be evaluated explicitly.

One key factor in the formulation of the problem addressed in this paper has been the assumption that only a relative index of stock size is available. In research to support management of orange roughy stocks in New Zealand, trawl surveys on

spawning grounds were initially used to try to estimate absolute stock sizes (Robertson 1991). However subsequent stock assessment has used the results of such surveys as relative indices of abundance. This seems to have worked well in providing relatively low variance estimates of B_0 for several of the New Zealand stocks (Francis and Robertson 1990, Clark and Francis 1990). This is because these stocks have apparently been fished down to relatively low proportions of their initial sizes, thus generating the required contrasts.

In Australia, trawl surveys were attempted very early in the development of the orange roughy fishery. However since 1990 there has been a switch to hydroacoustic and egg production surveys of spawning biomass (Smith and Koslow 1990), which in retrospect seem to have provided reasonable absolute estimates of biomass (Smith 1992). The behaviour of at least one stock of orange roughy in south east Australian waters, which has a short lived and highly localised major spawning aggregation, lends itself to these survey techniques.

It seems unlikely that absolute estimates of abundance could be obtained at reasonable cost for many species of fish which may be the target of a developing fishery. The problem addressed in this paper is therefore likely to be a general one, and the choice of development strategy, and in particular the rate of fishdown of the stock, a significant issue. Perhaps a new biological reference point, F_{fishdown}, is required! This study suggests that the choice of such an entity would depend on the trade-offs that are made between biological and economic risks, and between short term and longer term performance.

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Appendix

Given the dynamic model (equations 1 and 2) and the observation model (equation 3), the problem is to estimate B_0 at any time $T \geq 3$, given the complete catch history $\{C_t\}$ $(t = 0, \ldots, T)$, and given a set of observations $\{O_i\}$ $(i = 1, \ldots, m)$ where m is the number of observed indices $(m \leq T)$.

From the dynamic model, we can derive B_τ as a function of B_0, S and $\{C_t\}$:

$$B_\tau = B_0 - \sum_{j=0}^{t-1} S^j C_{t-1-j}$$

and rewrite it as

$$B_{t_i} = B_0 - f_i \tag{A1}$$

where

$$f_i = \sum_{j=0}^{t_i-1} S^j C_{t_i-1-j}$$

Since O_i is distributed as a normal distribution with mean qB_{t_i} and variance σ_i^2, we can write the likelihood function for the $\{O_i\}$ as

$$L = \prod_{i=1}^{m} \left\{ \frac{1}{\sigma_i \sqrt{2\pi}} \exp\left[-0.5 \frac{(O_i - qB_{t_i})^2}{\sigma_i^2} \right] \right\}$$

and the log likelihood, ignoring the terms which are independent of B_0 and q, as

$$\lambda = -0.5 \sum_{i=1}^{m} \frac{(O_i - qB_{t_i})^2}{\sigma_i^2} \tag{A2}$$

Expanding A2 and substituting for B_{t_i} from A1 gives

$$\begin{aligned} \lambda &= -0.5 \left[P_6 - 2qB_0P_4 + 2qP_5 + q^2B_0^2P_1 \right. \\ &\quad \left. - 2q^2B_0P_2 + q^2P_3 \right] \end{aligned}$$

where

$$\begin{aligned} P_1 &= \sum 1/\sigma_i^2 \\ P_2 &= \sum f_i/\sigma_i^2 \\ P_3 &= \sum f_i^2/\sigma_i^2 \\ P_4 &= \sum O_i/\sigma_i^2 \\ P_5 &= \sum f_i O_i/\sigma_i^2 \quad \text{and} \\ P_6 &= \sum O_i^2/\sigma_i^2 \end{aligned}$$

and all sums are over $i = 1, \ldots, m$.

Taking derivatives of λ with respect to B_0 and q

$$\frac{\partial \lambda}{\partial B_0} = qP_4 - q^2B_0P_1 + q^2P_2 \tag{A3}$$

$$\begin{aligned} \frac{\partial \lambda}{\partial q} &= B_0P_4 - P_5 - qB_0^2P_1 \\ &\quad + 2qB_0P_2 - qP_3 \end{aligned} \tag{A4}$$

Equating A3 and A4 to zero, and solving for B_0 and q gives

$$B_0 = \frac{(P_3P_4 - P_2P_5)}{(P_2P_4 - P_1P_5)} \tag{A5}$$

$$q = \frac{(P_2P_4 - P_1P_5)}{(P_1P_3 - P_2P_2)} \tag{A6}$$

which are the maximum likelihood estimates for B_0 and q.

To estimate the uncertainty in the maximum likelihood estimates, let $\boldsymbol{A}$ be a 2×2 matrix where

$$\boldsymbol{A} = -E \begin{bmatrix} \frac{\partial^2 \lambda}{\partial B_0^2} & \frac{\partial^2 \lambda}{\partial B_0 \partial q} \\ \frac{\partial^2 \lambda}{\partial B_0 \partial q} & \frac{\partial^2 \lambda}{\partial q^2} \end{bmatrix}$$

The equations for the individual elements are found by differentiating A3 and A4 with respect to B_0 and q, while the expectation is taken by calculating the elements of the matrix at the maximum likelihood estimates given in A5 and A6. Bard (1974) shows that the covariance matrix $\boldsymbol{V}$ of the vector of parameter estimates $[B_0, q]$ is approximated by

$$\boldsymbol{V} = \boldsymbol{A}^{-1}$$

The first element of the matrix $\boldsymbol{V}$ is an estimate of the variance in the estimate of B_0, from which the CV in $\widehat{B}_0$ is directly obtained.

Estimation of Density-Dependent Natural Mortality in British Columbia Herring Stocks Through SSPA and its Impact on Sustainable Harvesting Strategies

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A separable sequential population analysis model (SSPA), in which natural mortality is parameterized as a function of stock biomass, is developed for the analysis of British Columbia herring fisheries data. The parameters of the structural model are estimated using a maximum likelihood method. Likelihood-ratio tests are used for hypothesis testing and indicate that a model which includes density-dependent natural mortality provides a statistically better fit to the fishery data than does a model assuming a constant natural mortality rate. Results were generally consistent among the five stocks analyzed. The SSPA results were used to conduct simulations to evaluate alternative harvesting strategies for the five herring stocks. The simulation model includes stochasticity in recruitment variability and errors in forecasts of stock abundance. Evaluation criteria included average catch, variance of catch, proportion of years with fishery closures, and minimum spawning stock biomass. Results from simulations assuming constant natural mortality suggest that exploitation rates of 0.4 provide maximum catch and are sustainable providing appropriate minimum spawning stock biomass levels are maintained. Alternately, with a depensatory natural mortality assumption maximum catch is maximized at lower exploitation rates (0.2 for some stocks) and higher minimum spawning stock biomass levels are required to ensure sustainable production. Further evaluation of alternate management objectives will be required prior to advocating a change from the existing 20% harvesting strategy.

Nous élaborons un modèle d'analyse séquentielle séparable de population, dans lequel la mortalité naturelle est paramétrisée comme fonction de la biomasse du stock, pour l'analyse des données sur les pêches du hareng en Colombie-Britannique. Les paramètres du modèle structurel sont estimés à l'aide d'une méthode de la vraisemblance maximale. Des épreuves du rapport de vraisemblance servent à tester l'hypothèse et indique qu'un modèle qui intègre la mortalité naturelle dépendante de la densité offre un ajustement statistiquement meilleur aux données sur les pêches qu'un modèle posant que le taux de mortalité naturelle est constant. Dans l'ensemble, les résultats concordaient parmi les cinq stocks analysés. Les résultats de l'analyse séquentielle ont servi à effectuer des simulations en vue d'évaluer différentes méthodes de prélèvement pour les cinq stocks de hareng. Le modèle de simulation prévoit des variables stochastiques dans le recrutement et des erreurs dans les prévisions de l'abondance des stocks. Les critères d'évaluation étaient le volume moyen des captures, la variance des captures, la proportion des années où la pêche était fermée et la biomasse minimale du stock reproducteur. Les résultats des simulations, dans le cas d'une mortalité naturelle constante, semblent indiquer que des taux d'exploitation de 0,4 donnent des captures maximales et peuvent être soutenus à condition que l'on maintienne des niveaux minimaux suffisants de la biomasse du stock reproducteur. Par contre, si l'on pose l'existence d'une mortalité naturelle anticompensatoire, on maximise les captures maximales à des taux d'exploitation inférieurs (0,2 dans certains stocks), et il est nécessaire d'avoir des niveaux minimaux de la biomasse du stock reproducteur plus élevés pour assurer une production durable. Il sera nécessaire d'évaluer de façon plus approfondie d'autres objectifs de gestion avant de proposer de changer la stratégie actuelle de prélèvement, fixée à 20 %.

The importance of predator-prey species interactions on the population dynamics, and consequently the population response to exploitation, has received considerable attention by fisheries scientists in recent years. Multispecies ecosystem simulation models, such as those developed by Anderson and Ursin (1977) for the North Sea and Laevastu

and Favorite (1980) for the Bering Sea, showed that internal consumption within the ecosystem could far exceed any fishery impact on the abundance of the individual species. More recently, the results of multispecies VPA's have confirmed that predation can be a major influence in the population dynamics of some fish species and therefore have a substantial impact on the management of certain fisheries (Sissenwine and Daan 1991; and papers contained therein). The development of these multispecies assessment and simulation models requires extensive data on the distribution and abundance of the major predator and prey species as well as information on predator-specific prey selectivities.

The collapse of British Columbia's herring stocks in the late 1960's has been attributed to high fishing levels during a period of reduced recruitment (Hourston 1980). Healy (1976) looked for empirical evidence of an interaction between the abundance of salmon and herring through the period of the decline and recovery of the herring stocks, but found no relationship. He suggests that the inability to detect a relationship could be caused by shortcomings in the data rather than the lack of a real relationship. Since the recovery of the herring stocks in the early 1970's the fisheries have been managed with what are considered to be conservative harvest strategies.

For British Columbia herring the detailed data required for multispecies assessment models are not available. An alternate approach is adopted in this paper to evaluate the potential impact of predation on the population dynamics of the herring stocks. We develop a separable sequential population analysis (SSPA) model for analyzing British Columbia herring fisheries data in which natural mortality is parameterized as a function of stock biomass. Density-dependent natural mortality could be a consequence of an increased predation rate at low stock abundance. The inclusion of density-dependent effects in SSPA has previously been used in the analysis of Pacific Cod fisheries data (Fournier 1983). The density-dependent mortality parameterization adds structural complexity to the SSPA model and we require an objective method for testing the hypothesis that this assumption provides a significantly better fit to the fisheries data.

Two estimation methods, based on different statistical models, have had general application for estimating the parameters of SSPA models. The basic difference in the two approaches relates to the assumptions regarding errors in the catch and age composition data. The method proposed by Doubleday (1976) and developed by Deriso et al. (1985) assumes that catch-at-age data are distributed as log-normal random variates, and utilizes the non-linear least squares estimator. The method proposed by Fournier and Archibald (1982) assumes that errors in the observed total catches are independent of the errors in the age composition data and that the total catches are log-normal random variates and the proportions-at-age are multinomial variates. The maximum likelihood method is used to estimate model parameters. It has been shown through simulation analyses that with no error in the structural model and with small sampling error in the data, the two estimation methods are robust to each others' assumptions (Kimura 1990). That is, both methods provided fairly accurate estimates of the model parameters when data was simulated under the statistical assumptions of the alternate method. However, data from real fisheries are generally not very precise and the fishing process often deviates significantly from the idealized structure contained in the model.

We are concerned with aspects of robustness which will lead to correct results from hypothesis tests about alternative forms for the structural model. Hypothesis tests are valid only if the statistical model adequately describes the data. For SSPA models both an appropriate form of the error structure and estimates of the magnitude of the variance for the different types of data are required. Lewy (1988) has approached this problem by explicitly parameterizing the observational error as age-, fleet-, and survey-specific variance components and estimating these variance components simultaneously with parameters of the structural model. The estimates of the coefficients of variation are used as a basis for rejecting unsatisfactory data from the model and hypothesis tests are conducted on the fit to the remaining data. Kimura (1989) adopted a different approach and calculated test statistics for both a minimum and maximum potential value of the relative variance parameter of a Doubleday-Deriso model. For his example the null hypothesis was rejected at both levels of the relative variance parameter, however results from this approach would be ambiguous in situations where the null hypothesis is rejected at one level but not the other.

Fournier et al. (1990) have proposed a robust likelihood-based estimation procedure which they use to test alternative hypotheses about appropriate structural models for describing length-at-age data. The method incorporates a modified normal distribution for modelling the error structure of categorical data. Although their application was the analysis of length-frequency data the statistical model is applicable to other types of categorical data from fisheries. The statistical model they propose is attractive because it is structured on the kind of deviations which are common in fisheries data. The estimation method developed in this paper to test alternate structural hypotheses about natural mortality is based on the robust likelihood-based method suggested by Fournier et al. (1990).

In the first sections of this paper we describe the Herring SSPA model and the methods used to estimate model parameters and test alternative hypotheses. In the final sections we apply the results of the SSPA to harvest simulation analyses and evaluate the effect of density-dependent natural mortality assumptions on sustainable harvest strategies.

The Herring SSPA Model

The Herring Model is structured on the interaction of the life history of Pacific herring and the fisheries which target on them. A brief description of the life history of British Columbia herring and the historical fisheries follows.

Pacific herring spawn inter- and sub-tidally along the protected coastline of bays and inlets. After spawning the fish move offshore and spend the summer feeding on the productive banks of the continental shelf. Juvenile fish move offshore after their first or second year of life, but do not join the adult schools until they are sexually mature. Sexual maturation occurs between ages 1+ and 4+ with the majority of fish maturing at age 2+. During the fall maturing herring move inshore to overwintering areas where they remain until just prior to spawning. Spawning occurs over a short period, generally lasting not more than two to three weeks in any location.

Over the period for which consistent data are available (1951–91) the fisheries have targeted almost entirely on the mature component of the stock. Prior to 1970 the fisheries were for a fishmeal product and occurred on the overwintering fish concentrations. Since 1970 the fisheries have been primarily for a roe product and occur just prior to or during active spawning. Minor winter fisheries have continued since 1970. The fisheries data are readily separated into three types, reflecting both different periods of the year and different gear. The first fishery includes data from all winter fisheries. This fishery has been conducted primarily by seine gear which are non-selective. Therefore all maturing fish are potentially available to this fishery. The second fishery, the seine roe-fishery, occurs just prior (days) to spawning, and is also non-selective. All fish in the mature component of the stock should be equally catchable in this fishery. The final fishery, conducted by gillnets, occurs during active spawning and is selective for larger fish. Again, only the mature component of the stock is available to this fishery. Both age-specific maturation and size-specific gear selectivity will influence the age structure of the catch from the gillnet fisbery. These three fisheries occur during specific periods of the year so the Herring Model is structured with three time periods in each year, reflecting the different fisheries.

Let T_{ij} be the total number of fish in age class j at the beginning of year i, and l_{ij} be the partial recruitment of age class j fish to the spawning, and therefore catchable, component of the stock. Then N_{ij1}, the total number of age class j fish which are available to the fishery at the beginning of period 1 in year i is

$$N_{ij1} = l_{ij} T_{ij}. \tag{1}$$

The fishing process is modelled using a form of catch equations where fishing and natural mortality occur as continuous processes over period r:

$$C_{ijr} = \frac{F_{ijr}}{F_{ijr} + M_{ir}} \left(1 - \exp\left(-F_{ijr} - M_{ir}\right)\right) N_{ijr},$$

and, for $r < 3$

$$N_{ij,r+1} = N_{ijr} \exp\left(-F_{ijr} - M_{ir}\right).$$

$N_{i+1,j+1,1}$ is defined by equation 1, where

$$T_{i+1,j+1} = N_{ij3} \exp\left(-F_{ij3} - M_{i3}\right) + T_{ij}\left(1 - l_{ij}\right) \exp - \sum_{r=1}^{3} M_{ir}. \tag{2}$$

Fournier and Archibald (1982) have shown that if there are significant errors in the ageing of the older age classes model performance is enhanced by grouping these age classes. For these analyses fish aged 6+ and older are grouped and treated as one age class. Thus, for $j + 1 = 6$ equation 2 is replaced by

$$\begin{aligned} T_{i+1,6} = {} & N_{i53} \exp\left(-F_{i53} - M_{i3}\right) \\ & + T_{i5}\left(1 - l_{i5}\right) \exp\left[-\sum_{r=1}^{3} M_{ir}\right] \\ & + N_{i63} \exp\left(-F_{i63} - M_{i3}\right) \\ & + T_{i6}\left(1 - l_{i6}\right) \exp\left[-\sum_{r=1}^{3} M_{ir}\right]. \end{aligned}$$

Certain structural assumptions are made about the form of the partial recruitments and the mortalities to reduce the number of parameters which must be estimated. Age specific partial recruitment is assumed to be constant among years for fish aged 3+ and older and is fixed at 1 for age class 5 (i.e., 3 parameters, l_3, l_4, l_6, replace 4×41 parameters l_{ij}). The proportion of age 1+ and age 2+ fish which mature each year appears to be variable (Haist and Stocker 1985), therefore annual partial recruitment parameters (l_{i1} and l_{i2}) are estimated for these two age classes.

Fishing mortality is generally envisioned as an interaction of the intensity or effort of the fishing fleet and the selectivity of the fishing gear. Seine fishing gear is non-selective. That is, entire schools are caught and all fish in the fishing location are equally vulnerable. For the winter ($r = 1$) and seine roe-fisheries ($r = 2$) all sexually mature or maturing fishing are available, so fishing mortality is parameterized strictly as a function of fishing intensity. Fishing intensity parameters (a_{ir}) are estimated such that

$$\ln\left(F_{ijr}\right) = a_{ir}.$$

The gillnet fishery ($r = 3$) is mesh size restricted and is selective for larger and therefore older fish. The average size-at-age in the stocks varies substantially between years, and for some British Columbia herring stocks has shown notable trends. Therefore the selectivity of gillnets for an age-class is parameterized as a function of the geometric mean fish weight-at-age (g_{ij}). Fishing mortality is parameterized as the product of fishing intensity and gear selectivity as follows,

$$\ln\left(F_{ij3}\right) = a_{i3} + f\left(g_{ij}\right).$$

where,

$$f(g_{ij}) = \frac{1}{1 + \exp\left(b_1 - b_2 g_{ij}^{b_3}\right)}.$$

Two alternate hypotheses regarding natural mortality are evaluated in the SSPA model. The first is that natural mortality is constant among years. Then the instantaneous natural mortality rates for years i ($M_{i\cdot} = \sum_r M_{ir}$) are estimated by a single parameter m_1. The second hypothesis is that natural mortality is density-dependent, and the $M_{i\cdot}$ are parameterized as a function of the age 2+ and older stock biomass ($B_i = \sum_{j=2}^{6} w_{ij} T_{ij}$). The total natural mortality rate is estimated for each year by,

$$M_{i\cdot} = m_1 + \exp\left[-m_2 \frac{B_i}{B_1}\right].$$

Estimates of the proportion of the annual natural mortality rate occurring during each period are not well determined and are therefore held constant as follows:

$$\begin{aligned} M_{i1} &= 0.95 M_{i\cdot} \\ M_{i2} &= M_{i3} = 0.025 M_{i\cdot}. \end{aligned}$$

This partitioning of the annual natural mortality rate reflects that the first fishing period encompasses most of the year and the second and third fisheries occur in the final two to three weeks of the year.

Auxiliary information is available from spawn surveys which have been conducted annually to obtain indices of total egg deposition (Schweigert and Stocker 1988). Spawning occurs at the end of the year so the biomass of spawning fish in year i (G_i) is estimated by

$$G_i = \sum_j w_{ij} N_{ij3} \exp\left(-F_{ij3} - M_{i3}\right),$$

where w_{ij} is the average weight-at-age j in year i. The relationship between female fish weight and fecundity is linear (Hay 1985), so the estimated spawning stock biomass provides a reasonable index of the population fecundity (I_i),

$$I_i = q G_i,$$

where q is a spawn index conversion factor.

The fundamental model parameters, in terms of which all other model parameters can be expressed, are the $T_{i1}, T_{1j}, l_{i1}, l_{i2}, l_3, l_4, l_6, a_{ij}, b_1, b_2, b_3, m_1, m_2, q$. Note that the parameter m_2 is estimated only for the "density-dependent M" model. The w_{ij} and g_{ij} are data supplied to the model.

Likelihood-based Estimation Method

At this point we have specified the structural component of the Herring Model but not the statistical assumptions and the parameter estimation method. We assume that the errors in the proportion-at-age data ($\widetilde{P}_{ijr}$) are independent of the errors in the total catch data ($\widetilde{C}_{ir}$). These data relate to the following predicted quantities of the structural model:

$$\begin{aligned} C_{ir} &= \sum_j C_{ijr}, \\ P_{ijr} &= \frac{C_{ijr}}{C_{ir}}. \end{aligned}$$

We assume the statistical model for categorical data proposed by Fournier et al. (1990) is appropriate for the age-composition data ($\widetilde{P}_{ijr}$), and that the total catch data ($\widetilde{C}_{ir}$) and the spawn index data ($\widetilde{I}_i$) are distributed as independent log-normal variates. Maximum likelihood estimators are used to estimate model parameters.

Fournier et al. (1990) suggest that there are two types of deviation from model assumptions which are common in categorical data from fisheries and lead to non-robust parameter estimation. The first, called a type I deviation, results from the occasional occurrence of an event which has a very low probability. An example of this type of deviation for the Herring Model is the occurrence of a small proportion of fish in the first age class in samples from gillnet fisheries. In general, fish in this age class are too small to be retained by gillnet gear, however occasionally samples contain a small proportion of these fish. This may result from the use of illegal undersized fishing gear, a not uncommon practice in this fishery. The other type of deviation occurs in a region where the probability of observing fish at a particular age is high but the actual frequency of occurrence is either much lower or much higher than would be expected given the overall accuracy of the majority of the observations. This is called a type II deviation and could result from fishery bias, such as if fisheries occasionally target on a specific year-class.

If the $\widetilde{P}_{ijr}$ were multinomial variates, then the variance of the $\widetilde{P}_{ijr}$ would be $P_{ijr}(1 - P_{ijr})/S_{ir}$, where S_{ir} is the sample size. The $P_{ijr}(1 - P_{ijr})$ terms represent the variance due to the magnitude of the P_{ijr} and the S_{ir} represent the overall variance due to the size of the sample. Let ξ_{ijr} replace the terms $P_{ijr}(1 - P_{ijr})$. As P_{ijr} tends to zero, the corresponding ξ_{ijr} tends to zero as well. This reflects the fact that for random sampling an event with a very small probability will almost never occur. However, for real world data, highly unlikely events seem to occur quite often. To render the model less sensitive to the occurrence of unlikely events, a lower bound is placed on the relative variances of the $\widetilde{P}_{ijr}$. We assume that the age-composition samples from all fisheries are equally accurate and the sample size term S_{ir} is replaced by a parameter τ to be estimated within the model. The variance of the random variable $\widetilde{P}_{ijr}$ is assumed to be equal to $(\xi_{ijr} + 0.01)\tau^2$. The constant 0.01, is added to the relative variance component to render the model insensitive to type I deviations.

If the $\widetilde{P}_{ijr}$ were independent, normally distributed random variables, their likelihood function would be

$$\prod_{ijr}\left[\frac{1}{\tau\sqrt{2\pi\left(\xi_{ijr}+0.01\right)}}\times\left\{-\frac{\left(\widetilde{P}_{ijr}-P_{ijr}\right)^2}{2\tau^2\left(\xi_{ijr}+0.01\right)}\right\}\right]$$

The $\widetilde{P}_{ijr}$ are of course not independent observations and therefore this maximum likelihood function is not strictly correct. However, to motivate the use of this likelihood function to estimate model parameters Fournier et al. (1990) show that it is approximately equivalent to the minimum χ^2 method.

This likelihood function, based on the normal distribution, is too sensitive to large deviations from the expected value P_{ijr}. To make the function more robust and insensitive to type II deviations Fournier et al. (1990) assume there is a small uniformly distributed contamination of the normal distribution and an appropriate term is added to the normal density function. A different approach is adopted in this analysis to account for type II deviations. That is, all observations which are greater than three standard deviations from their predicted value are considered outliers, and are eliminated from the calculation of the likelihood function. The log-likelihood function for the parameters P_{ijr} and τ is

$$-\frac{1}{2}\sum_{ijr}\left[\ln\left(2\pi\left(\xi_{ijr}+0.01\right)\right)+\ln\left(\tau^2\right)+\frac{\left(\widetilde{P}_{ijr}-P_{ijr}\right)^2}{\tau^2\left(\xi_{ijr}+0.01\right)}\right],$$

where the summation is only over ijr where $C_{ijr}>0$ and $\frac{\widetilde{P}_{ijr}-P_{ijr}}{\tau\sqrt{\xi_{ijr}+0.01}}\leq 3$. To evaluate the effect of excluding the outliers from the calculation of the function value analyses are also conducted where they are not excluded.

The error structure of the total catch and spawn index observations are modelled as follows. We assume that the total catch ($\widetilde{C}_{ir}$) and the spawn index observations ($\widetilde{I}_i$) are distributed as independent log-normal random variates with variances of σ_C^2 and σ_I^2, respectively. Then the log-likelihood function for the parameters C_{ir} and σ_C^2 is

$$-\frac{1}{2}\sum_{ir}\left[\ln 2\pi+\ln\left(\sigma_C^2\right)+\frac{\left(\widetilde{C}_{ir}-C_{ir}\right)^2}{\sigma_C^2}\right].$$

The log-likelihood function for the parameters I_i and σ_I^2 is

$$-\frac{1}{2}\sum_{i}\left[\ln 2\pi+\ln\left(\sigma_I^2\right)+\frac{\left(\widetilde{I}_i-I_i\right)^2}{\sigma_I^2}\right].$$

It is not possible to estimate the variances of the model components simultaneously with parameters of the structural model and these parameters are estimated through an iterative reweighting method (Mosteller and Tukey 1977). The maximum likelihood estimator for the variance components is used to provide the iterative weights. The use of iterative reweighting to obtain variance estimates of abundance indices in age-structured analyses has been proposed by Powers and Restrepo (1992). The method we use is,

1. Trial values are set for the parameters τ, σ_C^2, σ_I^2. These are denoted by $S_\tau'^2$, $S_C'^2$, $S_I'^2$.

2. Minus the value of the log-likelihood function is minimized and the resultant parameters of the structural model are used to estimate the variances for each model component:

$$S_\tau^2 = \sum_{ijr}\frac{\left(P_{ijr}-\widetilde{P}_{ijr}\right)^2/\left(\xi_{ijr}+0.01\right)}{n_\tau}$$

$$S_C^2 = \sum_{ir}\frac{\left(C_{ir}-\widetilde{C}_{ir}\right)^2}{n_C}$$

$$S_I^2 = \sum_{i}\frac{\left(I_i-\widetilde{I}_i\right)^2}{n_I}$$

where the n_τ, n_C and n_I are the number of $\widetilde{P}_{ijr}$, $\widetilde{C}_{ir}$ and $\widetilde{I}_i$ observations in the summation, respectively.

3. The S_τ^2, S_C^2, S_I^2 obtained in step 2 replace the $S_\tau'^2, S_C'^2, S_I'^2$. Step 1 is repeated with these updated estimates of the variance components.

Steps 2 and 3 are repeated until the S_τ^2, S_C^2, S_I^2 converge to the $S_\tau'^2, S_C'^2, S_I'^2$ estimates from the previous iteration. A wide range of trial values were tested and in all cases the variance estimates converged to the same values.

A χ^2 test is used to determine what constitutes a significant increase in the maximum value of the log-likelihood function. We have employed the standard theory which states that under the hypothesis that the present model is the correct one, if t additional parameters are added to the model, then twice the increase in the maximum value of the log-likelihood function is asymptotically distributed as a χ^2 random variable with t degrees of freedom. In practice, minus the value of the log-likelihood function is minimized, therefore a decrease in the function value of 1.92 for the model incorporating the one additional density-dependent parameter implies a significant improvement in the model fit at the 0.05 probability level.

The SSPA is conducted for five British Columbia herring stocks (Fig. 1), for both the "constant M" and the "density-dependent M" structural model. A quasi-Newton function minimizer, which requires values of the function and its first derivatives with respect to the fundamental model parameters, is used to minimize minus the value of the log-likelihood function. The first derivatives are calculated analytically. The data observations are as presented in Haist and Schweigert (1992). The variance components are estimated only for the "density-dependent M" model; for other SSPA runs the error variances are fixed at these estimated values.

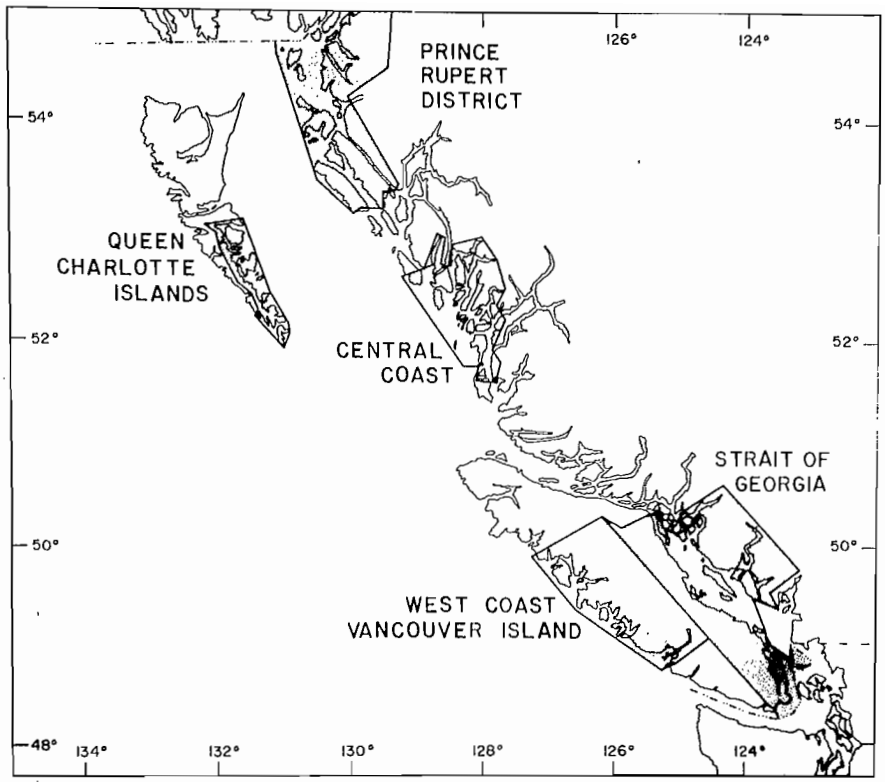


Fig. 1. Herring stock assessment regions in British Columbia. The stocks are; the Queen Charlotte Islands (QCI), the Prince Rupert District (PRD), the Central Coast (CENT), the Strait of Georgia (GULF) and the West Coast of Vancouver Island (WCVI).

SSPA Results

Attempts to estimate the variance of the total catch random variables ($\widetilde{C}_{ir}$) were unsuccessful. Even with trial values set extremely high, the estimated variance of these data approached zero after a few iterations. The reason for this is intuitively clear from the model structure. The model can fit the observed total catch data perfectly through adjustment of the fishing intensity parameters with minimal impact on the relative numbers at age in the population. Therefore, little improvement in model fit is obtained with parameters which lead to differences between the observed and predicted total catches. The trial variance of this model component was set at 0.0025 and not estimated through the iterative procedure.

Estimates of the parameters S_τ^2 and S_I^2 are shown in Table 1 for the five stock assessment regions. The estimates of the overall variance scaling parameter (τ) for the $\widetilde{P}_{ijr}$ range from 0.0017 for the CENT stock to 0.0073 for the PRD stock. When these are combined with the relative variance component for the observations, the standard errors for an expected proportion of 0.3 range from 0.0193 to 0.0401. Over all stocks, 0.0533 of the proportion-at-age observations were excluded from the calculation of the log-likelihood function value because their residual values were greater than 3. For a standard normal distribution the expected proportion of observations greater than 3 standard deviations from the mean is 0.0026. The most extreme outliers were > 10 standard deviations from the mean.

The estimates of the variance of the spawn index data also differ substantially among stocks (Table 1). For the WCVI and QCI stocks the variance of the residuals are greater than the variance in the data observations suggesting there is limited information regarding stock trends in the survey data. An assumption of a constant spawning stock would provide a better agreement with the catch and age-composition data than the actual spawn index observations. The best agreement between the observed and predicted spawn index data (i.e. the smallest S_I^2) is obtained for the GULF stock. This stock resides in the most populated area of the British Columbia coast and hence has received the most consistent spawn survey coverage over the time period of the analyses.

Table 1. Estimates of the residual variances for SSPA model components (S_τ^2, S_C^2, S_I^2) for the "density-dependent M" model. The variances for the total catch component (S_C^2) are estimated with the trial value fixed at 0.0025. The among years variance of the spawn index observations ($V(\widetilde{I}_i)$) and the proportion of the $\widetilde{P}_{ijr}$ observations which are greater than three standard deviations from their predicted value (outliers) are also presented.

Stock	S_τ^2	S_C^2 $\times 10^{-5}$	S_I^2	$V(\widetilde{I}_i)$	Proportion of outliers
QCI	0.0021	0.945	1.2886	1.0201	0.0775
PRD	0.0073	0.276	0.1613	0.4510	0.0670
CENT	0.0017	5.626	0.2820	0.4918	0.0526
GULF	0.0028	1.872	0.0433	0.3788	0.0400
WCVI	0.0031	0.726	0.3660	0.2114	0.0345

Table 2. The log-likelihood function value for the total function and for the age-composition and spawn index components for the "constant M" and the "density-dependent M" SSPA models for five British Columbia herring stocks. Results are presented for analyses where outliers are excluded from the calculation of the function value and also for analyses where they are included.

		Log-likelihood function value			
		Outliers excluded		Outliers included	
Stock	Model component	Constant M	Density-dependent M	Constant M	Density-dependent M
QCI	Total	197.08	163.37	189.52	173.81
	Age-composition	169.41	142.74	165.96	153.18
	Spawn index	26.93	20.54	23.22	20.38
PRD	Total	226.26	210.78	252.01	231.14
	Age-composition	197.79	190.55	221.44	210.63
	Spawn index	28.31	20.20	30.45	20.50
CENT	Total	202.80	192.24	210.60	201.23
	Age-composition	180.71	171.03	185.16	180.51
	Spawn index	21.50	20.54	25.12	20.47
GULF	Total	251.48	245.86	267.24	255.80
	Age-composition	223.44	225.07	237.52	235.06
	Spawn index	27.66	20.50	29.36	20.50
WCVI	Total	198.06	180.79	201.13	186.55
	Age-composition	166.15	160.16	171.92	165.99
	Spawn index	31.64	20.55	29.00	20.52

The log-likelihood function values (minus the constant component) for both the SSPA's assuming constant natural mortality and those assuming density-dependent natural mortality are shown in Table 2. The decrease in the total function value for the analyses assuming density-dependent mortality are all greater than 1.92 ranging from 5.62 for the GULF stock to 33.71 for the QCI stock. The probabilities of obtaining these decreases in the function value through chance alone are all < 0.001, indicating that the density-dependent model structure provides a significantly better fit to the observed data. The log-likelihood function contributions from the total catch data are all < 0.7 and are not presented in the table. The relationships between stock biomass and the instantaneous natural mortality rate are shown in Fig. 2. All relationships indicate depensatory mortality, consistent with the assumption of increased predation mortality as stock abundance declines.

To evaluate the effect of excluding outliers from the calculation of the function value additional analyses were conducted where the outliers were not excluded. The log-likelihood function values obtained for the SSPA's with the constant natural mortality assumption and with the density-dependent natural mortality assumption are shown in Table 2. For these analyses the decrease in the total function value for the analyses assuming density-dependence range from 9.37 for the CENT stock to 20.87 for the PRD stock. Although

Table 3. Estimates of the means and coefficients of variation (CV) of stock biomass and biomass lost to natural mortality for the "constant M" and "density-dependent M" analyses.

		Constant M		Density-dependent M	
Stock		Mean	CV	Mean	CV
QCI	Stock biomass	64,300	0.71	99,600	0.63
	Natural Mortality	22,100	0.77	37,800	0.51
PRD	Stock biomass	82,100	0.42	117,800	0.37
	Natural Mortality	19,800	0.45	29,400	0.21
CENT	Stock biomass	67,300	0.47	69,600	0.40
	Natural Mortality	18,300	0.56	19,400	0.30
GULF	Stock biomass	141,200	0.44	131,000	0.34
	Natural Mortality	60,000	0.46	53,300	0.27
WCVI	Stock biomass	71,500	0.66	78,300	0.41
	Natural Mortality	21,200	0.75	24,900	0.25

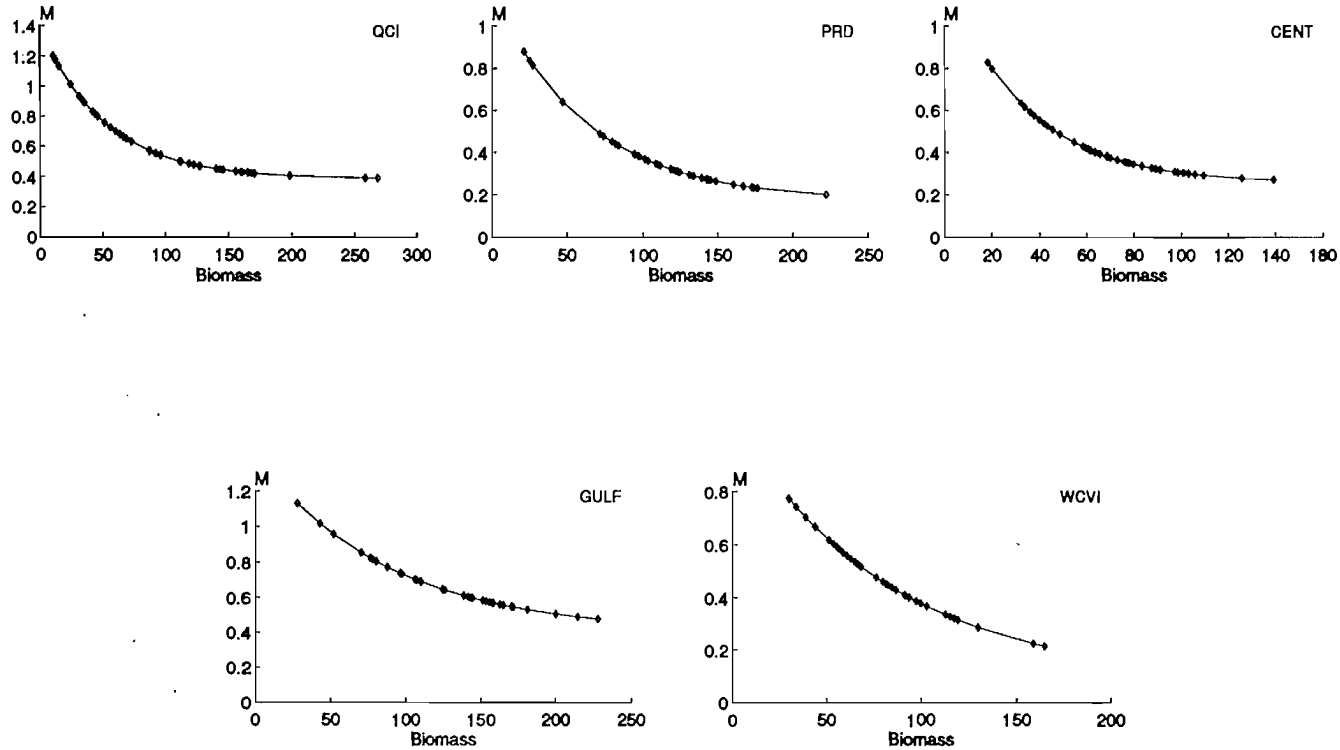


Fig. 2. The estimated relationship between the annual instantaneous natural mortality rate (M) and stock biomass for five British Columbia herring stocks.

the magnitude of the decrease in the function value changes substantially for some of the stocks the probabilities of observing these decreases through chance remains small (i.e., all are < 0.001).

Table 3 presents estimates of the means and coefficients of variation of the annual stock biomass and the biomass lost to natural mortality for both SSPA models. The estimated interannual variabilities in stock biomass are lower for the "density-dependent M" model. Also, the "density-dependent M" model suggests there is considerably less interannual variation in the biomass lost to natural causes than what is implied by the "constant M" model.

Figure 3 shows the log-likelihood function values obtained with M, or average M for the density-dependent model analyses, fixed at various levels. The 95% confidence limits on M, that is the points where the log-likelihood function value are 1.92 greater than the minimum value, are generally smaller for the analyses assuming constant M. The 95% confidence limits range from approximately $M \pm .05$ for the GULF stock to $M \pm .15$ for the WCVI stock with the density-dependent M assumption. A retrospective analysis, based on the Herring SSPA model, has shown that inappropriate specification of M leads to persistent biases in the stock assessments and forecasts for these stocks (Haist 1991).

Harvest Simulation Model

The harvest simulation model follows the structure of the Herring Model and uses parameters estimated from the SSPA to simulate stock and fishery dynamics. The simulations are conducted with only one type of fishery, a non-selective one, occurring in the third fishing period. This is consistent with the current industry focus on roe-herring fisheries. The harvest strategy evaluated is a fixed exploitation rate where annual quotas are determined as a proportion of the forecast stock abundance. For each exploitation rate evaluated, a minimum spawning stock biomass (MSSB) is determined such that the probability of stock collapse is zero. Stock collapse is empirically defined as the spawning stock biomass going below 2% of the estimated unexploited average spawning stock biomass.

The harvest simulation model contains two stochastic components, recruitment variability and errors in the forecasts of stock abundance. Stock forecasts are comprised of adult (age 3+ and older) and recruit (age 2+) fish, with different uncertainty for each. The error in forecasts of adult fish is proportional to their abundance,

$$\widehat{A}_k = A_k(1 + \varepsilon_k)$$

where A_k is the actual biomass of adult fish, $\widehat{A}_k$ is the forecast biomass of adult fish, and the ε_k are standard normal deviates with mean 0 and standard deviation of 0.25. This level of error is based on the probability distributions of stock forecasts

obtained from bootstrap analyses of the SSPA model (Haist 1991). The forecasts of recruit fish assume there is no information regarding the abundance of this year class on which to base the forecast, and the average biomass of recruits (V), estimated for the 1951–91 time period, is used. Annual quotas (Q_k) are calculated based on the forecast stock abundance, the exploitation rate (h), and the MSSB as follows:

$$Q_k = \begin{cases} h\left(\widehat{A}_k + V\right), & \text{if } \widehat{A}_k + V - Q_k \geq \text{MSSB} \\ \widehat{A}_k + V - \text{MSSB}, & \text{if } \widehat{A}_k + V - Q_k < \text{MSSB} \\ 0, & \text{if } \widehat{A} < \text{MSSB}. \end{cases}$$

Thus, when stock forecasts are less than the MSSB, no harvest occurs. When the specific exploitation rate (h) leads to quotas which would leave a spawning stock less than the MSSB, the harvest is curtailed.

The simulated population contains variability in recruitment which is generated assuming a log-normal distribution. The estimated mean ($\overline{\ln R}$) and variance (S_R^2) of the log of the recruitment estimates from the SSPA's are calculated for each stock (Table 4):

$$\overline{\ln R} = \frac{1}{n}\sum_{i=1}^{n} \ln T_{i1},$$

$$S_R^2 = \frac{1}{n-1}\sum_{i=1}^{n}\left(\ln T_{i1} - \overline{\ln R}\right)^2$$

Then the simulated series of recruitment estimates (R_k) are generated by,

$$R_k = \exp\left(\overline{\ln R} + \varepsilon_k\right),$$

where the ε_k are random normal deviates with mean 0 and variance S_R^2, and k indexes the simulation year.

Harvest simulations are conducted using the parameter estimates from the SSPA for the "constant M" and the "density-dependent M" analyses. For each stock, M assumption, and exploitation rate (ranging from 0 to 0.40), 1000 replicates of 25 year simulations are conducted. The evaluation criteria include; average catch, variance of catch, proportion of years with fishery closures, and MSSB. The initial value for MSSB is zero. If this level leads to stock collapse, the simulation is conducted with MSSB set at 0.10 of the estimated unexploited average stock biomass and this level is incremented in steps of 0.02 until the probability of stock collapse is zero.

Harvest Simulation Results

The harvest simulations indicate that under the assumption of a constant M there is no requirement for a threshold spawning stock biomass (MSSB) for exploitation rates below 0.4 (Table 5). All five stocks appear to be resilient to collapse at exploitation rates up to 0.3. The requirement for a MSSB at the high exploitation level is, of course, the result of inaccuracy in stock forecasts. The simulations with the density-dependent natural mortality model suggest that threshold spawning stock levels may be required at exploitation rates as low as 0.10. The results differ between stocks, and for the CENT and GULF stocks a MSSB is invoked only at exploitation rates of 0.3 and higher. For all stocks the level of MSSB, expressed as a proportion of the unexploited average stock biomass, increases as the exploitation rate increases. Coincident with the increase in the MSSB level, the number of years where the fisheries are closed or quotas reduced from the target exploitation level also increases (Table 6). Fisheries are closed or reduced on average 15% of the time at a 0.3 exploitation rate, and this increases to 51% for a 0.4 exploitation rate.

Table 4. The means ($\overline{\ln R}$) and standard deviations (SD) of the ln recruitment estimates (1951–90) from the "constant M" and the "density-dependent M" SSPA's for five British Columbia herring stocks.

	Constant M		Density-dependent M	
Stock	$\overline{\ln R}$	SD	$\overline{\ln R}$	SD
QCI	7.3241	1.0660	7.8783	0.9321
PRD	7.6231	0.9176	7.9334	0.8401
CENT	7.6833	0.8194	7.7613	0.7642
GULF	8.8657	0.6138	8.8478	0.4624
WCVI	7.7892	0.8162	8.0101	0.5929

Results from the harvest simulation model incorporating the constant M assumption indicate that average catch increases as the exploitation rate increases, at least to the highest exploitation rate examined (Fig. 4). The simulations assuming density-dependent M suggest somewhat different exploitation levels for MSY. For these simulations the exploitation rate at which the maximum catch is attained is stock dependent and ranges from 0.2 to 0.4.

The decrease in average spawning stock biomass from the estimated unexploited stock level resulting from various exploitation rates is shown in Fig. 5. The unexploited spawning stock biomass estimates from the simulation model assuming density-dependent M are higher than those from the model assuming constant M as is the apparent decrease in spawn levels resulting from exploitation. At an exploitation rate of 0.2 the average spawning stock biomasses are 56% of the unexploited level for the constant M model and 44% of the unexploited level for the density dependent M level. At an exploitation rate of 0.3 these estimates decrease to 44% and 31%, respectively.

Discussion

Studies to date which have looked for predator-prey interactions by relating the abundance of two species have generally been unsuccessful. Examples of predator-prey interactions affecting the recruitment of either the predator or the prey species are more common (eg., Fogarty et al. 1991; Tyler and Crawford 1991; Walters et al. 1986; Ware and McFarlane 1986). The reasons for the inability to detect relationships

between the abundance of a predator and the abundance of its prey may be related to the complexity of most ecosystems and the numerous species interactions. Most predator species feed on a variety of prey species and do so opportunistically. Additionally, when the abundance of an important prey species is significantly reduced there is often a concomitant increase in the abundance of a different species which may provide alternate forage.

Although herring are an important prey species in coastal British Columbia waters, their major predators are opportunistic, feeding on a variety of prey. For example, during the winter months when pre-spawning herring congregate within the Strait of Georgia, herring comprise 70% of the diet of harbour seals. For the remainder of the year, when the adult herring have moved offshore, the harbour seal diet is predominantly comprised of hake (Olesiuk et al. 1990). Thus, the

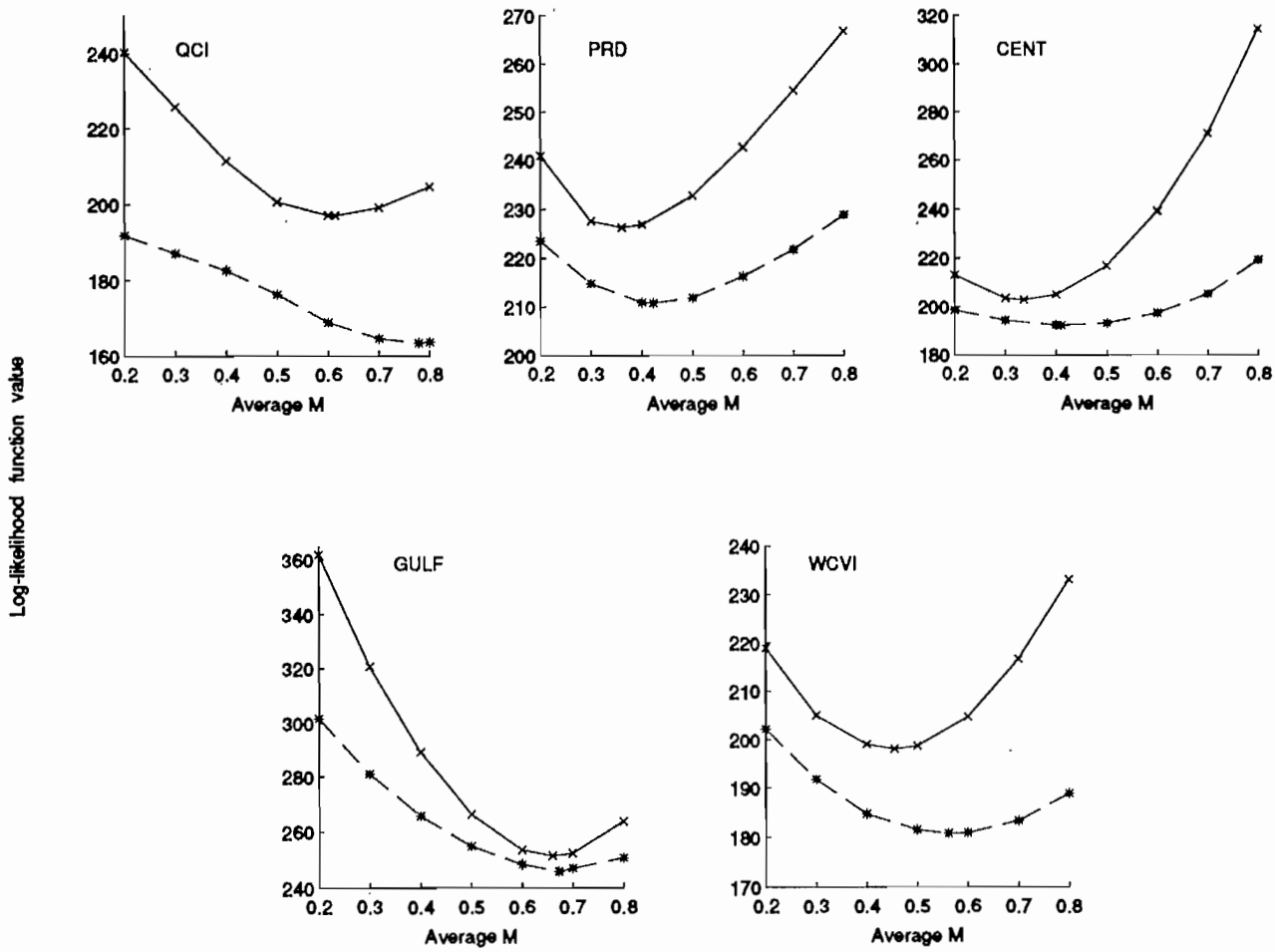


Fig. 3. The log-likelihood function value (minus the constant component) estimated for various levels of M for the "constant M" (—) and "density-dependent M" (– – –) herring models.

Table 6. Minimum spawning stock biomass (MSSB), expressed as a proportion of the unexploited average spawning stock biomass, required to avoid stock collapse for various exploitation rates for the "constant M" and "density-dependent M" models.

	Constant M				Density-dependent M			
	Exploitation rate				Exploitation rate			
Stock	0.1	0.2	0.3	0.4	0.1	0.2	0.3	0.4
QCI	0	0	0	0.16	0.10	0.14	0.14	0.18
PRD	0	0	0	0.14	0	0.10	0.10	0.22
CENT	0	0	0	0.14	0	0	0.12	0.16
GULF	0	0	0	0.24	0	0	0.18	0.22
WCVI	0	0	0	0.28	0	0.10	0.12	0.14

Table 7. Percentage of years with reduced fisheries or fishery closures due to a forecast stock abundance less than the MSSB for the "constant M" and "density-dependent M" model simulations.

	Constant M				Density-dependent M			
	Exploitation rate				Exploitation rate			
Stock	0.1	0.2	0.3	0.4	0.1	0.2	0.3	0.4
QCI	0	0	0	21.5	0.8	8.2	21.0	46.0
PRD	0	0	0	10.6	0	4.2	13.4	64.2
CENT	0	0	0	10.4	0	0	11.8	43.5
GULF	0	0	0	12.0	0	0	4.0	42.4
WCVI	0	0	0	54.2	0	1.4	25.4	60.8

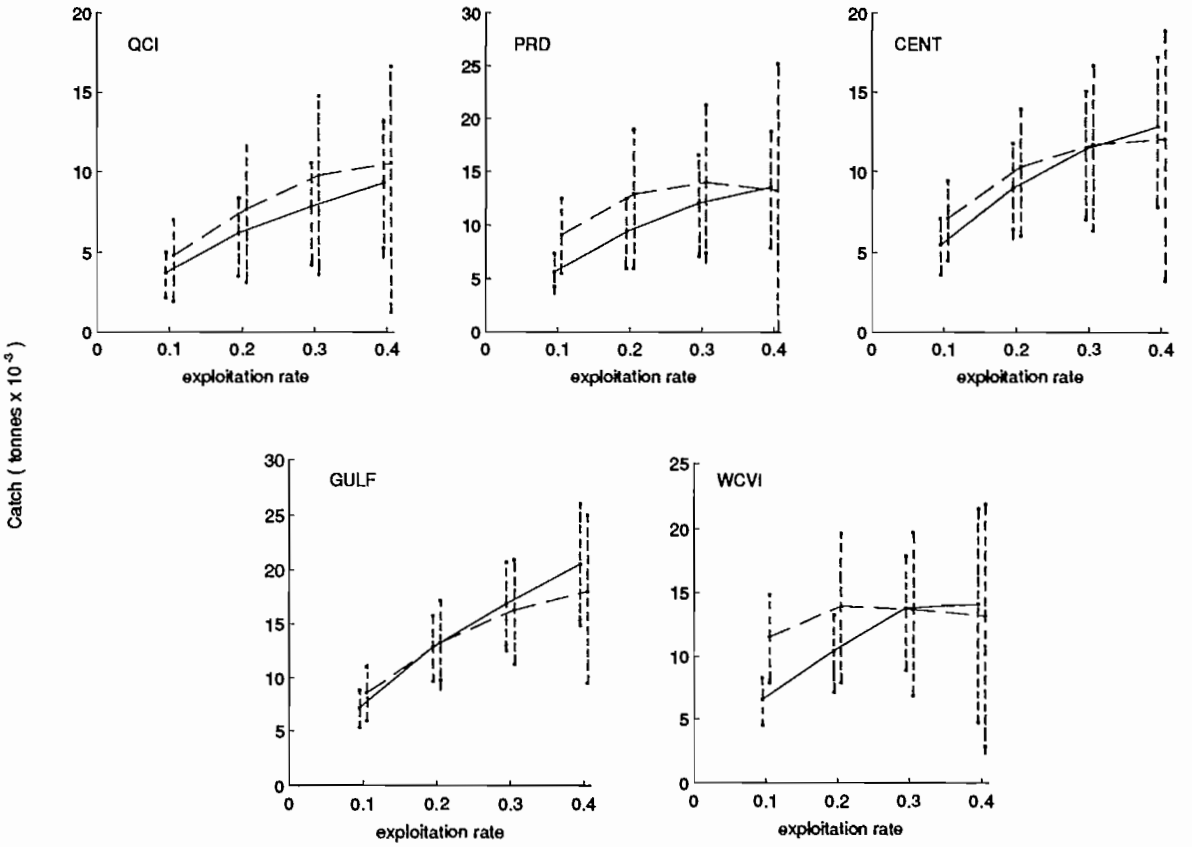


Fig. 4. The average catch attained at exploitation rates of 0.1 to 0.4 from harvest simulations assuming "constant M" (—) and "density-dependent M" (– – –). The vertical lines indicate the range of observations within the 20 and 80 percentile points.

predators may have a greater influence on the abundance of the herring stocks than the herring have on the abundance of their predators. Also, while individual predator populations fluctuate, the abundance of the total predator community may be relatively more stable. For the Peruvian upwelling system, Jarre et al. (1991) estimate the historical mortality rates of anchoveta caused by each of its major predators on the basis of predator-specific predation rates and stock abundance estimates. While the mortality caused by individual predator species was highly variable, the total mortality caused by the combined predator community was much more stable.

The results from our analyses show that an increase in the natural mortality rate at low herring stock abundance provides a significantly better fit to the catch-age fisheries data than an assumption of a constant natural mortality rate. While this result is consistent with the idea that the predator community consumes a higher proportion of the stock as the stock biomass decreases, the depensatory natural mortality could also result from other mortality sources such as disease, starvation, lethal environmental conditions or senescence. However, the impact of disease and starvation are more likely to increase with population biomass, and environmental conditions and senescence should be independent of stock biomass.

For southern British Columbia, estimates of herring consumption by some of their major predators are available from

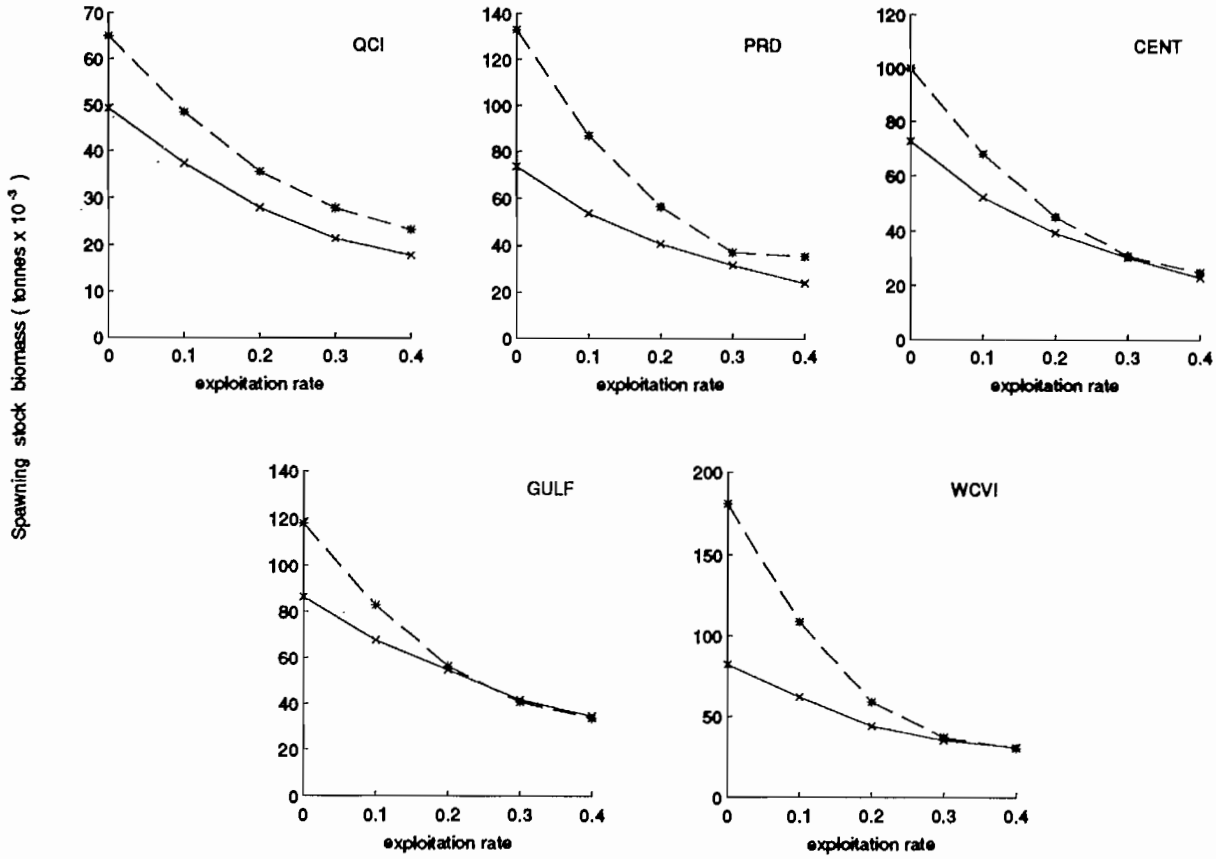


Fig. 5. The average spawning stock biomass (SSB) estimated at various exploitation rates for the "constant M" (—) and the "density-dependent M" (— — —) simulation models.

feeding studies and can be compared with the estimates of biomass lost to natural mortality obtained in this study. Both the GULF and WCVI herring stocks oversummer on the La Perouse Bank off the west coast of Vancouver Island. In our study the average annual loss of adult biomass to natural mortality for these two stocks was estimated at 78,200 tonnes (Table 3). D. Ware (DFO, Nanaimo, B.C. pers. comm.) has estimated that the major fish predators on La Perouse Bank consume approximately 60,000 t. of herring annually. Additionally, within the Strait of Georgia harbour seals consume 3,200 t. (Olesiuk et al. 1990), seal lions consume 2,400 t. (Olesiuk, pers. comm.) and salmon consume approximately 12,000 t. (Healy 1976) of the GULF herring stock annually. While these estimates include consumption of both immature and adult herring, it is an incomplete list of herring predators and does suggest that predation may be the major cause of natural mortality in these stocks.

It has been suggested that increased predation mortality may have been a contributing factor in the collapse of some of world's important pelagic fisheries (Saville 1980; Sissenwine et al. 1984). The results from these analyses imply that an increase in natural mortality contributed to the decline of British Columbia's herring stocks in the late 1960's. While a series of poor year-classes in conjunction with continued intensive fisheries initiated the stock declines, an increase in the natural mortality rate would have intensified the decline. While attributing the increase in natural mortality to predation is strictly speculative, our results suggest that density-dependent natural mortality may be an important consideration in the management of fisheries for these herring stocks.

Previous simulation studies have been conducted to evaluate alternate exploitation strategies for a number of herring stocks on the Pacific coast of North America. These studies incorporated a variety of models for the relationship between stock and recruitment but all assumed that natural mortality did not vary between years. Results obtained in this analysis from the harvest simulations assuming constant M are in general agreement with results from these previous studies. For a constant exploitation rate strategy, Trumble (1983), Hall et al. (1988), and Haist (1990) estimated that maximum sustainable yield (MSY) would be attained at an exploitation rate of 0.4. The analysis presented by Fried and Wespestad (1985) suggests that an exploitation rate of 0.3 will provide MSY for the eastern Bering Sea herring stocks. The studies by Hall et al. (1988) and Fried and Wespestad (1985) suggest that threshold spawning stock levels are not required, whereas the results of Trumble (1983) and Haist (1990) suggest a MSSB is required to avoid stock collapse at exploitation rates of 0.4 and higher. Funk (1991) conducted yield per recruit analysis for two Alaskan herring stocks and estimated $F_{0.1}$ values of 0.36 and 0.46.

The simulations conducted for this analysis suggest significantly different population responses to exploitation when natural mortality is modelled as a function of stock biomass rather than as a constant rate. In particular, our results suggest that a threshold spawning stock biomass may be required to avoid stock collapse at exploitation rates as low as 0.1, although some of the stocks appear to be resilient to collapse without a MSSB at exploitation rates up to 0.3. The exploitation rate which provided MSY was also variable among stocks and ranged from 0.2 to 0.4. Although the MSY exploitation rate varied, for all stocks there was little gain in average yield and a substantial increase in the variance of the yield at harvest rates above 0.2. These results suggest that the current harvesting strategy for British Columbia herring stocks, a 20% exploitation rate in conjunction with a MSSB, may be less conservative than previously thought.

The harvest simulations presented in this analysis are not intended to fully evaluate alternative harvesting strategies for British Columbia herring fisheries, but rather to assess the potential impact of density-dependent natural mortality on the stocks responses to exploitation. A more thorough study would need to consider alternative, more realistic, assumptions regarding the factors affecting recruitment. For example, studies have shown that the herring recruitment in both northern and southern British Columbia are related to the abundance of a key predator early in the life history (Walters et al. 1986; Ware and McFarlane 1986). Additionally, alternative harvesting strategies such as variable exploitation rates or constant catch levels should be examined. An evaluation of alternative harvesting strategies requires an explicit statement of management objectives for the stocks, on which to base evaluation criteria. While specific management objectives have not been stated for British Columbia herring stocks there is an intent to formulate these in the near future (Irvine et al. 1991). This will provide the framework for a more comprehensive evaluation of alternative harvest strategies for Pacific herring stocks in British Columbia.

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The Comparative Performance of Production-Model and *ad hoc* Tuned VPA Based Feedback-Control Management Procedures for the Stock of Cape Hake Off the West Coast of South Africa

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Punt, A. E. 1993. The comparative performance of production-model and *ad hoc* tuned VPA based feedback-control management procedures for the stock of Cape hake off the west coast of South Africa. p. 283–299. *In* S. J. Smith, J. J. Hunt and D. Rivard [ed.] Risk evaluation and biological reference points for fisheries management. Can. Spec. Publ. Fish. Aquat. Sci. 120.

South Africa's most valuable fishery is that for Cape hake off the country's west coast. The resource actually consists of two species (*Merluccius capensis* and *M. paradoxus*). These species are morphologically quite similar, so that the catch and effort statistics collected from the fishery have not distinguished between them. It has therefore been customary to assess and manage the resource as if it were a single species. However, applications of standard single-species production-model and VPA methods have been criticized because in reality two species are being exploited. A further matter of concern is that the VPA assessment indicates that the resource is depleted to a much greater extent than is estimated by the production-model. In order to determine the quantitative importance of these reservations in a management context, a Monte Carlo simulation exercise has been conducted. This allows the different management procedures to be compared in terms of their performance over a 20-year period under feedback-control. The results show that treating the two species as one for management purposes need not compromise either utilization or resource conservation objectives. In general, management procedures based on production-model assessments are found to perform more satisfactorily than those based on *ad hoc* tuned VPA primarily because the latter exhibit substantially larger inter-annual catch limit variability.

La ressource halieutique la plus précieuse de l'Afrique du Sud est le merlu du Cap, qui est pêché sur la côte ouest de ce pays. Cette ressource se compose en fait de deux espèces (*Merluccius capensis et M. paradoxus*). Les deux espèces sont très semblables morphologiquement, de sorte qu'on ne les distingue pas dans les statistiques sur les prises et l'effort de pêche. On a donc pris l'habitude d'évaluer et de gérer la ressource comme s'il s'agissait d'une seule espèce. Toutefois, étant donné que deux espèces sont exploitées, des critiques ont été émises à l'égard des applications de méthodes standards mono-spécifiques de modèles de la production et d'APV. Autre sujet d'inquiétude, l'évaluation fournie par l'APV indique que l'affaiblissement de la ressource est beaucoup plus grand que ne l'estimait le modèle de la production. Pour déterminer l'importance quantitative des réserves qui ont été formulées dans un contexte de gestion, on a procédé à l'exercice de simulation de Monte-Carlo. Cet exercice permet de comparer les différentes méthodes de gestion selon leur performance sur une période de 20 ans avec contrôle rétroactif. Les résultats montrent que le fait de traiter les deux espèces comme une seule aux fins de la gestion ne nuit pas nécessairement aux objectifs d'exploitation ou de conservation des ressources. En général, les méthodes de gestion fondées sur les évaluations selon le modèle de la production donnent des résultats plus satisfaisants que celles qui se fondent sur une APV spécialement ajustée, car cette dernière présente une variabilité nettement supérieure en termes de limites des prises d'une année à l'autre.

Cape hake form the basis of the South African demersal fishery (Crawford et al. 1987), contributing up to 70% of the catch by demersal vessels (Botha 1970), and constitute South Africa's most valuable renewable marine resource. Because of this dominant role, it is vital for the continued economic success of the demersal fishing industry that the hake resource be managed on a sound basis.

The fishery for hake off the west coast of South Africa (29°S–35°S latitude) commenced in the early part of this century, catches by the local industry increasing rapidly after the Second World War (Fig. 1). During the 1960's, distant-water fleets from Japan, Spain, the Soviet Union, Poland, Bulgaria, Romania and the German Democratic Republic entered the fishery for hake in the southeast Atlantic. As a result, catches increased substantially, but catch rates plummeted (Fig. 1). Total allowable catches (TAC's) for the west coast hake fishery for the period 1978–1989 were set by the South African authorities at levels which were deliberately less than the $f_{0.1}$ harvesting strategy TAC's suggested by production-model analyses (the $f_{0.1}$ harvesting strategy involves setting the level of fishing effort to that at which the slope of the sustainable

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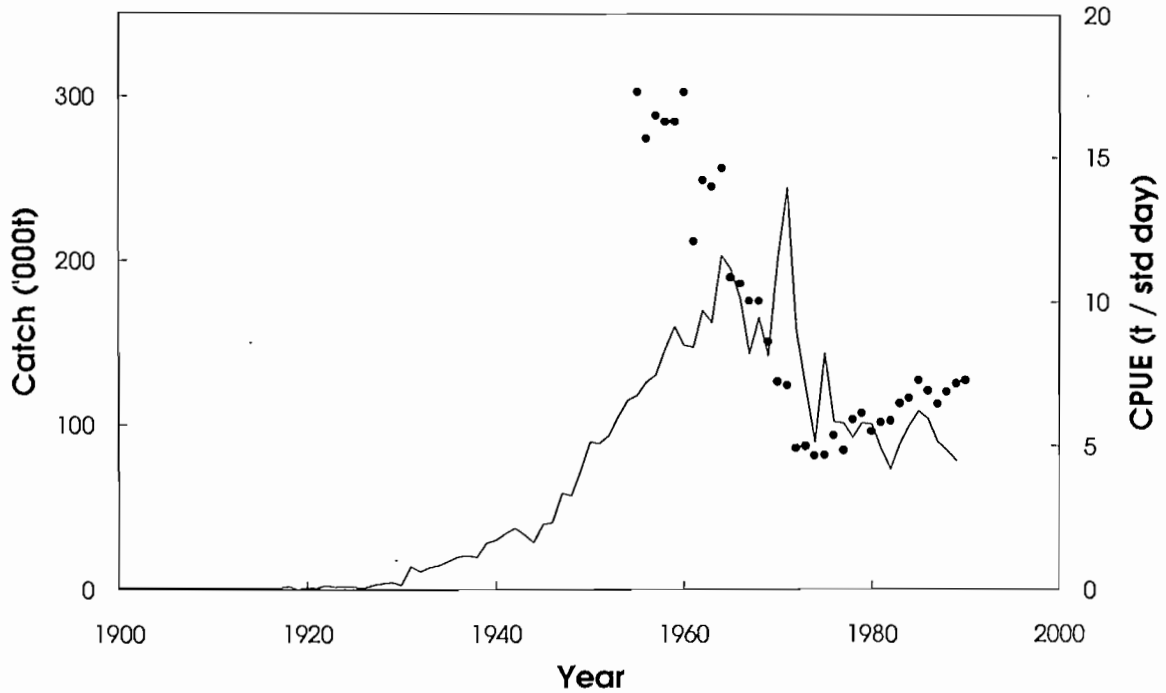


Fig. 1. Catch (1917–1989) and catch rate (1955–1989) history for the hake fishery off the west coast of South Africa.

yield versus fishing effort curve is 10% of the slope at the origin). This was to allow increases in the stocks and in the catch rates at a faster rate than was possible under the $f_{0.1}$ strategy (Payne 1989). As of 1990, scientific recommendations for TAC's for this stock have been based on $f_{0.2}$ harvesting strategy TAC's obtained from a production-model, the Butterworth-Andrew ($B_{1917} = K$; Schaefer form) observation error estimator (Butterworth and Andrew 1984; Punt and Butterworth 1991). As the production-model ignores the potentially valuable information contained in the age-structure of the catches, an *ad hoc* tuned VPA based on the Laurec-Shepherd tuning algorithm (Pope and Shepherd 1985) is also applied routinely. However, the results obtained from this latter approach are not currently used to provide TAC recommendations.

Both of these assessment approaches can be criticized because the hake resource off the South African west coast actually consists of two different species (*Merluccius capensis* and *M. paradoxus*) (Botha 1980; Payne 1989). Note that it is not possible to assess and manage these two hake species separately, because catch and effort statistics do not distinguish between them; their morphological similarity makes differentiation impractical during routine sampling. Other concerns with the current management approach are that the population dynamics equation implicit in the production-model is too simple to capture the dynamics of the resource because *inter alia*, it ignores age-structure, and that the VPA assessment indicates the resource to be smaller (Fig. 2), less productive and currently more depleted than the production-model assessment. [The VPA assessment gives MSY and depletion estimates of 107,000 tons and $0.19K$, while the production-model estimates 138,000 tons and $0.42K$ respectively, where K is the pre-exploitation abundance.]

As the TAC recommendations for the hake resource are derived in a feedback-control manner (i.e., the model parameters are updated annually as more data become available), it is not appropriate to consider the properties of alternative management procedures without allowing for "learning" effects. This has been investigated by means of a Monte Carlo simulation exercise, which is used to examine the likely qualitative consequences of the reservations mentioned above.

The Testing Approach Utilized

The performance of a management procedure has been determined by means of Monte Carlo simulation (see Fig. 3). Each simulation involved applying a candidate management procedure to data generated by a model of the resource for a twenty-year period. For each simulation, the historic catch data are unchanged and identical to past observations (Fig. 1). However, future (1990+) catches can vary because of stochastic effects — both recruitment variability and noise in past and future effort and catch-at-age data. Four age-structured models of the dynamics of the resource (operating models) have been constructed. Two of these models consider only a single species while the other two consider two species. One set of models has been parameterized to reflect the situation indicated by the production-model while the other set has been parameterized to reflect the situation indicated by the VPA. The single species models have been considered to examine whether substantial losses in performance arise because single species assessment approaches are applied to a situation in which the resource actually consists of two species. The two alternative parameterizations are considered to cover

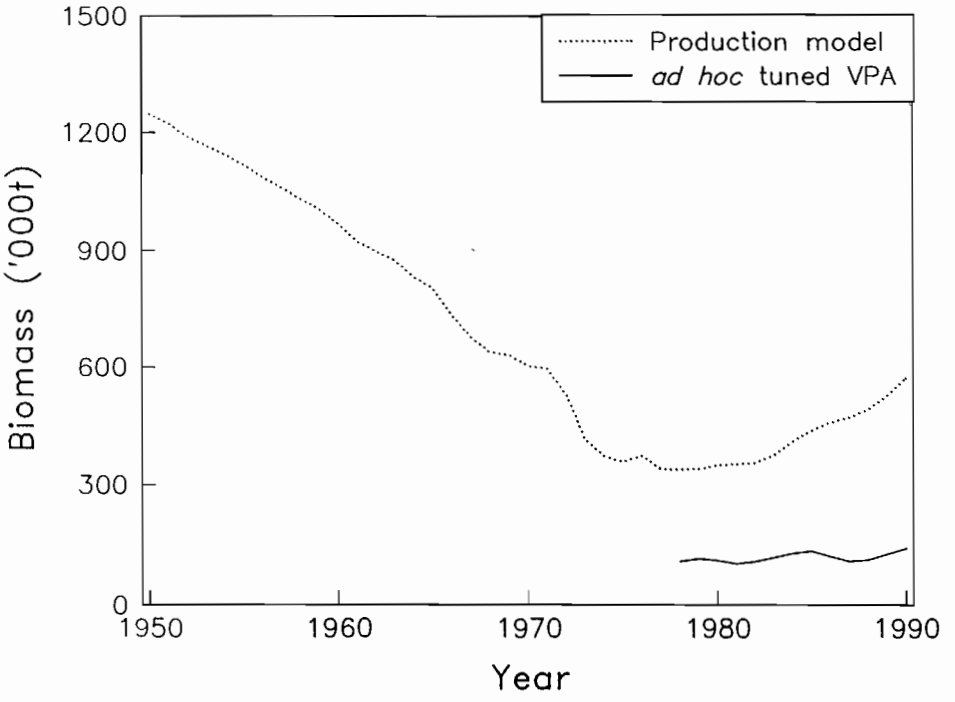


Fig. 2. Exploitable biomass trajectories for west coast hake obtained from the *ad hoc* tuned VPA (solid line) and the production-model (dotted line) assessments.

the likely range of possible scenarios for the fishery adequately, and to address the concern mentioned above that the VPA assessment is less optimistic than the production-model assessment. The details of these models are given in the Appendix.

The three main objectives which are generally identified by managers are continuing high total catch, low risk of unintended depletion and low inter-annual catch limit fluctuations. The four measures which have been chosen in this study to quantify these management objectives are:

a. Total catch over a 20-year period of management: $C = \sum_{y=1990}^{2009} C_y$.

b. Depletion at the end of the 20 years: $\left(B^e_{2009}/K^e\right)$.

c. Lowest depletion during the 20 year period: $(B/K)^e_{\min} = \min\left(B^e_y/K^e : y = 1990, \ldots, 2009\right)$.

d. Variability of catch limits, expressed as a proportion of the average catch and in percentage terms:

$$V = 100\frac{\sum_{y=1990}^{2009}\left|C_y - C_{y-1}\right|}{\sum_{y=1990}^{2009} C_y}$$

where[1]

B^e_y is the exploitable biomass at mid-season during year y, and

K^e is the exploitable component of the average pristine biomass.

The rationale for the choice of a 20-year management period was that this was sufficiently long to ensure that the initial conditions should not influence the final results excessively. The decision not to discount future catches was made primarily to be consistent with ICSEAF's (International Commission for South East Atlantic Fisheries) specification of an operating model and performance measures for management of the hake resource off northern Namibia, for which future catches were not discounted (ICSEAF was the international body which regulated the hake fishery off the South African west coast before the declaration of a 200 nm EEZ by South Africa in 1978). In addition, with a rapidly increasing population size, southern Africa's demand for protein is likely to increase markedly in the next 20 years. It thus seems socially inappropriate to give greater weight to immediate compared to future catches. This suggests a discount rate near zero. The selection of a 20-year management period is equivalent to setting the discount rate to infinity after 20 years. The final depletion statistic was added to ensure that no management procedure could appear to perform "well" by achieving a high total catch by exterminating the resource in the final (20th) year, as would be appropriate if the discount rate was indeed

[1]In the remainder of this paper B^e_y and K^e will be denoted B_y and K respectively.

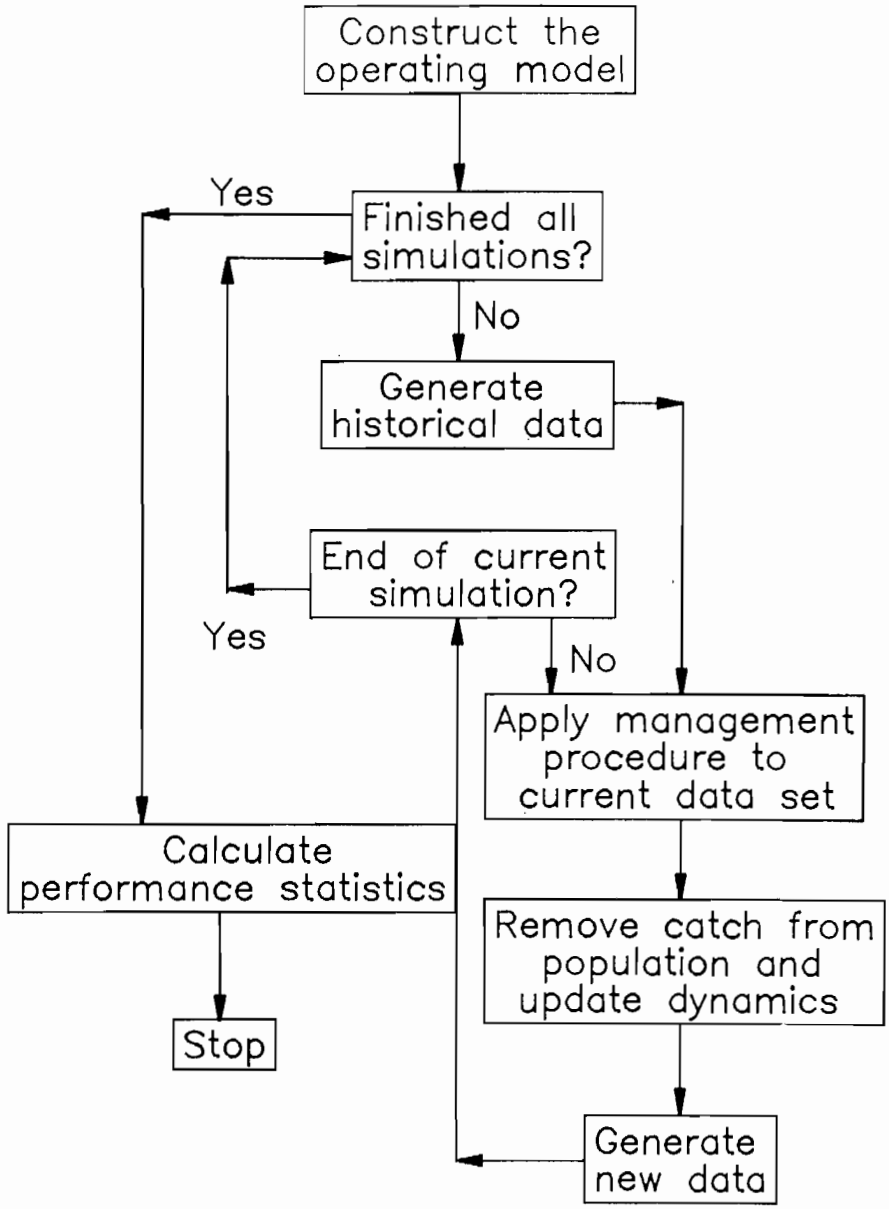


Fig. 3. Flowchart of the approach used to assess the performance of a management procedure.

infinite after the end of the 20-year period (i.e., it should still be possible to take high catches after the 20-year period). Were there to be depensatory mechanisms, the results for procedures which drive the resource to a very low level and then allow it to recover would be over optimistic. While there is no evidence to suggest that such mechanisms exist for this hake population as it has never been driven to a very low level, it is also not possible to exclude the possibility that they exist. The lowest depletion measure can be inspected to assess whether the population is ever reduced below the lowest depletion experienced thus far and hence driven into a region in which depensatory factors (if they do exist) may begin to play a role. The measure used to quantify inter-annual catch limit variability was selected because it can be interpreted easily (it is the average annual percentage change in the catch over the 20-year period).

In standard multiple-criteria decision making theory [e.g., Keeney and Raiffa (1976)], the best management procedure will be that which maximizes some function (the utility function) of the summary statistics chosen. However, a number of problems arise when attempting to define a utility function in this instance. In particular, there are difficulties associated with the time required to construct such a function (which may require considerable input from decision makers), and with the problems associated with making value judgments in a context-free environment (Stewart et al. 1990). For these reasons, even though the use of a utility function is the preferred approach in principle, no attempt will be made here to define such a function. Instead the overall performance of a management procedure will be judged by the range of trade-offs it achieves for each of the four operating models. In order to ease the comparison of the performance statistics of the set

of management procedures considered, a comparison plot is utilized. This plot involves showing the median and 90% limits for the total catch and final depletion distributions, the 5%, 10% and 50% distribution points of the lowest depletion distribution, and the median V statistic.

Management Procedures Considered

Ten alternative management procedures have been considered in this paper. Eight of these procedures are based on a observation-error production-model estimator. This estimator assumes that the biomass at the start of 1917 was equal to the average pristine level, that the biomass dynamics are deterministic, and utilizes CPUE and biomass survey data as relative indices of abundance. A full description of the estimator and an application of it to data for Cape hake is given in Andrew et al. (1989). Results have been calculated for variants of this estimator which incorporate the Schaefer (1954, 1957) and Fox (1970) forms of the surplus production function. The harvesting strategies considered in conjunction with the production-model estimators are $f_{0.1}$, $f_{0.2}$, $\mathrm{MAC}_{0.1}$ and $\mathrm{MAC}_{0.2}$. The $\mathrm{MAC}_{0.n}$ (Maximum Allowable Catch) strategy differs from the $f_{0.n}$ strategy in that when the current biomass exceeds the target biomass, the TAC is set equal to the equilibrium catch under the $f_{0.n}$ strategy (Butterworth 1987). The harvesting strategies considered in conjunction with the Laurec-Shepherd VPA assessment procedure (Pope and Shepherd 1985) are $F_{0.1}$ and $F_{\text{status-quo}}$ (Pope 1983; Pope and Gray 1983). To ease the presentation of the results, acronyms have been used in place of the full descriptions of the management procedures. The acronym for a management procedure is constructed by specifying its estimator (VPA, Schaefer or Fox) and its harvesting strategy. For example, "Schaefer-$f_{0.1}$" is used in place of "the combination of the production-model (Schaefer form) and the $f_{0.1}$ harvesting strategy".

Results and Discussion

Simulations for the Single-Species Operating Models

Figures 4 and 5 contrast the performances of the ten management procedures for the two single-species operating models. From these figures, it is evident that there is a trade-off between total catch and inter-annual catch limit fluctuation, and between total catch and final depletion.

The ability of management procedures to achieve their target levels can be assessed by considering the median final depletion values. For the "production-model" scenario (Fig. 4), the population is left, in more than 50% of cases, above the "true" B_{MSY} ($0.381K$) after the 20 years by all but one management procedure (VPA-$F_{0.1}$), and (slightly) above the "true" $B_{0.1}$ ($0.460K$) by some of them. For the VPA scenario (Fig. 5), none of the management procedures based on the $f_{0.1}$ or $\mathrm{MAC}_{0.1}$ harvesting strategies are able to leave the biomass above the "true" $B_{0.1}$, although most leave it at B_{MSY} or higher. The latter result is a consequence of the fact that, for the "VPA" scenario, the population is initially at a much lower fraction of its average pre-exploitation level. As the production-model based procedures achieve reasonable resource utilization and leave the resource reasonably close to acceptable target levels, it follows that these procedures are not too simple to capture the essential dynamics of the resource.

In terms of medians, the management procedures which are based on the Fox form of the surplus production function leave the resource at a level some 0.02–$0.12K$ higher than do those based on the Schaefer form. This relative behaviour of two forms was unexpected, because the nominal target levels for the Fox form are lower (for example, MSY for the Fox form occurs at $0.37K$ while it occurs at $0.5K$ for the Schaefer form). In addition (and partly in consequence), the procedures based on the Fox form also have lower 5% distribution points of the lowest depletion distributions (5%P_{low}'s) which are higher than those for the Schaefer form.

All of the management procedures, except VPA-$F_{0.1}$, achieve median lowest depletions (median P_{low}'s) which are larger than $0.3K$ for the "production-model" scenario. Thus, if this assessment does reflect the actual situation in the fishery, all but the VPA-$F_{0.1}$ procedure are effective in preventing severe unintended depletion. Further, for this scenario, only four of the procedures result in a 5%P_{low} smaller than $0.2K$ (notably, this number includes VPA-$F_{\text{status-quo}}$). For the "VPA" scenario, the production-model based procedures achieve median P_{low}'s and 5%P_{low}'s which are almost identical to the median and lower 5% distribution point of the distribution of the depletion at the middle of 1989. This means that these procedures are able to prevent further depletion of the resource if the VPA assessment reflects the actual situation in the fishery. The results for the two VPA based procedures are far less encouraging, because, for both scenarios, depletion to below $0.1K$ occurs in almost 30% of simulations.

The relative performances of the production-model and VPA based approaches can be assessed by comparing the results of these procedures in Fig. 4 and 5. The catch limit variability (V) statistics for the VPA based procedures are all substantially larger than those for the corresponding production-model procedures (often by as much as an order of magnitude). The reasons for this high inter-annual catch limit variability are discussed below. The final depletion and total catch distributions for the VPA based procedures are also wider (often substantially so) than those for the production-model based procedures. Given their occasional very poor performance in terms of conserving the resource (which often results from occasional outlying TAC's), it would have been expected that the VPA based procedures might have achieved much higher total catches than the production-model procedures. However, this is not the case — the differences in total catch between the VPA and the production-model procedures are seldom as large as they are in final depletion. This is a consequence of the fact that the production-model based procedures keep the resource closer to B_{MSY} than the VPA-based procedures. They can therefore utilize the enhanced surplus production available at this level of biomass.

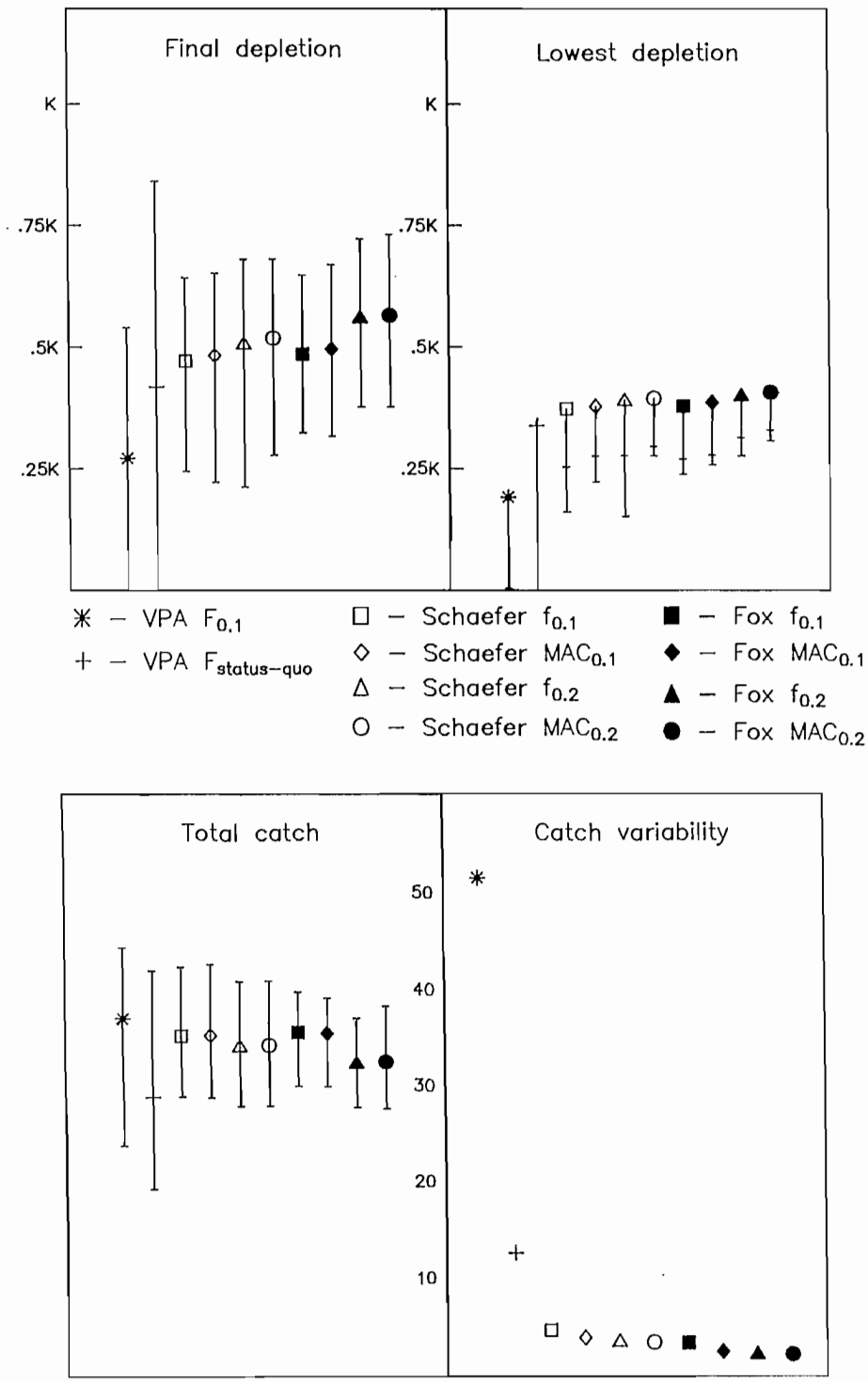


Fig. 4. Comparison of the performance of the ten management procedures for the "production-model" single-species operating model. The symbol indicates the median of the distribution. For the total catch and final depletion distributions, the ticks indicate the upper and lower fifth percentiles. For the lowest depletion distribution, the ticks indicate the lower fifth and tenth percentiles.

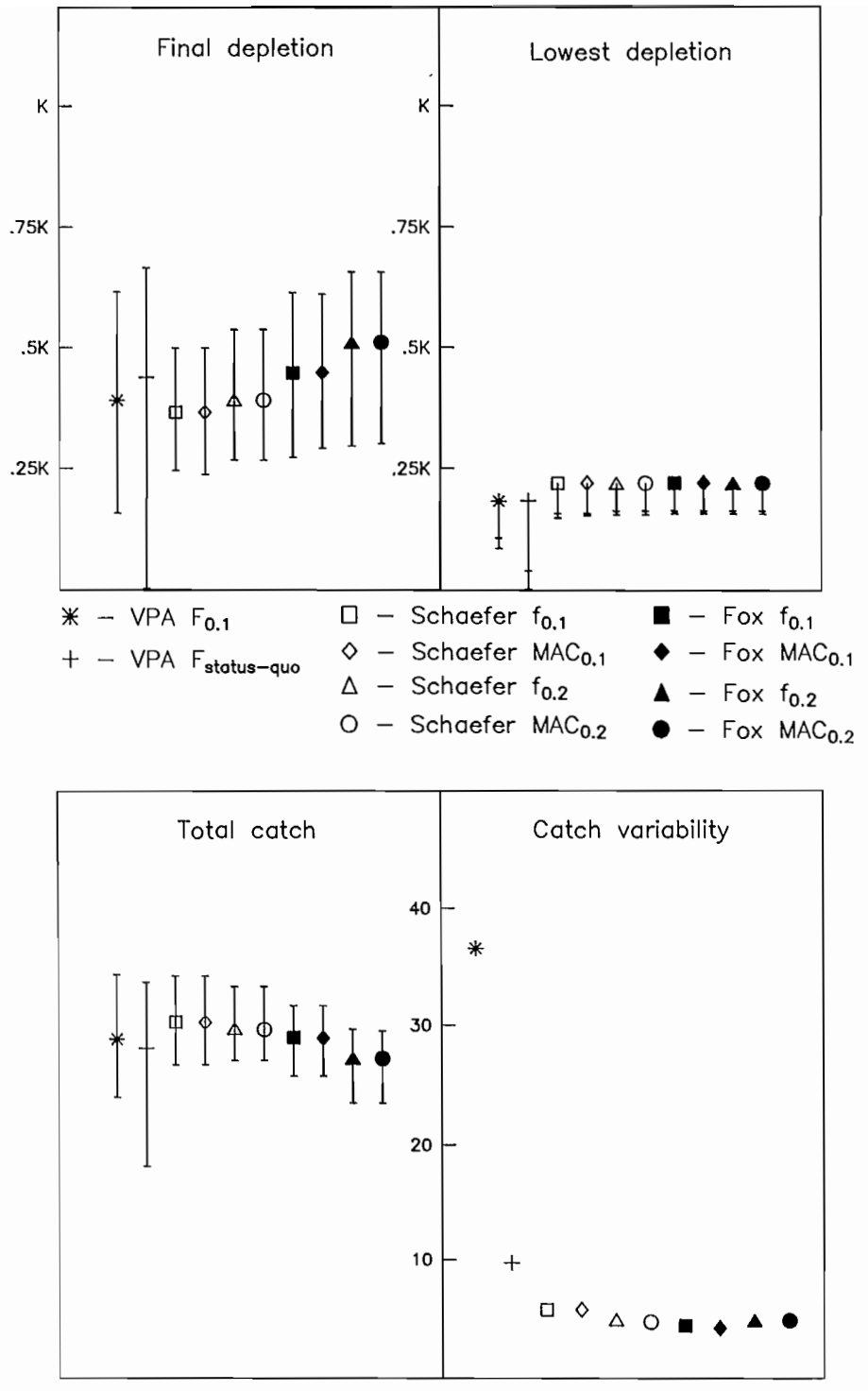


Fig. 5. Comparison of the performance of the ten management procedures for the "VPA" single-species operating model. The symbol indicates the median of the distribution. For the total catch and final depletion distributions, the ticks indicate the upper and lower fifth percentiles. For the lowest depletion distribution, the ticks indicate the lower fifth and tenth percentiles.

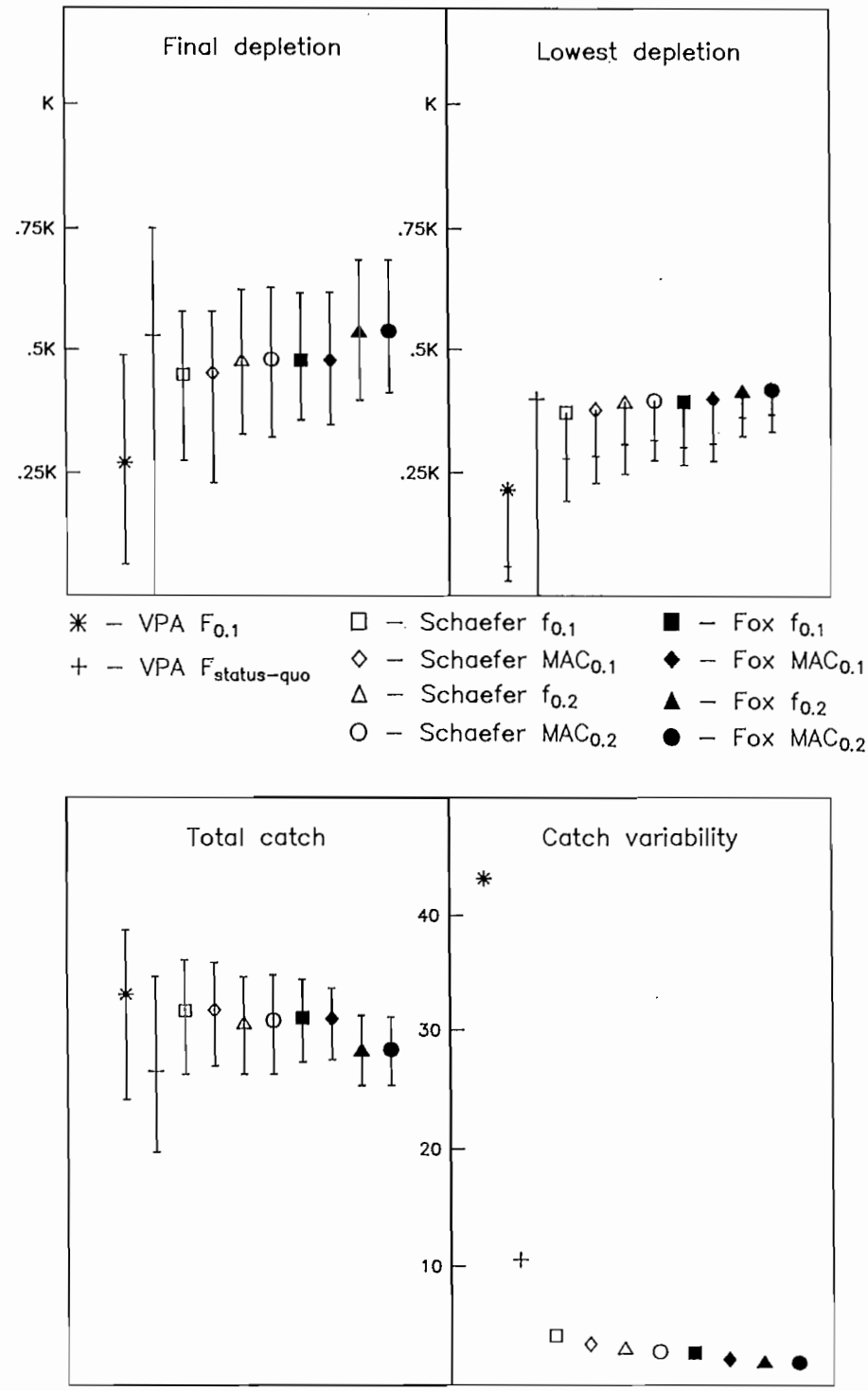


Fig. 6. Comparison of the performance of the ten management procedures for the "production-model" two species operating model. The symbol indicates the median of the distribution. For the total catch and final depletion distributions, the ticks indicate the upper and lower fifth percentiles. For the lowest depletion distribution, the ticks indicate the lower fifth and tenth percentiles.

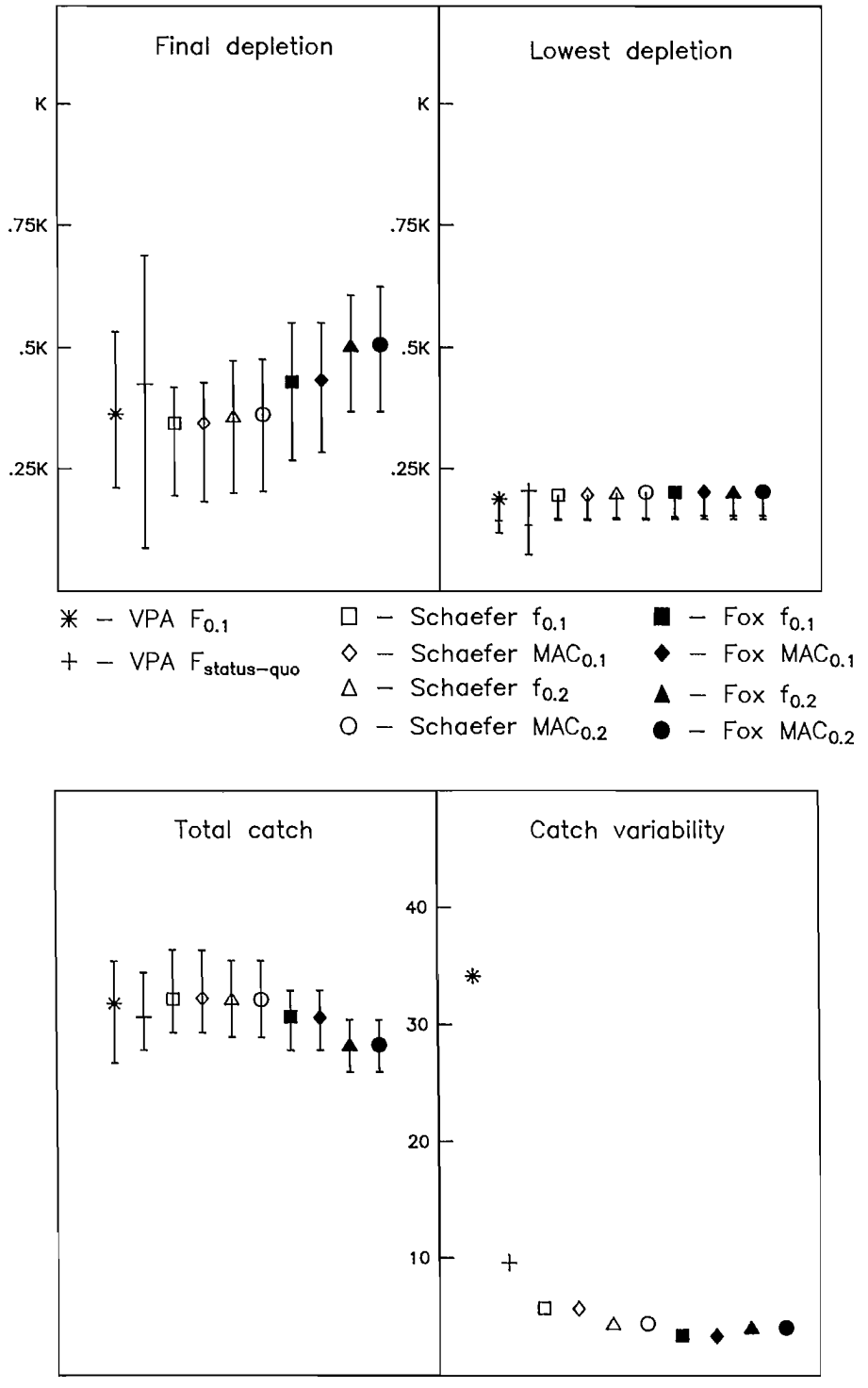


Fig. 7. Comparison of the performance of the ten management procedures for the "VPA" two species operating model. The symbol indicates the median of the distribution. For the total catch and final depletion distributions, the ticks indicate the upper and lower fifth percentiles. For the lowest depletion distribution, the ticks indicate the lower fifth and tenth percentiles.

The Two-Species Operating Models

The results for a two-species underlying situation are shown in Fig. 6 and 7. Comparing these with Fig. 4 and 5 shows that, in general, the performances of the management procedures for the single- and two-species operating models are not qualitatively different. Note that the results for the single- and two-species operating models are not precisely comparable, because the values for the stock-recruitment parameters differ for the two models. This results in the two models corresponding to slightly different MSY's and having slightly different initial (1989) depletion distributions.

There are, however, some notable differences between the results for the two- and single-species operating models. For example, the results for the former are less variable than those for the latter. This is reflected in narrower total catch and final depletion distributions, in higher $5\%P_{low}$'s and in lower inter-annual catch limit variability. A possible reason for this is that the effective recruitment variability for the two-species operating models is lower because the recruitment residuals generated for *M. capensis* and *M. paradoxus* are uncorrelated.

Thus, Fig. 4 to 7 reveal that managing the two hake species as a single species does not lead to any marked deterioration in performance compared to the single-species situation, both in terms of resource utilization and in terms of conservation. This result was examined for a wider range of operating models by Punt (1991a) and was found to be quite robust.

Aspects to Consider When Selecting a "Best" Management Procedure

It appears that the production-model based management procedures perform better than those based on VPA, even if the VPA assessment of the resource reflects the actual situation. Therefore, only production-model based procedures will be considered in this discussion.

The selection between the eight remaining procedures is problematic because they all achieve slightly different trade-offs. Nevertheless, certain key differences are apparent. The performance statistics for procedures based on the Schaefer form are more variable than the corresponding statistics for the Fox form, so that the Fox form based procedures involve less risk. However, selection of a Fox form based procedure would necessitate a loss in total catch. Another general result is that procedures based on $MAC_{0.n}$ strategies exhibit less risk and achieve lower inter-annual catch limit variability than those based on $f_{0.n}$ strategies, even though there is almost no difference in total catch. This suggests that $MAC_{0.n}$ strategies are to be preferred to $f_{0.n}$ strategies. Note, however, that a choice between the $MAC_{0.n}$ and the $f_{0.n}$ strategies is not an immediate concern for this hake resource because the biomass is assessed to be well below both $B_{0.1}$ and $B_{0.2}$.

Why the VPA Based Procedures Perform so Poorly

On the basis of simulation studies, Pope and Shepherd (1985) concluded that if there are no systematic changes in catchability, all the age-structured assessment methods which they tested worked reasonably well, even in the presence of substantial noise in the effort and catch-at-age data (CV's $\approx$ 0.5). ICES (1988) came to essentially the same conclusion. Pope and Shepherd (1985) were concerned with one-time estimation, whereas this paper is concerned with the longer-term implications under feedback-control, so that their conclusions do not necessarily imply that this study should show *ad hoc* tuned VPA's to be adequate for management purposes. It is nevertheless most surprising that the VPA based management procedures turn out to be so inferior compared to the production-model based procedures.

In order to contrast the performances of the *ad hoc* tuned VPA based management procedures with those based on production-models, "true" and estimated mid-year exploitable biomass trajectories have been plotted for two management procedures for the first two stochastic simulations of four variants of the single-species "production-model" operating model. These plots are given in Fig. 8. The estimate of the mid-year exploitable biomass during year y is obtained from the application of the estimator to data for the years up to and including year y. The two management procedures considered are: (a) VPA-$F_{0.1}$ and (b) Schaefer-$f_{0.1}$. The four variants of the operating model considered are:

i. $\sigma_s = 0.2, \sigma_q = 0, \sigma_c = 0$;

ii. $\sigma_s = 0.2, \sigma_q = 0, \sigma_c = 0.4$;

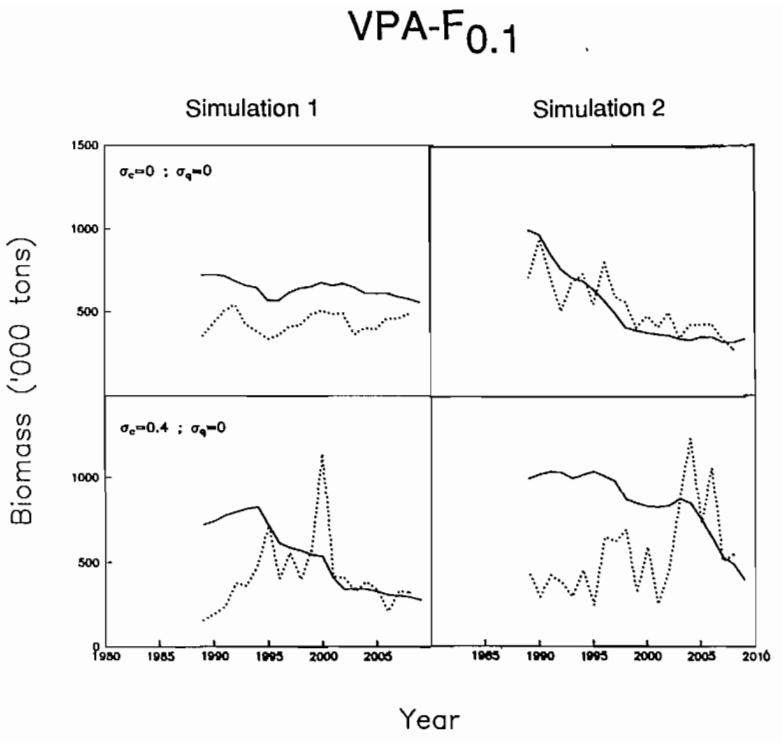
iii. $\sigma_s = 0.2, \sigma_q = 0.16, \sigma_c = 0$; and

iv. $\sigma_s = 0.2, \sigma_q = 0.16, \sigma_c = 0.4$.

(Recall that σ_s, σ_q and σ_c reflect the extent of the variability in the selectivity-at-age, catchability and catch-at-age data respectively.) Trajectories for the Schaefer-$f_{0.1}$ procedure are provided for operating models (i) and (iii) only, because this procedure does not make use of catch-at-age data and hence is unaffected by the size of σ_c.

Error Free Data

Even when the catch-at-age data are exact and there are no inter-annual fluctuations in catchability (corresponding to "exact" effort data, i.e., operating model (i): $\sigma_s = 0.2, \sigma_q = 0$, $\sigma_c = 0$), the catch trajectories for the VPA based procedures exhibit quite marked fluctuations while the production-model catch trajectories for operating model (i) are far less variable. This must be a consequence of the fluctuations in selectivity-at-age, because (given deterministic data) VPA can take account of fluctuations in recruitment exactly. The estimates of exploitable biomass provided by the production-model-estimation procedure are very different from the "true" values, even when the data are generated without error (this is particularly the case for simulation 2, see Fig. 8b) although the estimates of the depletion (i.e., B_y/K) are generally less in error. In contrast, the *ad hoc* tuned VPA estimates of exploitable biomass are rather closer to the true values (particularly for simulation 2). When the data are generated without error, only one of the management procedures depletes the resource to a low level (VPA-$F_{0.1}$ in simulation 2) — this is probably a consequence of this procedure making no allowance for a stock-recruitment relationship when estimating $F_{0.1}$ and predicting future recruitment.



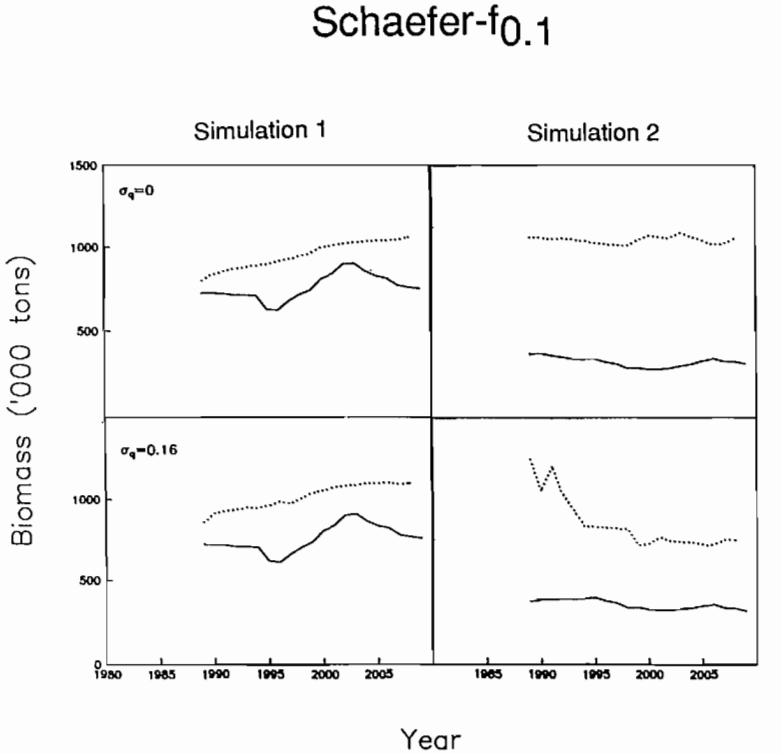


Fig. 8. Estimated (dotted lines) and "true" (operating model — solid lines) mid-year exploitable biomass trajectories obtained by applying a management procedure to two simulations for each of four variants of the single-species "production-model" operating model. The variants are indicated on the plots. Results are shown for the following two procedures: (a) VPA-$F_{0.1}$, and (b) Schaefer-$f_{0.1}$.

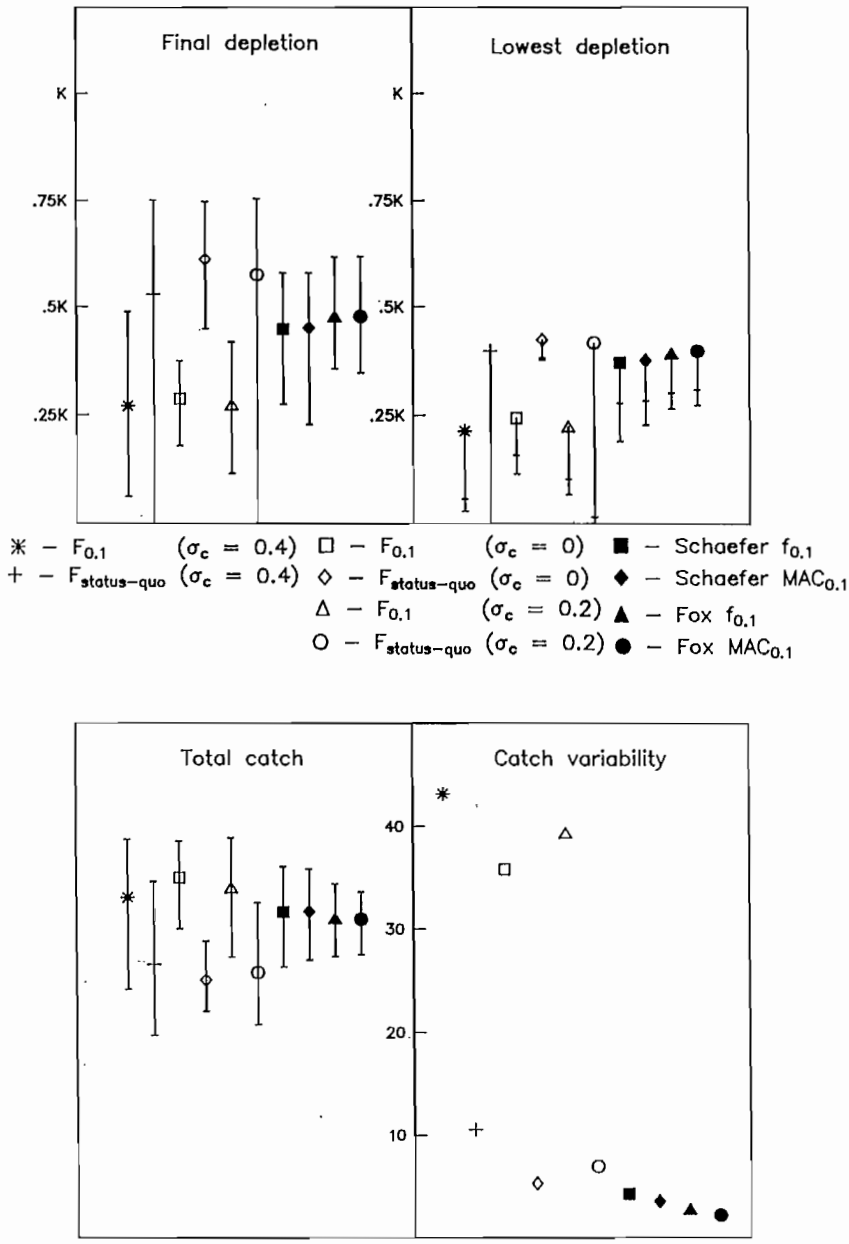


Fig. 9. Comparison of the performance of ten management procedures for the "production-model" two-species operating model. Results for the VPA-based procedures are shown for three choices for the extent of catch-at-age variability (σ_c). The symbol indicates the median of the distribution. For the total catch and final depletion distributions, the ticks indicate the upper and lower fifth percentiles. For the lowest depletion distribution, the ticks indicate the lower fifth and tenth percentiles.

Data With Error

Introducing error into the fishing mortality-fishing effort relationship [$\sigma_q = 0.16$ — operating models (iii) and (iv)], serves to increase the inter-annual limit catch variability. While the effect of these catchability fluctuations on the production-model based procedure is almost negligible, this is not the case for *ad hoc* tuned VPA based procedure. The assessments (and hence the TAC's) based on the VPA are sensitive to catchability fluctuations because the VPA estimator assumes that the reported effort for the most-recent-year is exact. These fluctuations result in the magnitude of the entire biomass time series varying from one assessment to the next — this phenomenon has been observed when applying VPA to actual data for the hake resource off the South African south coast. The production-model based management procedures do not make this assumption, and in consequence their performance appears to be far more robust to fluctuations about the CPUE-abundance relationship.

Introducing error in the catch-at-age data [operating models (ii) and (iv)] increases the size of the fluctuations in the estimates of exploitable biomass markedly for the VPA based management procedures. This, in turn, leads to substantial

increases in inter-annual catch limit variability. It may therefore be that the major cause of the poor performance of the two VPA based procedures is that the extent of variability associated with the catch-at-age data is unrealistically large, and that if it were to be reduced, the performances of the VPA based approaches would improve markedly. In order to examine whether this is the case, Fig. 9 provides the performance statistics which result from simulations for three different values of σ_c (0, 0.2 and 0.4, respectively). The operating model in this case is the "production-model" two-species model. Reducing the extent of catch-at-age error to zero leads to a reduction in inter-annual catch limit variability. However, the VPA-$F_{0.1}$ procedure still leaves the resource below B_{MSY} in over 50% of cases, and again achieves unacceptably high inter-annual catch limit variability. The performance of the VPA-$F_{status\text{-}quo}$ procedure deteriorates noticeably in terms of resource conservation when σ_c is increased from 0 to 0.2. The extent of performance degradation for this procedure as σ_c is increased from 0.2 to 0.4 is far less marked.

Concluding Comments

The results obtained in this paper suggest that the criticisms that the production-model approaches are oversimplified and may fail as they do not take account of the two-species nature of the hake resource off the South African west coast, are unfounded. This paper shows that some production-model approaches which treat the two species as one for management performances do not lead to poor utilization or an inability to achieve conservation objectives. On the other hand, approaches based on VPA perform poorly as they achieve very high inter-annual catch limit variability, even if the resource consists of only one species. Of the production-model approaches considered, those based on $MAC0.n$ harvesting strategies appear to be superior to those based on $f_{0.n}$ strategies, and those based on the Fox form of the surplus production function are less likely to deplete the resource to undesirably low levels than those based on the Schaefer form. The results reported in this paper have been obtained for a specific set of growth/mortality parameters, a particular exploitation history and one set of noise parameters. Certainly, the somewhat controversial conclusion of the paper that production-models are to be preferred to VPA might change if different stocks were used to parameterize the operating model. Therefore, until further analyses are conducted, this result should be considered to be stock-specific. Nevertheless, the results do indicate that seemingly oversimplified approaches such as production-models should not be ignored when selecting management procedures, as they may prove to be the most satisfactory in some cases. The management procedures considered in this paper are not exhaustive (Punt (1991a) considers a much larger number, though those discussed here number among the best of these). Another reason for their selection is that these are the particular procedures under consideration for use in the management of the South African west coast resource, which is the subject of this paper. It is possible (in fact probable) that by merging features of one approach with those of another, performance better than that achieved by either can be obtained. One change which might lead to substantial improvements in performance would be to place bounds on the percentage by which a catch limit can change from one year to the next. This may improve the performance of the VPA based procedures substantially, as currently these show considerable inter-annual catch limit fluctuations without apparent compensatory performance gains in other respects.

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Appendix: The Operating Models

The operating model specified below is age-structured, considers both hake species, relates recruitment to spawner-stock size, and includes a number of error terms. This model will be described first, followed by the process of generating the "observed" data used for assessment purposes.

The Model

Basic Dynamics:

$$N^s_{y+1,a+1} = N^s_{y,a} e^{-Z^s_{y,a}} \quad \text{(A1)}$$

where

$N^s_{y,a}$ is the number of fish of species s aged a years at the start of year y,

$Z^s_{y,a}$ is the total mortality on fish of species s aged a years during year y,

$a = 0, 1, 2, \ldots, a_{\max}(= 9)$, and

$y = 1917, 1918, , \ldots, n$.

Total Mortality:

$$Z^s_{y,a} = M + F^s_{y,a} \quad \text{(A2)}$$

where

M is natural mortality (assumed independent of age and year, and taken to be 0.3 yr^{-1}),

$F^s_{y,a}$ is the fishing mortality on fish of species s aged a years during year y:

$$F^s_{y,a} = \tilde{S}^s_{y,a} F_y \quad \text{(A3)}$$

F_y is the fully-selected fishing mortality during year y (fishing mortality on fish for which $\tilde{S}^s_{y,a} \to 1$), and

$\tilde{S}^s_{y,a}$ is age-specific selectivity for species s in year y ($S^s_{y,a}$ is the deterministic age-specific selectivity function for species s — $\tilde{S}^s_{y,a}$ includes a stochastic component — see Equation A8).

Exploitable Biomass:

The exploitable biomass at the start of year y is defined by:

$$\tilde{B}_y = \sum_s \sum_a w^s_a S^s_{y,a} N^s_{y,a} \quad \text{(A4)}$$

where w^s_a is the mass of a fish of species s aged a years, at the start of the year:

$$w^s_a = 0.5\left(w^s_{a+1/2} + w^s_{a-1/2}\right)$$

and $w^s_{a+1/2}$ is the mass of a fish of species s aged a years, in the middle of the year (see Table A1).

The exploitable biomass in the middle of year y is defined by:

$$B^e_y = \sum_s \sum_a w^s_{a+1/2} S^s_{y,a} N^s_{y,a} e^{-Z^s_{y,a}/2} \quad \text{(A5)}$$

Table A1. The mean mass in mid-year, of fish aged $a + 1/2$ years (i.e., $w_{a+1/2}$). Units are kg. Source: Punt and Leslie (1991).

AGE	*M. capensis*	*M. paradoxus*	Both species
0	0.016	0.013	0.013
1	0.068	0.058	0.061
2	0.196	0.182	0.190
3	0.454	0.440	0.452
4	0.853	0.795	0.827
5	1.406	1.267	1.341
6	2.124	1.886	2.005
7	2.988	2.614	2.785
8	4.141	3.345	3.694
9	5.280	4.228	4.735

Births:

$$\text{(A6)} \quad N^s_{y,0} = \frac{e^{\varepsilon^s_{y,r} - \sigma^s_r/2}\alpha B^s_y}{\beta + B^s_y}, \quad \varepsilon^s_{y,r} \sim \mathrm{N}\left(0, \sigma^2_r\right)$$

$$B^s_y = \sum_a f^s_a N^s_{y,a}$$

where

- α, β are (Beverton-Holt) stock-recruitment relationship parameters,
- B^s_y is the spawner biomass of species s at the start of year y, and
- f^s_a is the fecundity index of a fish of species s aged a years.

Values for α and β were selected so that the relative increase in the exploitable biomass over the past nine years and the current exploitable biomass each matched specified values. Two alternative scenarios are considered, and provide different operating models. These correspond to matching the exploitable biomass increase rate and current exploitable biomass obtained from the results of the *ad hoc* tuned VPA and production-model assessments of the South African west coast hake resource.

Consideration of the variability of the recruitment estimates obtained from the VPA analyses (Punt 1991b) suggests that σ_r must be less than 0.2. However a value of σ_r of this size seems unrealistically low to describe the likely extent of recruitment variability for a species such as hake, so that a value of 0.3 was chosen for this parameter.

Catch-by-Mass:

$$\text{(A7)} \quad C_y = \sum_a \sum_s \frac{w^s_{a+1/2} F^s_{y,a} N^s_{y,a} \left[1 - \exp\left(-Z^s_{y,a}\right)\right]}{Z^s_{y,a}}$$

where C_y is the catch-by-mass in year y.

Selectivity:

Selectivity as a function of age is given by the formula:

$$\text{(A8)} \quad S^s_{y,a} = \left[1 + e^{-(a - a^s_r)/\delta^s_r}\right]^{-1}$$

$$\tilde{S}^s_{y,a} = S^s_{y,a} e^{\varepsilon_{y,a} - \sigma^2_s/2}, \quad \varepsilon_{y,a} \sim \mathrm{N}\left(0, \sigma^2_s\right)$$

where

- δ^s_r determines the width of the selectivity ogive for species s, and
- a^s_r is the age-at-50%-selectivity for species s.

The values for the selectivity function parameters for the single species operating model are obtained by fitting a logistic curve to selectivities-at-age provided by the *ad hoc* tuned VPA model-estimation procedure (Punt 1991b). The resultant estimates are $a_r = 1.67$ yr and $\delta_r = 0.29$ yr. The corresponding values for the two-species operating model are chosen so that the age-at-50%-selectivity for *M. capensis* (species 1) is larger than that for *M. paradoxus* (species 2) [because the fishery concentrates further offshore than the shallower waters where young *M. capensis* are to be found], and so that the average of the two selectivity curves is similar to the curve for the VPA assessment referenced above. The resultant estimates are $a^1_r = 3$ yr, $\delta^1_r = 0.5$ yr, $a^2_r = 1.5$ yr and $\delta^2_r = 0.25$ yr.

The value chosen for σ_s is 0.2. This selection was made to be consistent with ICSEAF's specification of an operating model for the hake resource off northern Namibia (ICSEAF 1990).

Fecundity

The fecundity index is given by:

$$\text{(A9)} \quad f^s_a = \begin{cases} w^s_a & \text{if } a > a_m; \\ 0 & \text{otherwise.} \end{cases}$$

where a^s_m is the age-at-50%-maturity of species s.

The selection of 4 years for a^s_m was made by converting the length-at-50%-maturity for both species and sexes combined (Punt and Leslie 1991) into an age.

Generation of Data

Setting up the Simulations

The initial conditions for each simulation (for year $y = 1917$) correspond to a resource whose biomass is drawn from the distribution about an average pre-exploitation level which would be expected to result from the assumed level of random recruitment fluctuation. Such a situation would seem to be realistic for the hake resource off the South African west coast, because any catches prior to 1917 would have had a negligible effect on the resource biomass. The numbers-at-age for each of the 100 Monte-Carlo data sets at the start of year $y = 1917$ are thus generated as follows.

1. The numbers-at-age corresponding to deterministic equilibrium are calculated.
2. The resource is then projected forward for 20 more years with no catches, but with stochastically fluctuating recruitment (i.e., $\varepsilon^s_{y,r} \neq 0$; $F_y = 0$).
3. The resultant numbers-at-age after the 20 years are taken to be the numbers-at-age at the start of 1917.

The numbers-at-age at the start of management (1990) are generated by projecting the numbers-at-age at the start of 1917 forward using Equation (A1). If, after this process, the depletion B^e_{1989}/K^e) differs by more than 0.05 from the actual value predicted by the assessment which is being reflected by the operating model — 0.42 and 0.19 for the "production-model" and "VPA" scenarios respectively, the simulation is aborted. This selection criteria was applied so that the initial depletion distribution for a particular scenario (VPA or production-model) did not differ too markedly from the estimate on which it was based. When performing the projections of the operating model, catches from 1917 to 1989 are set equal to their historical values, and those subsequent to 1989 are set to the TAC's estimated by the management procedure under consideration (i.e., TAC's set are assumed always to be taken exactly).

Catch Data

The reported catch-by-mass data (C^{obs}_y) are available for every year from 1917 until year $n-1$ (where n ranges from 1990 to 2009). The reported catches for the years prior to 1954 are subject to log-normally distributed random error with a CV of 5% to make allowance for the errors inherent in the estimation of total catches from logbooks by Chalmers (1976).

Effort/CPUE Data

Effort data are available for all years from 1955 until year $n-1$. The effort in year y, (E^{obs}_y), generated by the operating model is calculated as follows:

$$\text{(A10)} \quad E^{\text{obs}}_y = F_y/q e^{\varepsilon_{y,q}-\sigma^2_q/2}, \ \varepsilon_{y,q} \sim \text{N}\left(0, \sigma^2_q\right)$$

where

- F_y is the fully-selected fishing mortality during year y,
- q is the catchability coefficient, and
- σ_q is the observation error standard deviation — taken to be 0.16. This value was obtained from consideration of the standard deviations of the residuals of the fishing mortality versus fishing effort relationships for the *ad hoc* tuned VPA (≈ 0.2) (Punt 1991b), and the standard deviation of the residuals of the production-model fit to the CPUE data (≈ 0.12) (Punt 1991b).

Survey Biomass Data

Survey biomass estimates of abundance are available twice a year, starting in the middle of 1983. The mid-year survey biomass estimates are generated using the formula:

$$\text{(A11)} \quad \left(\widetilde{B}^e_y\right)^{\text{obs}} = \Omega B^e_y e^{\varepsilon'_{y,u}-\sigma^2_u/2}, \ \varepsilon'_{y,u} \sim \text{N}\left(0, \sigma^2_u\right)$$

and the begin-year survey biomass estimates as follows:

$$\text{(A12)} \quad \widetilde{B}^{\text{obs}}_y = \Omega \widetilde{B}_y e^{\varepsilon''_{y,u}-\sigma^2_u/2}, \ \varepsilon''_{y,u} \sim \text{N}\left(0, \sigma^2_u\right)$$

where

- Ω is the bias of the survey biomass estimates — taken to be 0.5 on the basis of the results of the application of the production-model (Punt 1991b), and
- σ_u is the CV of the survey biomass estimates — taken to be 0.21. This value reflects the combined effects of sampling error (CV's typically of the order 0.15) and other factors which contribute to the precision of the survey biomass indices [e.g., weather, skipper performance (even for a planned survey grid, catch rates obtained depend on the skill of the skipper at the time of setting the net), area coverage, and movement, etc.] — also taken (somewhat arbitrarily) to be 0.15.

Catch-at-Age/Mass-at-Age Data

Catch-at-age and mass-at-age data are generated for every year from 1978 to year $n-1$.

$$\text{(A13)} \quad C^{\text{obs}}_{y,a} = GC_{y,a} e^{\varepsilon_{y,a,c}-\sigma^2_{y,a,c}/2}$$

where

- $C^{\text{obs}}_{y,a}$ is the estimate made of the catch (in number) of fish aged a years during year y,
- C^{obs}_y is the reported catch-by-mass in year y,
- $C_{y,a}$ is the "true" catch (in numbers) of fish aged a years during year y,

$$\varepsilon_{y,a,c} \sim \text{N}\left(0, \sigma^2_{y,a,c}\right)$$

$$G = C^{\text{obs}}_y \Big/ \left(\sum_{a=0}^{a_{\max}} C_{y,a} w_a e^{\varepsilon_{y,a,c}-\sigma^2_{y,a,c}/2}\right)$$

$$\sigma^2_{y,a,c} = \sigma^2_c \left[\left\{\sum_{a'=0}^{a_{\max}} \frac{C_{y,a'}}{(a_{\max}+1)}\right\} / C_{y,a}\right]^{\beta'}$$

Note that $C_{y,a}$ is not known to an assessor, who has to use the $C_{y,a}^{obs}$ data in assessment calculations. Provided β' is positive, this formulation ensures that, in a given year, the variances for catches-at-age which are small will be larger than those for catches-at-age which are large. This mimics the effects of sampling error. The adjustment factor, G, is chosen to ensure that observed values of catch-at-age always correspond to the same fixed reported catch mass. β' has been set to 1.0 and σ_c to 0.4. These choices were made because they correspond to the selections made by ICSEAF (1990) in their specification of the error structure for catch-at-age data. This selection was based on mimicking the levels of variability typical of catch-at-age data for hake off northern Namibia. However, the sampling of hake catches off northern Namibia is known to be poorer than off the west coast of South African (pers. obs.), so the choice $\sigma_c = 0.4$ may be slightly too extreme. Comparison of the extent of real and operating model catch-at-age variability, indicates that the operating model results in values of $\sigma_{y,a,c}$ which are too slightly too large for ages 3+, and slightly too small for ages 1 and 2.

Groundfish Exploitation Rates Based on Life History Parameters *

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The problem considered is how to choose a fixed exploitation rate that will provide a high yield at low risk, when the investigator has no knowledge of the yield curve or the spawner-recruit relationship of the stock. Commonly employed expedients are $F_{0.1}$ and $F = M$, but these have little empirical or theoretical support. Calculations made with a range of life history parameter values typical of demersal fish and a range of realistic spawner-recruit relationships show that yield will be at least 75% of maximum sustainable yield so long as the spawning biomass is maintained in the range of about 20–60% of the unfished level, regardless of the form of the spawner-recruit relationship. A relative spawning biomass in this range can be achieved by choosing a fishing mortality rate that will reduce the spawning biomass *per recruit* to about 35% of the unfished level. This is the level of fishing mortality that maximizes the minimum yield among all of the spawner-recruit relationships considered ("maximin yield" rate, F_{mmy}). The actual value so calculated depends on the relative timing of growth, maturity, and recruitment to the fishery. In practice, F_{mmy} appears to be very close to $F_{0.1}$ except where recruitment and maturity schedules do not coincide.

Cet article traite du choix d'un taux d'exploitation fixe permettant d'obtenir un rendement élevé tout en minimisant les risques quand on ne connaît pas la courbe de rendement ou la relation reproducteurs-recrues du stock. On utilise souvent les valeurs $F_{0.1}$ et $F = M$, mais les fondements théoriques et empiriques de ces solutions sont faibles. Des calculs fondés sur différentes valeurs typiques des paramètres du cycle vital des poissons de fond et sur différentes relations reproducteurs-recrues réalistes montrent que le rendement sera d'au moins 75% du rendement maximal soutenu dans la mesure où la biomasse des reproducteurs est maintenue entre 20 et 60% de celle qui existerait si le stock n'était pas exploité et ce, quelle que soit le type de relation reproducteurs-recrues. On peut obtenir une biomasse relative de reproducteurs de cet ordre en choisissant un taux de mortalité due à la pêche réduisant la biomasse de reproducteurs par recrue à environ 35% de celle existant en l'absence d'exploitation. Parmi toutes les relations reproducteurs-recrues considérées, un tel taux de mortalité due à la pêche est celui qui maximise le rendement minimum (taux de «rendement maximin», F_{mmy}). La valeur réelle ainsi calculée est fonction des relations temporelles entre la croissance, le moment de l'arrivée à maturité et le recrutement de la pêche. En pratique, la valeur de F_{mmy} est très près de celle de $F_{0.1}$, sauf quand le recrutement et l'arrivée à maturité ne coïncide pas.

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A Proposal for a Threshold Stock Size and Maximum Fishing Mortality Rate

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Thompson, G. G. 1993. A proposal for a threshold stock size and maximum fishing mortality rate. p. 303–320. *In* S. J. Smith, J. J. Hunt and D. Rivard [ed.] Risk evaluation and biological reference points for fisheries management. Can. Spec. Publ. Fish. Aquat. Sci. 120.

Any harvest policy that causes a stock to collapse must certainly be viewed as an extreme example of recruitment overfishing. A depensatory generalization of the Beverton-Holt stock-recruitment relationship is used here to develop a set of constraints designed to safeguard against overfishing of this type: 1) a threshold biomass level set at 20% of the pristine level, 2) a maximum fishing mortality rate corresponding to a 30% relative biomass-per-recruit ratio, and 3) an alternative maximum fishing mortality rate (for stocks lacking growth data) set at 80% of the natural mortality rate. Given certain assumptions, the ability of these constraints to prevent stock collapse is independent of the parameter values used in the stock-recruitment relationship. Both a general theoretical evaluation and a comparison with actual fishing mortality rates applied to 22 North Pacific groundfish stocks indicate that the constraints should not impose new restrictions on fisheries that are already managed for maximum sustainable yield, even when the stock-recruitment relationship deviates from the form used to generate the constraints.

Toute politique de capture qui provoque l'effondrement d'un stock doit certainement être considérée comme un exemple extrême de surpêche du recrutement. Une généralisation anticompensatoire de la relation stock-recrutement de Beverton-Holt est utilisée ici pour élaborer une série de contraintes conçues pour prévenir une surpêche de cette nature : 1) un seuil de biomasse fixé à 20 % du niveau du stock vierge, 2) un taux maximal de mortalité par pêche correspondant à un rapport de 30 % de la biomasse relative par recrue et 3) à un autre taux maximal de mortalité par pêche (pour les stocks pour lesquels il n'y a pas de données sur la croissance) fixé à 80 % du taux de mortalité naturelle. ëtant donné certaines hypothèses, la capacité qu'ont ces contraintes d'empêcher l'effondrement des stocks est indépendante des valeurs des paramètres utilisées dans le cas de la relation stock-recrutement. Une évaluation théorique générale et une comparaison aux taux actuels de mortalité par pêche appliquées à 22 stocks de poisson de fond du Pacifique nord montrent que les contraintes ne doivent pas imposer de nouvelles restrictions sur les pêches qui sont déjà gérées en fonction du rendement maximal soutenu, même lorsque la relation stock-recrutement s'écarte de la forme utilisée pour produire ces contraintes.

Throughout the history of fisheries science, one of the central questions has been the question of overfishing: How much fishing is too much? Typically, attempted answers have defined "too much fishing" (at least implicitly) as fishing beyond the optimal level. The optimal rate of fishing has usually been specified as the rate corresponding to maximum sustainable yield (MSY, e.g., Graham 1935), maximum sustainable rent (e.g., Gordon 1954), maximum discounted yield (e.g., Plourde 1970), maximum discounted rent (e.g., Clark 1973), or other value that maximizes some specified objective function.

An alternative though more complicated approach deals with the overfishing question in terms of multiple equilibria, or bifurcations in stock dynamics. Early expositions of the theory behind this approach were presented by Lewontin (1969), Holling (1973), Austin and Cook (1974), Rapport and Turner (1975), Gilpin and Case (1976), Hassell and Comins (1976), May (1977), and Reed (1978). A special case of the multiple equilibrium phenomenon is the "Allee effect" (Allee 1931), in which the mechanisms that allow a population to persist begin to fail when numbers decrease below some threshold level. In recent reviews of the subject, Dennis (1989) and Fowler and Baker (1991) found widespread support for the existence of Allee effects in natural populations.

In the multiple equilibrium approach, "too much fishing" could be defined as any fishing that causes the stock to fall below an undesirable point of no return. Fishing at such a rate would constitute an extreme example of Cushing's (1977) "recruitment overfishing". Although it is difficult to provide conclusive proof of the existence of multiple equilibria in natural systems (Connell and Sousa 1983), Table 1 lists some stocks that have been suggested to exhibit such behavior, in the sense of experiencing a severe decline and subsequently failing to recover despite a reduction in the fishing mortality rate.

In the simplest case, the multiple equilibrium approach defines overfishing as any harvest policy that causes the stock to collapse. Implicitly, at least, this is the approach endorsed by the U.S. National Oceanic and Atmospheric Administration, whose Guidelines for Fishery Management Plans (NOAA Guidelines, 50 CFR Part 602) contain the following general definition:

Table 1. A sampling of stocks suggested to exhibit multiple equilibria.

Stock	Scientific name	Citation(s)
Great Lakes lake sturgeon	*Acipenser fulvescens*	Smith 1968, Holling 1973
Antarctic fin whale	*Balaenoptera physalus*	Jones and Walters 1976
California Dungeness crab	*Cancer magister*	Botsford 1981
North Sea herring	*Clupea harengus*	Ulltang 1980
Norwegian spring-spawning herring	*Clupea harengus*	Ulltang 1980
Georges Bank herring	*Clupea harengus*	Beddington 1986
Lake Huron lake whitefish	*Coregonus clupeaformis*	Smith 1968, Holling 1973
Lesser Slave Lake lake whitefish	*Coregonus clupeaformis*	Bell et al. 1977
Lake Erie lake herring	*Leucichthys artedi*	Smith 1968, Holling 1973
British Columbia pink salmon	*Oncorhynchus gorbuscha*	Neave 1953, Ricker 1954, Peterman 1977
Lake Michigan yellow perch	*Perca flavescens*	Wells 1977, Botsford 1981
Lake Windermere (England) perch	*Perca fluviatilis*	Le Cren et al. 1972 Holling 1973
Pacific sardine	*Sardinops caerulea*	Murphy 1977, Beddington 1986

> Overfishing is a level or rate of fishing mortality that jeopardizes the long-term capacity of a stock or stock complex to produce MSY on a continuing basis.

In other words, an optimal level of production (in this case, MSY) is not the object of concern. Rather, the object of concern is the stock's long-term productive capacity (e.g., Warren et al. 1979).

One problem with this approach to the overfishing question is that none of the models commonly used for quantitative stock assessment exhibit the requisite behavior, namely a critical point ("threshold") at which the stock moves from a favorable domain (such as the one containing the MSY point) to an unfavorable one (such as a domain of inevitable extinction), except in the limiting case where the unfavorable domain consists only of the origin. The purposes of this paper are therefore to develop a model that exhibits multiple domains of attraction, to derive from the model a set of constraints that can be used to prevent overfishing, and to evaluate some of the likely impacts of imposing these constraints on a fishery insofar as such impacts can be assessed using deterministic models of stock dynamics.

Approach

A Generalized Beverton-Holt Stock-Recruitment Relationship

Typical stock-recruitment curves, such as those of Ricker (1954), Beverton and Holt (1957), and Cushing (1971), generate only two equilibria (one of which is at the origin) if post-recruitment growth and mortality parameters are independent of stock size. However, Ricker (1954) also pointed out that an appropriately drawn stock-recruitment curve can generate multiple (i.e., more than two) equilibria. The key attribute of such curves is an ability to account for depensatory pre-recruitment mortality, in which relative losses decrease with stock size (Neave 1953). Basically, this means that the second derivative of the stock-recruitment curve must be positive over some range of stock sizes below the point (if any) where the curve reaches its peak.

Much of the work in the analysis of such curves has been qualitative (e.g., Ricker 1954; Takahashi 1964; Paulik 1973; Clark 1974; Gulland 1977). However, a few formal equations describing such curves have been developed. These include three-equilibrium forms of the Ricker curve developed by Paulik and Greenough (1966), Southwood and Comins (1976), and Parrish and MacCall (1978), four-equilibrium forms of the Ricker curve developed by Larkin et al. (1964), May (1977), and Peterman (1977), a three-equilibrium form of the Beverton-Holt curve developed by Maynard Smith and Slatkin (1973), a three-equilibrium form of the Cushing curve developed by Parrish and MacCall (1978), and original multiple-equilibrium curves developed by McQueen (1975), DeAngelis et al. (1977), and Dennis (1989).

Unfortunately, these studies have mostly been used to demonstrate the fact that stock collapse is at least a theoretical possibility, without generating much in the way of quantitative management advice. This may largely be due to the fact that the depensatory stock-recruitment curves suggested to date have been fairly complex (e.g., in none of the curves listed

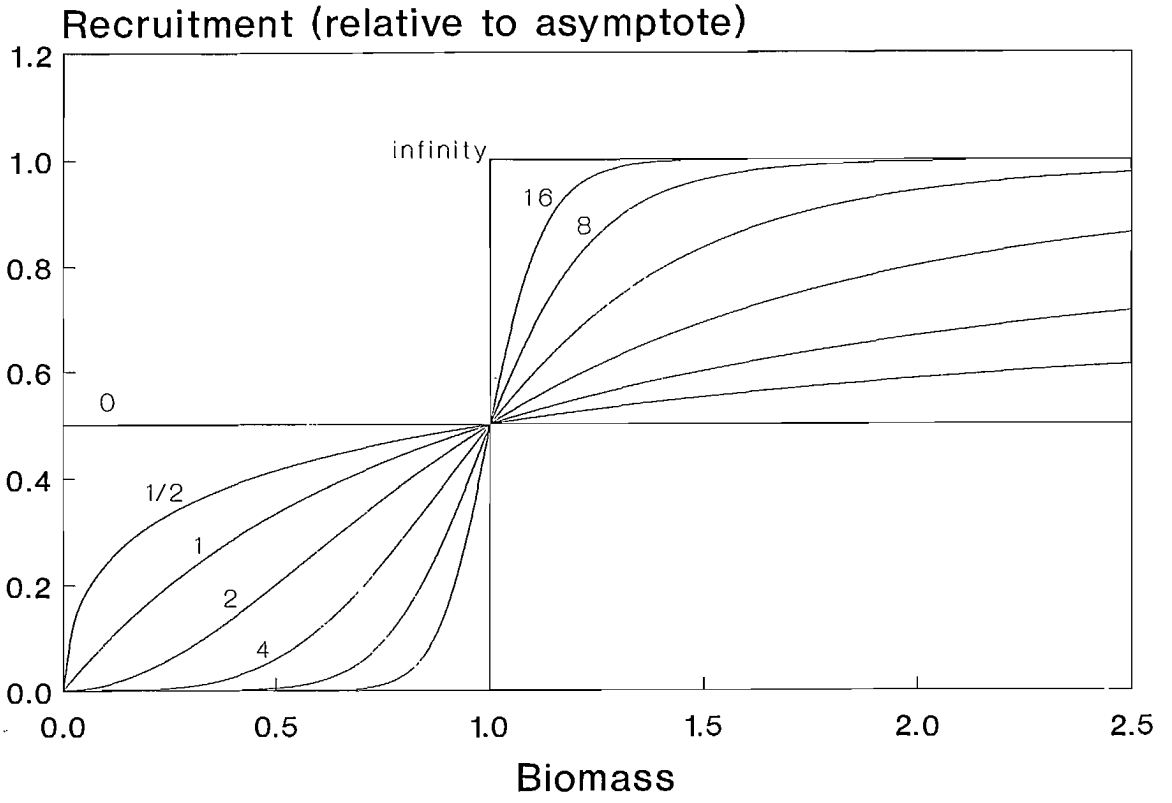


Fig. 1. A generalized Beverton-Holt stock-recruitment curve, shown for various values of the parameter r_3. Limiting cases corresponding to $r_3 = 0$ and $r_3 = \infty$ are shown, along with six intermediate cases corresponding to $r_3 = 0.5$, 1, 2, 4, 8, and 16. The parameter r_2 has been fixed at a value of 1.0.

above is it possible to solve for stock size as an explicit function of recruitment). However, it is possible to specify a particular stock-recruitment relationship that is both sufficiently complex to allow for multiple equilibria and sufficiently simple to permit analytic treatment. To begin development of such a curve, note that the Beverton- Holt stock-recruitment relationship can be written

$$R(B) = \frac{r_1}{1 + r_2 B^{-1}}, \tag{1}$$

where B = spawning stock biomass, R = recruitment (lagged appropriately), and r_1 and r_2 are positive constants. The parameter r_1 gives the value of the recruitment asymptote as biomass approaches infinity, and r_2 is a shape parameter governing the degree of curvature in the relationship.

Equation (1) can be viewed as a special case of the following three-parameter function:

$$R(B) = \frac{r_1}{1 + r_2 B^{-r_3}}, \tag{2}$$

where r_3 is a positive constant. Figure 1 shows the behavior of Equation (2) for several values of r_3, including the special case of Beverton-Holt recruitment ($r_3 = 1$).

The first and second derivatives of Equation (2) are given in Table 2. Equation (2) and its first two derivatives are plotted in Fig. 2 for particular values of r_2 and r_3.

Equation (2) and its first two derivatives attain their respective maxima at the critical points B_1, B_2, and B_3, obtained by differentiating the respective equations and setting the resulting expressions equal to zero. These critical points can all be written in the form

$$B_i = \left[\frac{r_2}{f_i(r_3)}\right]^{1/r_3}, \tag{3}$$

where $f_1(r_3) = 0$, and $f_2(r_3)$ and $f_3(r_3)$ are given in Table 2. It can be shown that a positive value of B_2 exists only for $r_3 > 1$, and a positive value of B_3 exists only for $r_3 > 2$. Critical points are indicated in Fig. 2 by vertical dashed lines.

The recruitment levels corresponding to the critical points B_i are given by

$$R_i = \frac{r_1}{1 + f_i(r_3)}. \tag{4}$$

The Relationship Between Threshold and Pristine Biomass

A common assumption is that post-recruitment growth and mortality are density independent (e.g., the "simple" yield-per-recruit model of Beverton and Holt [1957]). If this is the case, then equilibrium stock biomass will be proportional to recruitment (Goodyear 1993), or

$$B = u(F)\, R, \tag{5}$$

where F is the instantaneous rate of fishing mortality and $u(F)$ describes the equilibrium ratio of spawning stock biomass

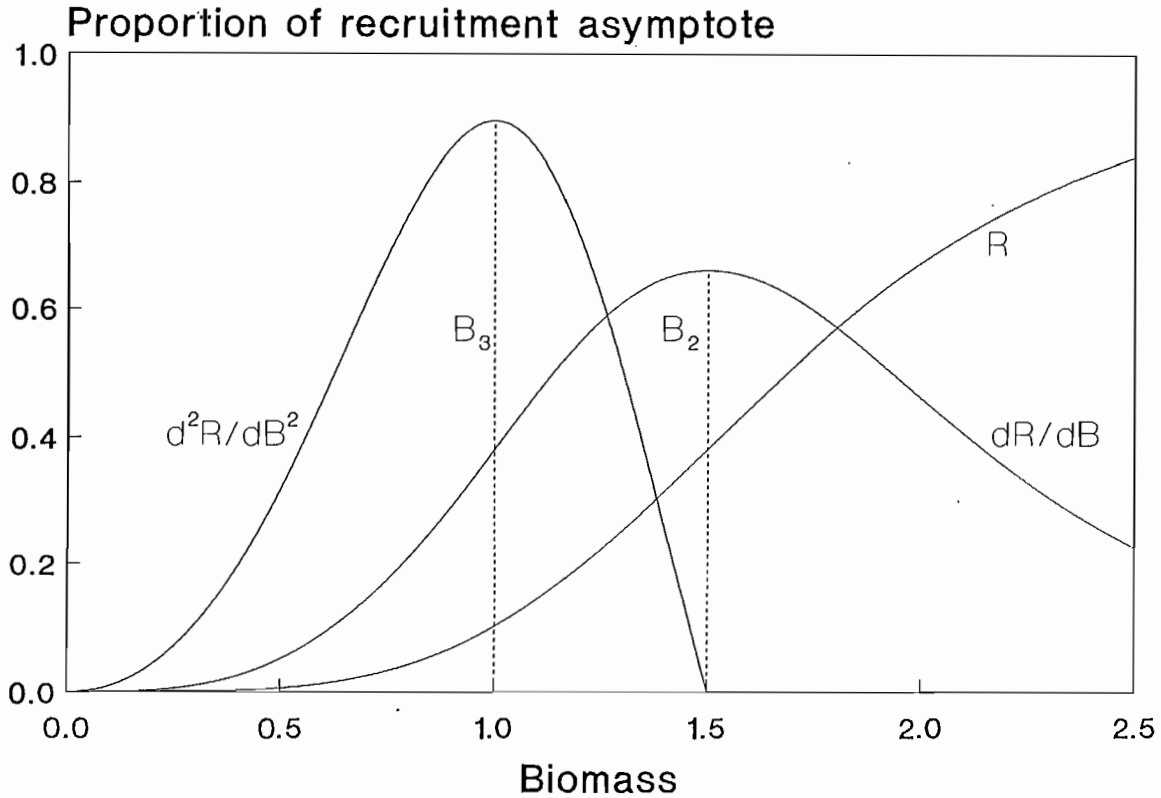


Fig. 2. An example of the stock-recruitment curve R(B) along with its first and second derivatives [$\partial R/\partial B$ and $\partial^2 R/\partial B^2$, respectively]. Values of the stock-recruitment parameters r_2 and r_3 used to generate the curves were $r_2 = 8.932$ and $r_3 = 4.204$. Maxima are indicated by the vertical dashed lines.

Table 2. The first and second derivatives of Equation (2), and the two $f_i(r_3)$ used in Equation (3) to define the second and third critical points of Equation (2) (the values of B at which the first and second derivatives peak).

$$\frac{\partial R}{\partial B} = \frac{r_2 r_3 R^2}{r_1 B^{r_3+1}}$$

$$\frac{\partial^2 R}{\partial B^2} = \frac{r_2 r_3 R^3 \left[r_2 (r_3 - 1) - (r_3 + 1) B^{r_3}\right]}{r_1^2 B^{2r_3+2}}$$

$$f_2(r_3) = \frac{r_3 + 1}{r_3 - 1}$$

$$f_3(r_3) = \frac{r_3 + 2}{2(r_3 - 1) - r_3\sqrt{\frac{3(r_3-1)}{r_3+1}}}$$

to recruitment. Equation (5) is the age-structured, biomass-based generalization of Ricker's (1954) 45-degree replacement line (with the axes reversed).

Equilibrium stock biomass and recruitment will be determined by the intersection of Equations (2) and (5), as shown for various hypothetical values of $u(F)$ in Fig. 3 (each diagonal line in Fig. 3 corresponds to an equation of the form $R = B/u(F)$). Note that for values of $r_3 > 1$ and sufficiently high values of $u(F)$, two intersections will exist. When $F = 0$, the upper intersection corresponds to pristine biomass (B_p), while the lower one corresponds to threshold biomass (B_t). Setting $F = 0$ and solving Equations (2) and (5) simultaneously gives

$$B_p^{r_3} - (r_1 u_p) B_p^{r_3-1} + r_2 = 0, \tag{6}$$

where $u_p = u(0)$. Note that B_t can be substituted for B_p in Equation (6) without loss of generality.

Unfortunately, Equation (6) cannot be solved explicitly except for low integer values of r_3 (e.g., the linear and quadratic cases). However, Equation (6) can be simplified somewhat for the special case where B_t takes the form of Equation (3), that is,

$$B_t = \left[\frac{r_2}{f(r_3)}\right]^{1/r_3}. \tag{7}$$

For this special case, Appendix A shows that Equation (6) can be manipulated to yield

$$f(r_3)\,\beta_t^{r_3} - [1 + f(r_3)]\,\beta_t + 1 = 0 \tag{8}$$

where β is the ratio of B to B_p (e.g., β_t represents B_t/B_p).

Like the general case of Equation (6), Equation (8) has the difficulty of not being explicitly solvable except for low

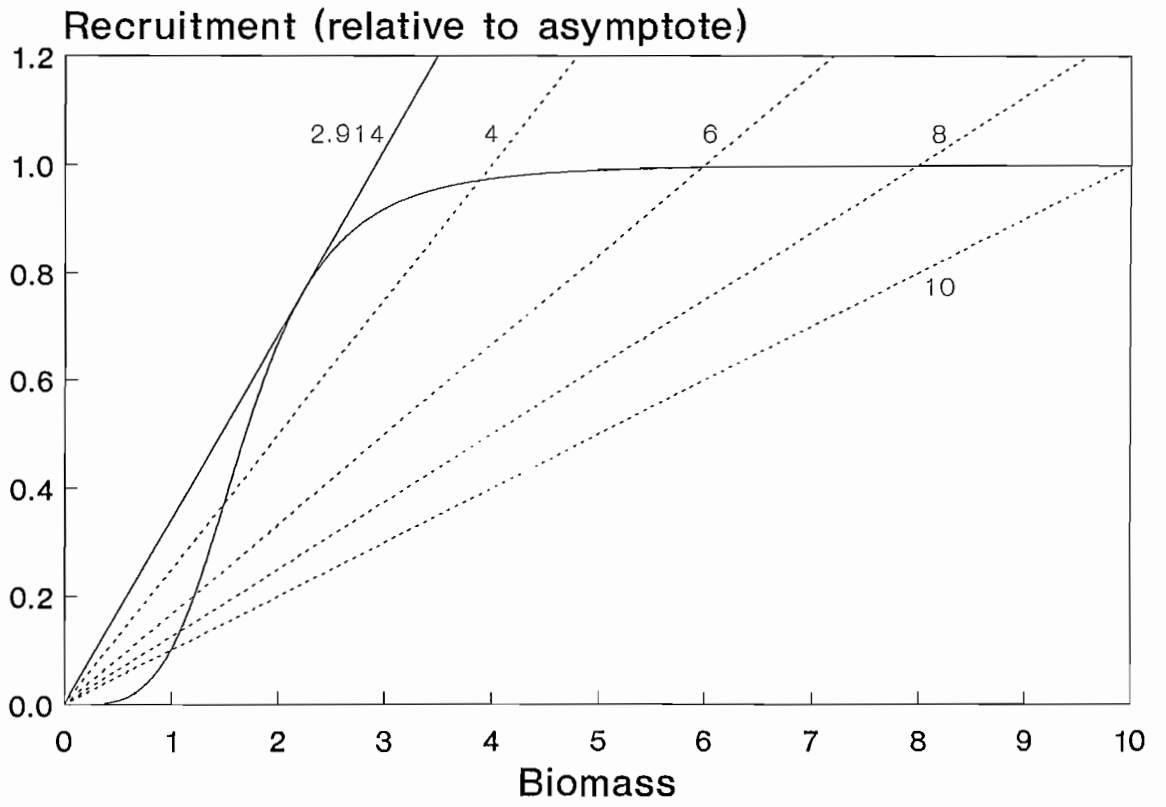


Fig. 3. Multiple equilibria as defined by different values of the biomass-per-recruit ratio $u(F)$. Each dashed line corresponds to a different value of $u(F)$. As the value of $u(F)$ decreases, the nonzero equilibria become closer, finally converging when $u(F) = 2.914$. Values of the stock-recruitment parameters r_2 and r_3 used to generate the curves were $r_2 = 8.932$ and $r_3 = 4.204$.

integer values of r_3, but it does have the advantages of eliminating all but one parameter (r_3) and employing the useful ratio β_t as the only variable. In other words, it indicates that in the special case described by Equation (7), the ratio of B_t to B_p is dependent only on r_3.

A Hypothesis Concerning Threshold Biomass

Equation (7), which implies that B_t can be described in the form of Equation (3), constitutes the critical assumption in deriving Equation (8) (Appendix A). Is there any reason to think that this assumption is appropriate? One possible rationale can be drawn by considering the problem in the context of population evolution. The derivative of Equation (2) (Table 2) describes the instantaneous rate at which recruitment changes with respect to biomass. The area around B_2 (the peak of the derivative) is the region of greatest recruitment productivity, that is, the region in which the greatest gains in recruitment (per increment in biomass) are realized. It might be reasonable to assume that natural selection would tend to act in a manner that keeps the most productive portion of the stock-recruitment curve available for the stock's use. In other words, natural selection would not tend to generate a value of B_t so high that the most productive portion of the stock-recruitment curve is sacrificed (i.e., encountered only enroute to extinction).

If this is the case, the question then becomes one of defining the lower bound of the highly productive region that surrounds B_2. One possible choice in this regard is B_3 (the peak of Equation (2)'s second derivative). Moving upward along the stock-recruitment curve, B_3 defines the point at which recruitment acceleration is maximized. Put another way, B_3 marks the place where recruitment productivity (Equation (2)'s first derivative) begins to level off. Assuming that B_3 does constitute the lower bound of the highly productive region surrounding B_2, and assuming that natural selection will tend to act so as to keep this region available for the stock's use, the following constraint will hold:

$$B_t \leq B_3 \tag{9}$$

Given Equation (9), Equation (8) can be made more specific by setting $f(r_3) = f_3(r_3)$, in which case β_t now describes the upper limit to the ratio B_t/B_p (i.e., $B_t \leq B_3$). When this specification is made, the solution to Equation (8) takes the form shown in Fig. 4, where the asymptote corresponds to the limit

$$\lim_{r_3 \to \infty} \beta_t = \frac{2-\sqrt{3}}{3-\sqrt{3}} \approx 0.211 \tag{10}$$

Equation (10) indicates that so long as a stock's biomass is kept above about 20% of its pristine level, collapse is unlikely. While this result is dependent on Equations (2), (5), and (9), it is independent of all parameter values.

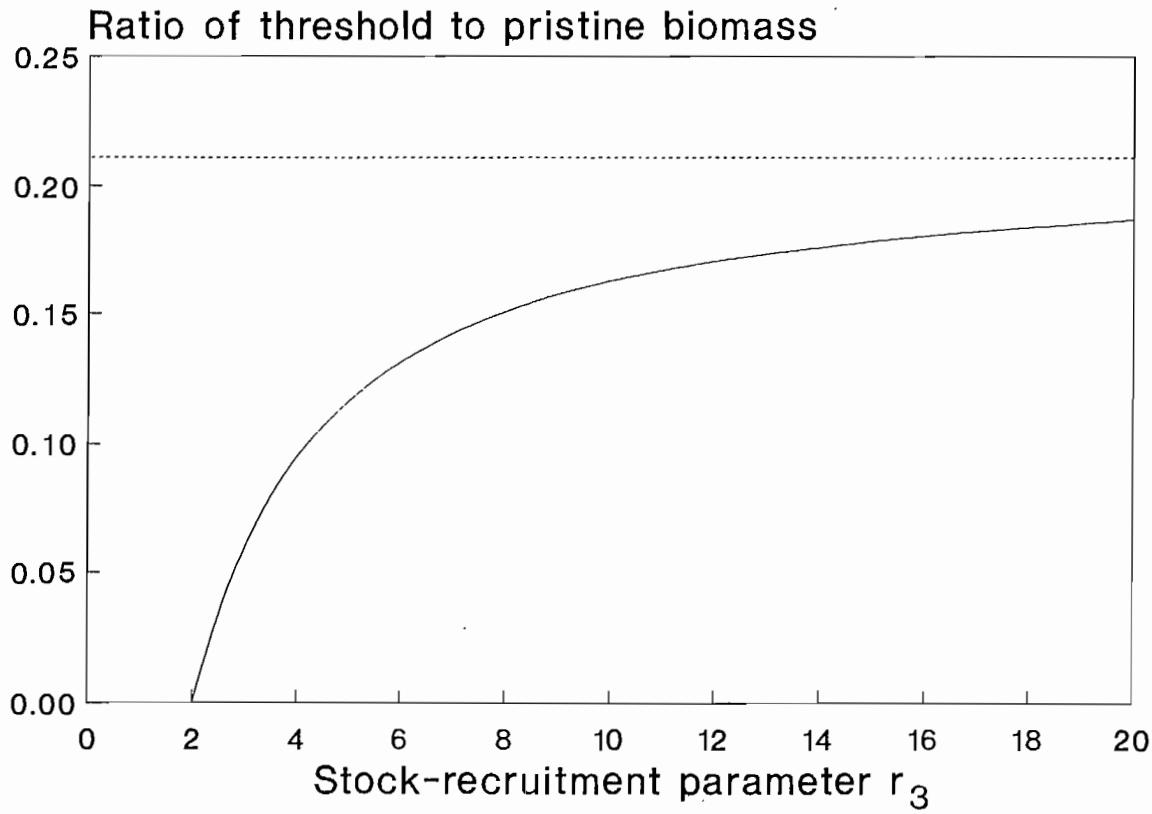


Fig. 4. Ratio of threshold to pristine biomass, plotted as a function of the stock-recruitment parameter r_3. The curve corresponds to the solution of Equation (8) when the threshold is set equal to the third critical point of the stock-recruitment curve. The horizontal dashed line denotes the asymptote of the curve as r_3 approaches infinity.

Minimum Safe Biomass-per-Recruit Ratio

As mentioned earlier, two equilibria exist in this model so long as $r_3 > 1$ and $u(F)$ is sufficiently high. As $u(F)$ decreases from its pristine value, B_p and B_t become increasingly close, ultimately converging when $u(F)$ reaches a limiting value (u_e). For all values of $u(F)$ lower than u_e, the stock will go extinct. The spawning stock biomass corresponding to u_e (B_e) is the value at which the tangent to the recruitment curve passes through the origin, or

$$B_e = [r_2 (r_3 - 1)]^{1/r_3} . \tag{11}$$

The recruitment at B_e (R_e) is given by

$$R_e = r_1 \left(\frac{r_3 - 1}{r_3} \right) . \tag{12}$$

(Note that R_e is exactly twice R_2, regardless of parameter values.) The ratio of B_e to R_e gives the value of u_e:

$$u_e = \frac{r_3 [r_2 (r_3 - 1)]^{1/r_3}}{r_1 (r_3 - 1)} . \tag{13}$$

In cases where estimates of B_p are unavailable, Equation (8) does not provide much useful information, and specifying a value for B_t becomes highly problematic. As an alternative (or in addition to specifying a threshold biomass), it may be desirable to specify a limit to the amount by which the pristine biomass-per-recruit ratio u_p can be reduced without causing the stock to collapse (assuming that the stock starts at a value of $B > B_t$). In general, define

$$\alpha = \frac{u(F)}{u_p} , \tag{14}$$

so that the goal is to determine the maximum feasible value for α_e ($= u_e/u_p$). The parameter α has been called "spawning potential ratio" by Goodyear (1993) and "percent spawning per recruit" by Mace and Sissenwine (1993).

An upper limit to α_e can be determined by assuming Equation (9), in which case B_3/R_3 becomes the lower limit to u_p and

$$\alpha_e = \frac{r_3 [(r_3 - 1) f_3 (r_3)]^{1/r_3}}{(r_3 - 1) [1 + f_3 (r_3)]} \tag{15}$$

Equation (15) is plotted in Fig. 5.

Unlike β_t, α_e exhibits a maximum at a finite value of r_3. The value of r_3 that maximizes α_e is approximately 3.776, corresponding to an α_e value of about 0.294. Thus, if the biomass-per-recruit ratio is kept above 30% of the pristine level, stock collapse would appear to be unlikely.

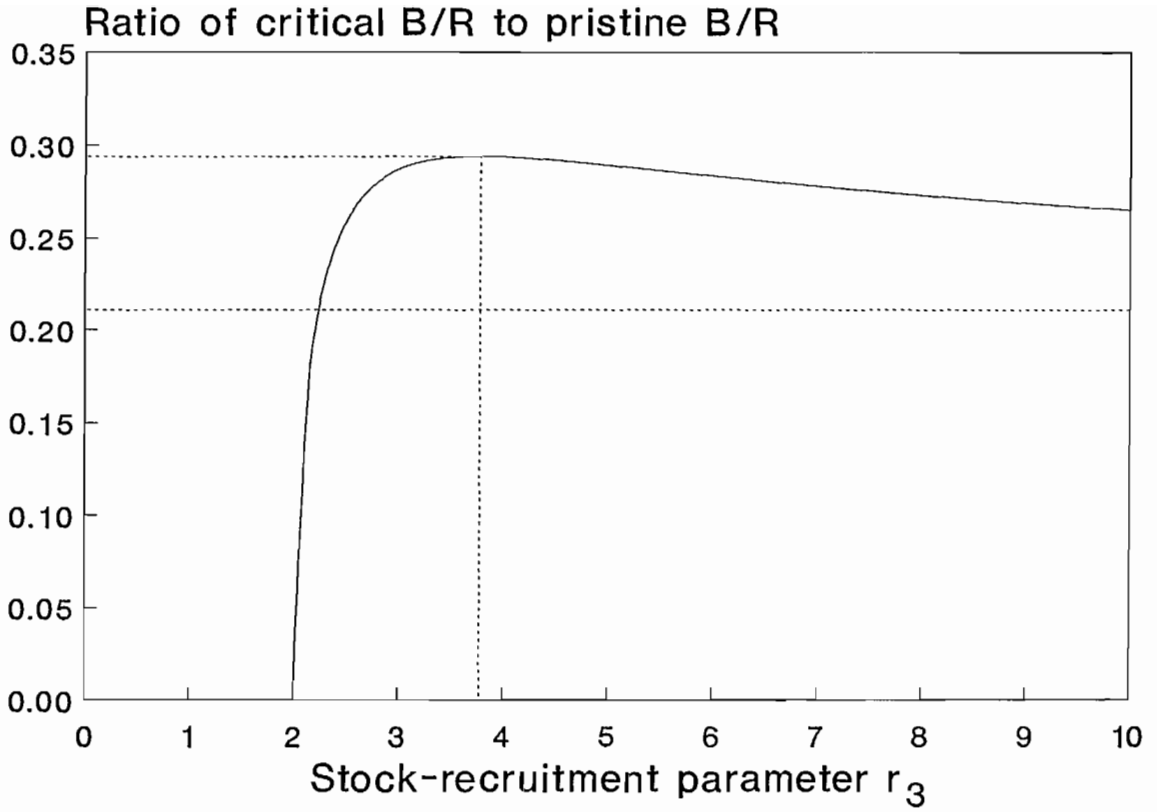


Fig. 5. Threshold biomass-per-recruit ratio as a proportion of the pristine biomass-per-recruit ratio, plotted as a function of the stock-recruitment parameter r_3. By setting the threshold equal to the third critical point of the stock-recruitment curve, Equation (15) describes the curve shown here. The horizontal dashed line extending all the way across the figure denotes the asymptote of the curve as r_3 approaches infinity. The horizontal dashed line extending only part way across the figure denotes the maximum value of Equation (15). The vertical dashed line denotes the value of r_3 that maximizes Equation (15).

Maximum Safe Fishing Mortality Rate

The preceding section describes a minimum safe value for the biomass-per-recruit ratio. To convert this ratio into a measure of the maximum safe fishing mortality rate, it is necessary to assume some functional form for the ratio $u(F)$.

Thompson (1992) developed a dynamic pool model that can serve as a simple example. In terms of biomass per recruit, the model is basically the same as that of Hulme et al. (1947), where body weight is assumed to be a linear function of age. As configured by Thompson,

$$u(F) = \frac{1 + F' + K''}{M\,(1 + F')^2}, \tag{16}$$

where M is the instantaneous rate of natural mortality, $F' = F/M$, and K'' is the ratio of growth to recruitment in the pristine stock (given in this model by $1/[M(a_r - a_0)]$, where a_r is the age of recruitment and a_0 is the age intercept of the weight-at-age relationship).

Equation (16) can be used to compute the F level that results in u_p being reduced proportionately by the factor α:

$$F' = \frac{1 + \sqrt{1 + 4\alpha K''\,(1 + K'')}}{2\alpha\,(1 + K'')} - 1. \tag{17}$$

Equation (17) attains an upper limit of $(1/\alpha) - 1$ when $K'' = 0$. It also exhibits the following lower limit as K'' approaches infinity:

$$\lim_{K'' \to \infty} F' = \frac{1}{\sqrt{\alpha}} - 1. \tag{18}$$

If α is set at the level of 0.3 recommended in the previous section, this model indicates that F' should range between about 0.826 and 2.333. Thus, if M is the only life history parameter for which an estimate is available, a fishing mortality rate set at or below about 80% of that estimate should keep the stock from collapsing.

Impacts on the Fishery: Theoretical Considerations

Threshold Biomass

It seems unlikely that any short-term economic benefit to be derived from overfishing (as defined here) would outweigh the corresponding cost of irreversible damage to the stock's productive capacity. In other words, the long-term economic effect of avoiding overfishing in principle should be positive. However, since the constraints given in the preceding sections are derived from a specific model, it is conceivable that

they could be overly restrictive (i.e., go well beyond what is necessary to prevent overfishing) in particular fisheries whose behavior differed significantly from the forms encompassed by this model. That is, even though the model presented here encompasses a wide range of possible behaviors, a particular fishery might be better modelled by alternative relationships. For example, under an alternative form of the stock-recruitment relationship, the biomass level associated with MSY (B_{MSY}) could fall well below the suggested threshold set at 20% of B_p. If fishing is halted whenever the stock falls below its threshold (as is required by the NOAA Guidelines), the threshold rule could result in some cost to the fishery, even in the case of a stock managed for MSY. Likewise, an alternative form of the stock-recruitment relationship might yield a fishing mortality rate at MSY (F_{MSY}) which is higher than the rate that sets α equal to 0.3.

Thus, it is important to test the performance of the constraints generated by the model in the context of alternative model assumptions. For example, instead of restricting the shape parameter r_3 to values greater than 1 (giving a depensatory relationship), values between 0 and 1 (no depensation) could be considered. More specifically, a parameter q can be defined over the range 0 to 1 and substituted for the free parameter r_3 in Equation (2). Then, if the ratio between r_1 and r_2 is fixed at a value of p (i.e., $r_2 = r_1/p$), Equation (2) gives the Cushing (1971) stock-recruitment relationship in the limit as r_1 approaches infinity:

$$R = \lim_{r_1 \to \infty} \frac{p r_1}{p + r_1 B(F)^{-q}} = pB(F)^q, \tag{19}$$

where $B(F)$ is the equilibrium spawning stock biomass obtained under a fishing mortality rate of F. In the limiting case of $q = 0$, recruitment is constant, while in the other limiting case of $q = 1$, recruitment is proportional to biomass.

Given Equation (19), the model described in the preceding section can be solved for those parameter combinations that result in a B_{MSY}/B_p value of 0.2. Substituting Equations (16) and (19) into Equation (5) and rearranging terms gives the following equation for equilibrium stock biomass (Thompson 1992):

$$B(F) = \left[\left(\frac{p}{M}\right)\left(\frac{1 + F' + K''}{(1 + F')^2}\right)\right]^{1/1-q}. \tag{20}$$

Multiplying both sides of Equation (20) by F then gives the equation for equilibrium yield $Y(F)$ shown below:

$$Y(F) = F\left[\left(\frac{p}{M}\right)\left(\frac{1 + F' + K''}{(1 + F')^2}\right)\right]^{1/1-q}. \tag{21}$$

Differentiating Equation (21) with respect to F and setting the resulting expression equal to zero gives the following equation for F_{MSY}:

$$\begin{aligned} F'_{MSY} &= \Big(\Big(-(q+1)K'' + 1 \\ &\quad + \sqrt{(q+1)^2 K''^2 + (6q-2)K'' + 1}\Big)/2q\Big) \\ &\quad -1, \end{aligned} \tag{22}$$

where $F'_{MSY} = F_{MSY}/M$.

Equations (20–22) can be used to examine the possible impacts of setting a threshold at 20% of B_p. Here, the ratio of B_{MSY} to B_p can range anywhere from 0 to $1/e$ (≈ 0.368). While it is difficult to predict what percentage of stocks might have a B_{MSY} value less than 20% of B_p, one option is to assume the conventional wisdom (Clark 1991) that equates F_{MSY}, M, and $F_{0.1}$ (Gulland and Boerema 1973). In the above model, this assumption holds only when $K'' = 1.5$ and $q = 2/7$ (giving approximately 82% of pristine recruitment at 50% of B_p). Under these parameter values, B_{MSY} is approximately 23% of B_p.

More generally, Equation (20) can be manipulated to show that α and β are related as follows:

$$\beta = \alpha^{1/(1-q)}. \tag{23}$$

Substituting Equation (23) into Equation (17) and solving the resulting expression simultaneously with Equation (22) yields the following polynomial in K'':

$$\begin{aligned} &\left[4\beta_{MSY}^{2-2q} - \left(q^2 + 2q + 1\right)\beta_{MSY}^{1-q}\right] K''^5 \; + \\ &\left[16\beta_{MSY}^{2-2q} - \left(3q^2 + 10q + 3\right)\beta_{MSY}^{1-q}\right] K''^4 \; + \\ &\left[24\beta_{MSY}^{2-2q} - \left(3q^2 + 18q + 2\right)\beta_{MSY}^{1-q} - q\right] K''^3 \; + \\ &\left[16\beta_{MSY}^{2-2q} - \left(q^2 + 14q - 2\right)\beta_{MSY}^{1-q} - 3q\right] K''^2 \; + \\ &\left[4\beta_{MSY}^{2-2q} - (4q - 3)\beta_{MSY}^{1-q} - 3q\right] K''^1 \; + \\ &\left[\beta_{MSY}^{1-q} - q\right] K''^0 \; = \; 0 \end{aligned} \tag{24}$$

where $\beta_{MSY} = B(F_{MSY})/B_p$.

Equation (24) has at most one positive root. This solution is plotted for $\beta_{MSY} = 0.2$ in Fig. 6, along with the loci at which F'_{MSY} takes on various constant values. Note that the curve corresponding to $\beta_{MSY} = 0.2$ is almost identical to the curve corresponding to $F'_{MSY} = 1.5$. Thus, so long as F_{MSY} does not exceed M by more than about 50%, B_{MSY} should not violate a threshold set at 20% of B_p.

Still another way to approach the problem is to look at the yield that might be forgone under a threshold set at 20% of B_p. The ratio of yield to MSY is given by

$$\begin{aligned} \frac{Y}{MSY} &= \left[\frac{F'}{F'_{MSY}}\right] \\ &\times \left[\frac{\left(1 + K'' + F'\right)\left(1 + F'_{MSY}\right)^2}{\left(1 + K'' + F'_{MSY}\right)\left(1 + F'\right)^2}\right]^{1/1-q}. \end{aligned} \tag{25}$$

For a given value of K'', Equation (25) reaches its lower bound at $q = 0$, while for a given value of q, the lower bound is reached at $K'' = 0$. These two worst-case scenarios ($K'' = 0$ with q variable, $q = 0$ with K'' variable) are shown for the case $B(F) = 0.2B_p$ in Fig. 7. Note that only those values to the left of the vertical dashed lines ($q = 0.353$ and

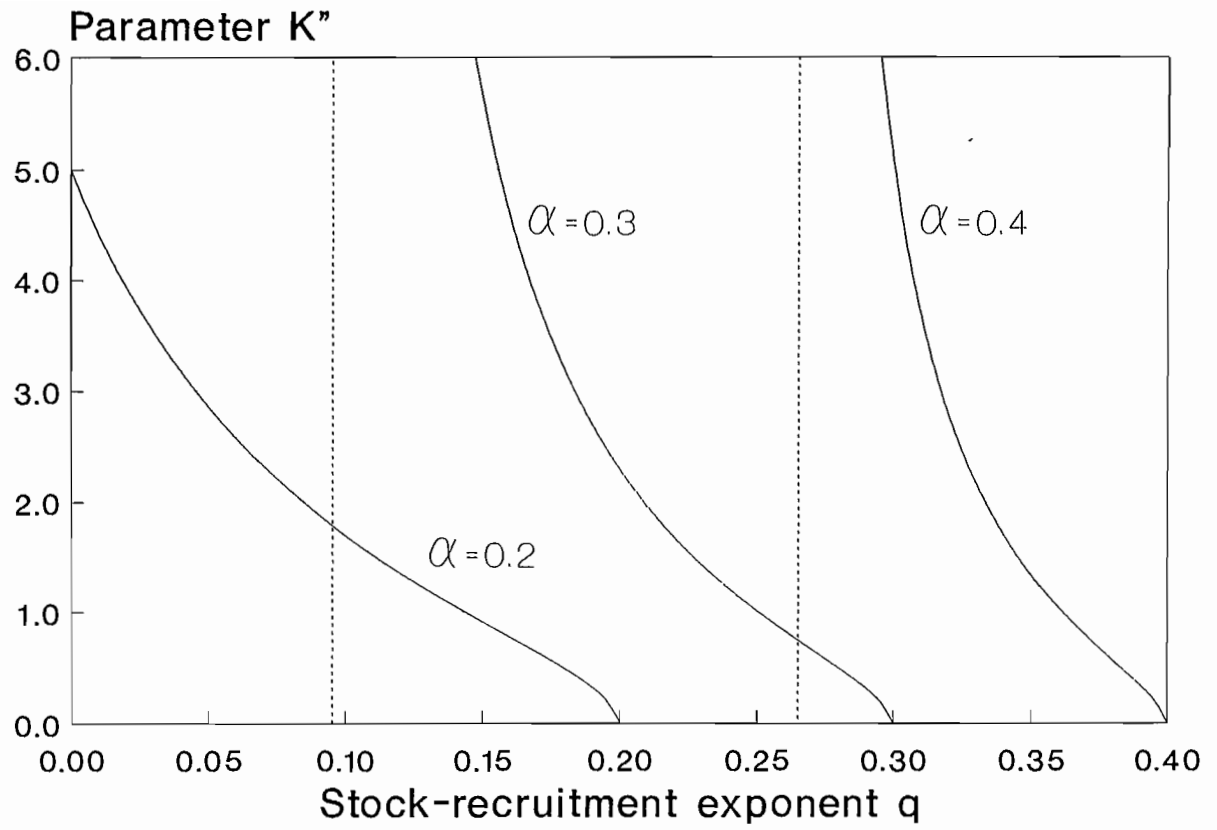


Fig. 6. Parameter combinations (K'', the pristine ratio of growth to recruitment, and q, the Cushing stock-recruitment exponent) at which the ratio of biomass at maximum sustainable yield (B_{MSY}) to pristine biomass (B_p) is 0.2 (solid curve). This locus is bounded by the values $K'' = 5$ and $q = 0.353$. Also shown are parameter combinations at which the fishing mortality rate (F) at maximum sustainable yield corresponds to fixed multiples (1.0, 1.5, and 2.0) of the natural mortality rate (M).

$K'' = 5$, respectively) are relevant, since the threshold does not constrain the fishery at values to the right (Equation (24), Fig. 6). The main conclusion to be drawn from Fig. 7 is that even when a threshold set at $0.2B_p$ does constrain the fishery, the loss in yield is probably very small (in no case exceeding 20%).

Minimum Safe Biomass-per-Recruit Ratio

The other problem to be considered here is whether constraining F by the value that sets $\alpha = 0.3$ might place undue hardship on the fishery. Equating the right-hand sides of Equations (17) and (22) gives the parameter values that set F_{MSY} equal to the F level corresponding to a given value of α:

$$K'' = \left(2\alpha(\alpha - q) + \sqrt{\alpha(q-\alpha)\left[4\alpha(q-\alpha) - (q+1)^2 + 4\alpha\right]}\right) \div \alpha[(q+1)^2 - 4\alpha]. \quad (26)$$

Equation (26) is illustrated in Fig. 8 for three different values of α. For (q, K'') combinations above and to the right of a given curve, an F_{MSY} harvest strategy will not be constrained by setting α at the associated value. Note that for $\alpha > 0.25$, a vertical asymptote exists at $q = 2(\alpha^{1/2}) - 1$. Thus, for $\alpha = 0.3$ and $q < 0.095$, an F_{MSY} harvest strategy will always be constrained, regardless of the value of K''. At higher values of q, the impact on the fishery will depend on the value of K''. For example, using the "conventional wisdom" parameters $K'' = 1.5$ and $q = 2/7$, the biomass-per-recruit ratio under an F_{MSY} harvest strategy is exactly 35% of the pristine value (cf. Clark 1991), so the constraint imposed by setting α equal to 0.3 would not be binding.

Of course, stocks are not always managed according to F_{MSY}. Another common strategy is to harvest the stock at the $F_{0.1}$ rate. The $F_{0.1}$ rate is the value at which the slope of the yield-per-recruit curve (as a function of F) is one tenth of the value at the origin. For the model described by Equation (16), Thompson (1989) showed that this rate could be computed as a special case of the following polynomial in F':

$$PF'^3 + 3PF'^2 + \left[3P + \frac{K''-1}{K''+1}\right]F' + P - 1 = 0, \quad (27)$$

where P is the slope of the yield-per-recruit curve relative to the slope at the origin ($P = 0.1$ in the case of $F_{0.1}$).

As noted earlier, the F value corresponding to α is bounded above by $(1/\alpha) - 1$ and below by Equation (18). Inserting these limiting values into Equation (27) together

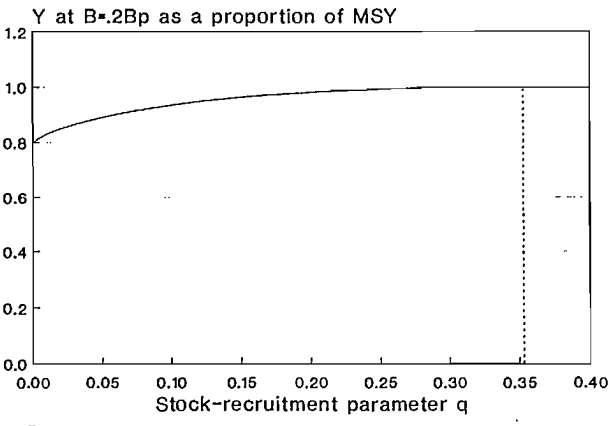


Figure 7a.

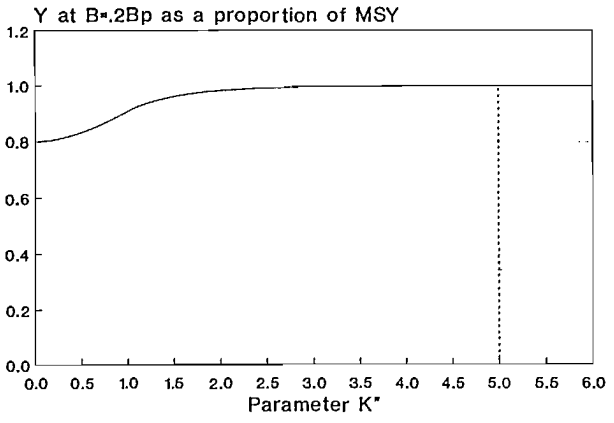


Figure 7b.

Fig. 7. The ratio between yield (Y) at the suggested threshold (20% of pristine biomass, B_p) and maximum sustainable yield (MSY). The dashed lines indicate the parameter values at which the threshold corresponds to the biomass level (B) at MSY. a) Relative yield when the pristine growth-to-recruitment ratio K'' is zero and the Cushing stock-recruitment exponent q is allowed to vary. b) Relative yield when q is zero and K'' is allowed to vary.

with the limiting values of K'' (0 and infinity, respectively) and solving for α gives the following lower and upper bounds, respectively:

$$\alpha = \sqrt{P}, \tag{28}$$

and

$$\alpha = \left(\frac{\sqrt{24P+1}}{6}\right) \times \cos\left[\left(\frac{1}{3}\right)\cos^{-1}\left(\frac{216P^2+36P+1}{(24P+1)^{3/2}}\right)\right] + \frac{1}{12}. \tag{29}$$

For the special case of $P = 0.1$, Equations (28) and (29) give limits of $\alpha = 0.316$ and $\alpha = 0.393$, respectively. Thus, constraining F by the value that sets $\alpha = 0.3$ should not impact a fishery managed at the $F_{0.1}$ rate (a P value of 0.09 would be required to observe an α value as low as 0.3).

Application to North Pacific Groundfish

To illustrate how the constraints suggested above might impact actual fisheries, the groundfish stocks managed by the North Pacific Fishery Management Council (the "Council") in the U.S. Exclusive Economic Zone can be considered as examples. Currently, the Council sets annual harvest limits for 27 groundfish species or species complexes ("management categories") in the eastern Bering Sea (BS), Aleutian Islands (AI), and Gulf of Alaska (GOA) regions, as shown in Table 3. Note that the Council applies the term "groundfish" somewhat loosely. Not all species listed in Table 3 are true groundfish; some, such as squid, are not even fish.

Impacts Resulting from the Threshold Exceeding B_{MSY}

As noted earlier, it is theoretically possible for a threshold set at 20% of pristine biomass to exceed B_{MSY}. The following discussion evaluates this possibility for North Pacific groundfish given the present state of knowledge concerning those stocks.

In setting annual harvest limits, the Council's principal source of scientific information is the Stock Assessment and Fishery Evaluation (SAFE) report. The 1989 SAFE reports for North Pacific groundfish (plus a study by Quinn et al. (1990)) contain F_{MSY} or pristine biomass estimates for ten stocks, as shown in Table 4 (Bering Sea/Aleutian Islands Groundfish Plan Team et al. 1989; Gulf of Alaska Groundfish Plan Team 1989). Of the seven stocks for which estimates of pristine biomass are available, Table 4 indicates that a threshold set at 20% of pristine biomass poses no obvious constraint on the fishery, either in terms of current biomass levels or biomass at MSY. The stocks with the greatest potential for falling beneath the suggested threshold appear to be the Pacific ocean perch stocks. Current biomass and B_{MSY} for these two stocks are estimated to fall between 24% and 29% of pristine biomass (however, it should be noted that the figures for Pacific ocean perch are based on subjective estimates of stock-recruitment parameters).

The preceding section on "Theoretical Considerations" evaluated the conditions under which a threshold set at 20% of pristine biomass might exceed B_{MSY}. The evaluation showed that the threshold should exceed B_{MSY} only when F_{MSY} exceeds the natural mortality rate by more than about 50%. According to Table 4, of the three stocks for which estimates of pristine biomass are unavailable but estimates of F_{MSY} are available, the ratios of F_{MSY} to M range from 0.62 to 1.00, well below the 1.5 figure that would place B_{MSY} close to the suggested threshold.

Impacts Resulting from Constraints on Fishing Mortality

As a preliminary step toward setting final harvest limits, the Council adopts a set of "acceptable biological catch" (ABC) levels based largely on information contained in the

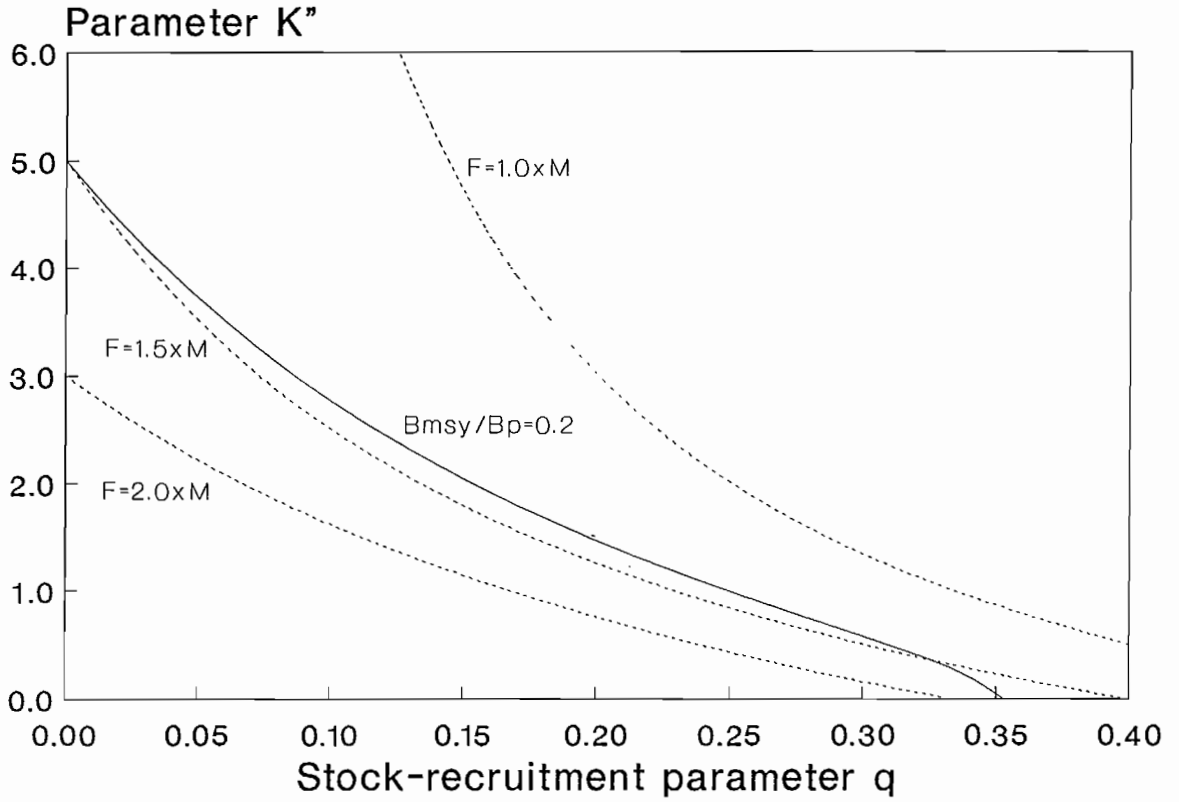


Fig. 8. Parameter combinations at which the fishing mortality rate at maximum sustainable yield sets the biomass-per-recruit ratio equal to three constant proportions (α) of its pristine value (0.2, 0.3, and 0.4). The vertical dashed lines indicate asymptotes for $\alpha = 0.3$ and $\alpha = 0.4$.

SAFE reports. Table 5 lists the fishing mortality rate corresponding to the Council's 1990 ABC (F_{ABC}) for each management category, along with alternative maximum allowable fishing mortality rates. Since the 1989 SAFE reports do not provide the fishing mortality rate at which the biomass-per-recruit ratio is reduced to 30% of its pristine value, the following approaches were used to complete Table 5 (square brackets enclose the list of management categories to which each approach was applied):

1. Age-specific schedules of maturity and weight [BS — walleye pollock, GOA — walleye pollock].
2. Beverton and Holt's (1957) "simple" model [BS — yellowfin sole, arrowtooth flounder, other flatfish (using male Alaska plaice parameters), and Atka mackerel; GOA — deep flatfish (using flathead sole parameters), shallow flatfish (using rock sole parameters), arrowtooth flounder, demersal shelf rockfish (using male yelloweye rockfish parameters), and thornyhead (using shortspine thornyhead parameters)].
3. Deriso's (1980, generalized by Schnute 1985) delay-difference model [BS — Greenland turbot, sablefish, and Pacific ocean perch; GOA — Pacific cod, sablefish, and slope rockfish (using Pacific ocean perch parameters)].
4. Equation (17) [BS — Pacific cod and rock sole].

Of the 27 groundfish management categories, the Council sets F_{ABC} values for 24. Of these 24, two (BS and AI "other" rockfish) could not be evaluated in terms of the suggested constraints due to lack of data. Of the remaining 22 management categories, Table 5 shows that the suggested constraints would have reduced F_{ABC} in only two cases. The first is GOA Pacific cod, where the Council's F_{ABC} exceeded F_{MSY} by about 58% (however, it should be noted that this was probably due to a belief on the part of the Council that F_{MSY} was underestimated rather than a deliberate attempt to exceed the true F_{MSY}). The second case is GOA pelagic shelf rockfish, where the Council's F_{ABC} exceeded 80% of M by 25%.

Discussion

The above sections developed a set of three constraints designed to insure against overfishing, where overfishing is defined as any harvest policy that causes a stock to collapse. These constraints (which can be used separately or in combination) are as follow:

A. When an estimate of pristine biomass is available, fishing should cease whenever the stock falls to a level less than about 20% of this estimate.

Table 3. "Groundfish" species currently managed by the North Pacific Fishery Management Council.

Bering Sea and Aleutian Islands

Management category	Species included
Pollock	walleye pollock (*Theragra chalcogramma*)
Pacific cod	Pacific cod (*Gadus macrocephalus*)
Yellowfin sole	yellowfin sole (*Limanda aspera*)
Greenland turbot	Greenland halibut (*Reinhardtius hippoglossoides*)
Arrowtooth flounder	arrowtooth flounder (*Atheresthes stomias*)
Rock sole	rock sole (*Lepidopsetta bilineata*)
Other flatfish	all species of the families Bothidae and Pleuronectidae not included in the above four categories, principally flathead sole (*Hippoglossoides elassodon*) and Alaska plaice (*Pleuronectes quadrituberculatus*)
Sablefish	sablefish (*Anoplopoma fimbria*)
Pacific ocean perch	Pacific ocean perch (*Sebastes alutus*), northern rockfish (*S. polyspinis*), rougheye rockfish (*S. aleutianus*), shortraker rockfish (*S. borealis*), and sharpchin rockfish (*S. zacentrus*)
Other rockfish	all Sebastes species (about 29) not included in the Pacific ocean perch category, along with shortspine thornyhead (*Sebastolobus alascanus*) and longspine thornyhead (*S. altivelis*)
Atka mackerel	Atka mackerel (*Pleurogrammus monopterygius*)
Squid & other species	all squid species, all octopus species, all shark species, and all species of the families rajidae, osmeridae, and cottidae

Gulf of Alaska

Management category	Species included
Pollock	walleye pollock
Pacific cod	Pacific cod
Deep flatfish	rex sole (*Glyptocephalus zachirus*), Dover sole (*Microstomus pacificus*), and flathead sole
Shallow flatfish	all species of the families Bothidae and Pleuronectidae (except arrowtooth flounder) not included in the deep flatfish category, principally rock sole, starry flounder (*Platichthys stellatus*), yellowfin sole, butter sole (*Isopsetta isolepis*), and English sole (*Parophrys vetulus*)
Arrowtooth flounder	arrowtooth flounder
Sablefish	sablefish
Slope rockfish	Pacific ocean perch, northern rockfish, rougheye rockfish, shortraker rockfish, sharpchin rockfish, harlequin rockfish (*Sebastes variegatus*), redbanded rockfish (*S. babcocki*), greenstriped rockfish (*S. elongatus*), yellowmouth rockfish (*S. reedi*), darkblotched rockfish (*S. crameri*), aurora rockfish (*S. aurora*), blackgill rockfish (*S. melanostomus*), chilipepper (*S. goodei*), pygmy rockfish (*S. wilsoni*), shortbelly rockfish (*S. jordani*), splitnose rockfish (*S. diploproa*), stripetail rockfish (*S. saxicola*), and vermilion rockfish (*S. miniatus*)
Pelagic shelf rockfish	dusky rockfish (*Sebastes ciliatus*), black rockfish (*S. melanops*), widow rockfish (*S. entomelas*), blue rockfish (*S. mystinus*), and yellowtail rockfish (*S. flavidus*)
Demersal shelf rockfish	Bocaccio (*Sebastes paucispinis*), canary rockfish (*S. pinniger*), China rockfish (*S. nebulosus*), copper rockfish (*S. caurinus*), quillback rockfish (*S. maliger*), rosethorn rockfish (*S. helvomaculatus*), silvergray rockfish (*S. brevispinis*), tiger rockfish (*S. nigrocinctus*), yelloweye rockfish (*S. ruberrimus*), and redstripe rockfish (*S. proriger*)
Thornyheads	shortspine thornyhead and longspine thornyhead
Other species	all squid species, all octopus species, all shark species, Atka mackerel, and all species of the families rajidae, osmeridae, and cottidae

Table 4. Statistics relating to maximum sustainable yield and pristine biomass for various groundfish stocks.

Bering Sea (BS) and Aleutian Islands (AI)							
Management Category	F_{MSY}	M	B_p	B_{MSY}	$B(90)$	β_{MSY}	β_{90}
Pollock (BS)	0.31	0.30	13.830	6.120	5.844	0.44	0.42
Pacific cod	0.18	0.29	n/a	0.879	1.335	n/a	n/a
Greenland turbot*	0.07	0.18	1.073	0.399	0.357	0.37	0.33
Rock sole	0.18	0.20	n/a	n/a	1.194	n/a	n/a
Sablefish* (BS)	0.27	0.10	0.083	0.018	0.033	0.21	0.39
Sablefish* (AI)	0.27	0.10	0.186	0.040	0.082	0.21	0.44
P. ocean perch* (BS)	0.06	0.05	0.267	0.076	0.068	0.29	0.26
P. ocean perch* (AI)	0.06	0.05	0.600	0.173	0.158	0.29	0.26
Gulf of Alaska (GOA)							
Management Category	F_{MSY}	M	B_p	B_{MSY}	$B(90)$	β_{MSY}	β_{90}
Pacific cod	0.12	0.29	0.695	0.272	0.505	0.39	0.73
Sablefish*	0.27	0.10	0.675	0.145	0.311	0.21	0.46
Slope rockfish*	0.08	0.05	1.391	0.383	0.329	0.28	0.24
Thornyhead*	0.07	0.07	n/a	n/a	0.080	n/a	n/a

Legend: F_{MSY} = MSY fishing mortality rate
M = natural mortality rate
B_p = pristine biomass
B_{MSY} = biomass at MSY
$B(90)$ = projected biomass in 1990
$\beta_{MSY} = B_{MSY}/B_p$
$\beta_{90} = B(90)/B_p$

Notes:
1) All biomass estimates are in millions of metric tons.
2) An asterisk (*) by a species indicates that the corresponding figures are based on subjective (as opposed to empirical) estimates of stock-recruitment parameters or B_{MSY}.
3) All information was taken from the final 1989 Stock Assessment and Fishery Evaluation (SAFE) reports, except for BS pollock. For this stock, B(90) was taken from the SAFE document and all other information was taken from Quinn et al. (1990).
4) For some stocks, the documents provide estimates based on several different model versions. In such cases, single values were chosen as follows: BS pollock — Beverton-Holt version; BSAI and GOA sablefish — version tuned to survey point estimates; BSAI POP, GOA slope rockfish, and GOA thornyhead — versions that give 90% of pristine recruitment at 50% of pristine biomass.
5) Biomass figures for BSAI POP and GOA slope rockfish refer to *Sebastes alutus* only.

B. When estimates of the relevant life history parameters are available, the fishing mortality rate should be set so as to maintain the (spawning) biomass-per-recruit ratio at a level no less than about 30% of the pristine level.

C. When the natural mortality rate is the only life history parameter for which an estimate is available, the fishing mortality rate should be set at a level no higher than about 80% of this estimate.

It should be emphasized that Constraints (A) and (B) are dependent only on Equations (2), (5), and (9). In addition to these three equations, Constraint (C) is dependent on Equation (16). The constraints are totally independent of the parameter values used in these equations. Because they are directly related to the potential for stock collapse and because they are explicitly derivable from a small number of qualitative assumptions regarding population dynamics, the constraints pose three significant advantages over some other overfishing criteria that have been proposed: 1) they avoid the problem of requiring types or amounts of data that are often unavailable (e.g., stock-recruitment parameters required to compute F_{MSY}), 2) they avoid the problem of being critically dependent on arbitrarily chosen parameter values (e.g., the "0.1" in $F_{0.1}$), and 3) they avoid the problem of viewing historical averages as necessarily prescriptive (e.g., F_{rep} [Sissenwine and Shepherd 1987]).

Perhaps the most tenuous of the assumptions used to derive the suggested constraints is Equation (9), which states that the threshold will never exceed the third critical point of the stock-recruitment relationship. Although clearly speculative at present, this assumption finds support in the following arguments: 1) it is defensible in terms of evolutionary theory, as discussed earlier; 2) ecological theory is beginning to find significance in analogous critical points (e.g., Fowler 1988); and 3) the resulting management implications are reasonable.

Note that the evolutionary argument advanced here in support of a threshold below B_3 implies that biological communities, as well as individual populations, are organized by a process of natural selection. This view is discussed in detail by Loehle and Pechmann (1988), who suggest that ecological niches are occupied by those populations that are able to out compete others, for example by being more resilient.

Table 5. 1990 harvest strategies and fishing mortality rates compared with three overfishing criteria.

Bering Sea (BS) and Aleutian Islands (AI)

Management category	Strategy	F_{ABC}	F_{MSY}	$F_{30\%}$	$0.8M$
Pollock (BS)	$F_{0.1}$	0.31	0.31	0.49	0.24
Pollock (AI)	$F_{0.1}$	0.31	n/a	0.49	0.24
Pacific cod	F_{MSY}	0.18	0.18	0.31	0.23
Yellowfin sole	$F_{0.1}$	0.17	n/a	0.17	0.10
Greenland turbot*	F_{BYC}	0.02	0.07	0.19	0.14
Arrowtooth flounder	$F_{0.1}$	0.18	n/a	0.25	0.16
Rock sole	F_{MSY}	0.18	0.18	0.21	0.16
Other flatfish	F_{PRX}	0.18	n/a	0.23	0.16
Sablefish* (BS)	$F_{0.1}$	0.13	0.27	0.18	0.08
Sablefish* (AI)	$F_{0.1}$	0.13	0.27	0.18	0.08
Pacific ocean perch* (BS)	F_{MSY}	0.06	0.06	0.08	0.04
Pacific ocean perch* (AI)	F_{MSY}	0.06	0.06	0.08	0.04
Other rockfish (BS)	F_{PRX}	0.06	n/a	n/a	n/a
Other rockfish (AI)	F_{PRX}	0.06	n/a	n/a	n/a
Atka mackerel	$F_{0.1}$	0.27	n/a	0.33	0.16
Squid & other species	F_{HIS}	n/a	n/a	n/a	n/a

Gulf of Alaska

Management category	Strategy	F_{ABC}	F_{MSY}	$F_{30\%}$	$0.8M$
Pollock	?	0.10	n/a	0.45	0.32
Pacific cod	?	0.19	0.12	0.31	0.23
Deep flatfish	$F_{0.1}$	0.20	n/a	0.26	0.16
Shallow flatfish	$F_{0.1}$	0.20	n/a	0.26	0.16
Arrowtooth flounder	$F_{0.1}$	0.17	n/a	0.22	0.18
Sablefish*	$F_{0.1}$	0.13	0.27	0.18	0.08
Slope rockfish*	$M/2$	0.03	0.08	0.10	0.04
Pelagic shelf rockfish	M	0.05	n/a	n/a	0.04
Demersal shelf rockfish	?	n/a	n/a	0.07	0.03
Thornyheads*	F_{HIS}	0.04	0.07	0.08	0.06
Other species	?	n/a	n/a	n/a	n/a

Legend (F = fishing mortality rate):
$F_{0.1}$ = F value at which the yield-per-recruit curve's slope is 10% of the slope at the origin.
$F_{30\%}$ = F value at which the biomass-per-recruit ratio is reduced to 30% of its pristine value.
F_{ABC} = F value used to calculate acceptable biological catch.
F_{BYC} = F value that allows for bycatch only.
F_{HIS} = F value that sets ABC equal to the historic average.
F_{MSY} = F value corresponding to maximum sustainable yield.
F_{PRX} = proxy target F adopted from the preceding species.
M = natural mortality rate.
Note: For species marked with an asterisk (*), F_{MSY} is based on subjective estimates of stock-recruitment parameters or B_{MSY}.

Consequently, Loehle and Pechmann contend that evolutionary processes can result in the restriction of population-level traits to certain regions of the logically permissible strategy space. Walters (1975) also suggested that evolutionary considerations could be used to constrain parameter space in population dynamic models. While this argument clearly requires a form of group selection, it does not necessarily require selection for traits that are nonadaptive at the individual level, and so should escape the pitfalls involved in advancing group selection as an explanation for the evolution of altruism (Wade 1978; Goodnight et al. 1992).

With regard to their overall "reasonableness", it is interesting to note how well Constraints (A–C) conform to standards with which fishery scientists already seem to feel comfortable. As shown above, Constraints (A–C) are unlikely to impinge severely on some of the more common management measures recommended by fishery scientists, at least insofar as such impingements can be assessed using deterministic models of stock dynamics.

Other examples of concordance can be found as well, although mostly in studies that employ the constraints on an *ad hoc* basis or derive them as target values for particular stocks only. For example, the 20% figure used to define a threshold biomass corresponds exactly to the figure employed by Beddington and Cooke (1983), Clark et al. (1985), and Butterworth and Bergh (1993). It is within the range of 20%–30% derived by Quinn et al. (1990), the 20%–50% range derived

by Clark (1991), and the 7%–27% range used by Basson and Beddington (1993). Likewise, the 30% figure used to define a minimum relative biomass-per-recruit ratio is identical to the value derived by Gabriel et al. (1989). It is close to the 35% figure derived by Clark (1991), and within the range of 20%–35% employed by many fishery management plans in the U.S. (Mace and Sissenwine 1993). Finally, the 80% figure used to define a maximum F/M ratio is not far from the 100% value suggested in several studies as a reasonable approximation of either F_{MSY} or $F_{0.1}$ in the absence of more detailed information (e.g., Alverson and Pereyra 1969; Shepherd 1982; Deriso 1987; Kimura 1988; Clark 1991).

Nevertheless, certain caveats regarding the development of the three constraints should be noted. First, the stock-recruitment relationship described by Equation (2) allows for only two stable domains of attraction, whereas all of the studies described in Table 1 suggest the existence of at least three: an upper domain, a middle domain, and a lower domain containing (perhaps limited to) the origin. This restriction may not be too problematic in practice, however, since the applicability of the constraints requires only that the form of Equation (2) hold for values of $B > B_t$ (i.e., any number of stable domains could exist below the uppermost domain without changing the results).

Second, the assumption that the threshold falls below B_3 cannot hold for values of r_3 between 1 and 2, since B_3 is negative over this range. However, as shown in Appendix B, the constraints resulting from the assumption that $B_t < B_3$ should prove sufficiently conservative even when $r_3 \leq 2$, so long as the assumption that the threshold falls well below the inflection point continues to hold (for the case where $r_3 = 2$, a maximum B_t/B_2 value of 75% renders Constraint (A) sufficiently conservative, while a maximum B_t/B_2 value of 25% renders Constraint (B) sufficiently conservative).

Third, the constraints all consist of values drawn from the conservative ends of their respective ranges. For example, the 20% figure in Constraint (A) could range anywhere from 0% to 20%. Likewise, the 30% figure in Constraint (B) could range anywhere from 0% to 30%, and the 80% figure in Constraint (C) could range anywhere from 80% to 230%.

Finally, the entire analysis presumes a deterministic system in which the manager has good knowledge of life history parameters (except stock-recruitment parameters), catch, and biomass. The existence of either significant process error or significant measurement error could alter the conclusions reached here.

In conclusion, it should be emphasized that although the analyses presented here have shown that Constraints (A) and (B) should impose few additional costs on fisheries that are already managed for MSY, such fisheries are not really the constraints' intended target. Rather, it is anticipated that the constraints' main benefit would accrue from applying them to fisheries where good estimates of F_{MSY} are not available. There, the suggested constraints should provide a relatively painless, objective, and scientifically defensible means of safeguarding against overly aggressive harvest strategies.

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Appendix A

One special case of Equation (6) results in a simplified relationship between B_t and B_p. First, assume that B_t can be written in the form of Equation (3), giving

$$r_2 = f(r_3)\, B_t^{r_3}. \tag{A1}$$

Second, substitute B_t for B_p and Equation (A1) for r_2 in Equation (6), giving

$$B_t^{r_3} - (r_1 u_p)\, B_t^{r_3-1} + f(r_3)\, B_t^{r_3} = 0. \tag{A2}$$

Third, solve Equation (A2) for the product $r_1 u_p$:

$$r_1 u_p = [1 + f(r_3)]\, B_t. \tag{A3}$$

Fourth, substitute Equations (A1) and (A3) into Equation (6):

$$B_p^{r_3} - [1 + f(r_3)]\, B_t B_p^{r_3-1} + f(r_3)\, B_t^{r_3} = 0. \tag{A4}$$

Fifth, define

$$\beta = \frac{B}{B_p} \tag{A5}$$

for any value of B (e.g., β_t would represent B_t/B_p).

Sixth, use Equation (A5) to substitute B_t/β_t for B_p in Equation (A4), giving

$$\left[\frac{B_t}{\beta_t}\right]^{r_3} - [1 + f(r_3)]\, B_t \left[\frac{B_t}{\beta_t}\right]^{r_3-1} + f(r_3)\, B_t^{r_3} = 0. \tag{A6}$$

Finally, eliminate B_t in Equation (A6) and rearrange terms to yield

$$f(r_3)\, \beta_t^{r_3} - [1 + f(r_3)]\, \beta_t + 1 = 0. \tag{A7}$$

Appendix B

Equation (9) ($B_t \leq B_3$) can hold only for values of $r_3 > 2$. For $1 < r_3 \leq 2$, a threshold will exist, but it will necessarily occur at $B > B_3$ since B_3 is negative over this range of r_3. As discussed in the main text, it seems reasonable to assume that the threshold will occur at a biomass level below the inflection point. The question is how far below. One way to approach this question is to take the constraints derived in the main text as given, then determine the conditions under which they would hold in the case where $1 < r_3 \leq 2$. At values of r_3 close to 1, it seems likely that these constraints will always be sufficiently conservative regardless of the amount by which the inflection exceeds the threshold, since the inflection will always be very close to the origin ($B_2 = 0$ at $r_3 = 1$). Values of r_3 close to 2 are more problematic. Fortunately, B_2, B_t, and B_p can be derived explicitly for the case of $r_3 = 2$, making the problem quite tractable for this end of the range.

For the special case of $r_3 = 2$, the equations for B_2, B_t, and B_p are

$$B_2 = \sqrt{\frac{r_2}{3}}, \tag{B1}$$

$$B_t = \left(\frac{r_1 u_p}{2}\right)\left[1 - \sqrt{1 - r_2\left(\frac{2}{r_1 u_p}\right)^2}\right], \tag{B2}$$

and

$$B_p = \left(\frac{r_1 u_p}{2}\right)\left[1 + \sqrt{1 - r_2\left(\frac{2}{r_1 u_p}\right)^2}\right]. \tag{B3}$$

A convenient normalization is obtained by defining

(B4) $$x = \frac{\sqrt{r_2}}{r_1 u_p}.$$

Using Equation (B4), the ratio between B_t and B_2 (ρ) can be written

(B5) $$\rho = \left(\frac{\sqrt{3}}{2}\right)\left(\frac{1-\sqrt{1-4x^2}}{x}\right)$$

and β_t can be written

(B6) $$\beta_t = \frac{1-\sqrt{1-4x^2}}{1+\sqrt{1-4x^2}},$$

or

(B7) $$\beta_t = \frac{\rho^2}{3}$$

Thus, Constraint (A) ($\beta_t < 20\%$) will be sufficiently conservative for the case of $r_3 = 2$ whenever ρ is less than about 75%.

Constraint (B) ($\alpha_e < 30\%$) may be examined by noting that α_e can be written (for the case where $r_3 = 2$) simply as a function of β_t:

(B8) $$\alpha_e = \sqrt{1-\left(\frac{1-\beta_t}{1+\beta_t}\right)^2}$$

Equation (B8) indicates that Constraint (B) will be sufficiently conservative only when β_t is fairly low (< 0.024). From Equation (B7), this in turn implies that Constraint (B) will be sufficiently conservative whenever ρ is less than about 25%. It should be noted that levels of ρ less than the limiting values derived here (75% and 25%) are also required by Equation (9) ($B_t \leq B_3$) for certain ranges of r_3 values. The ratio of B_3 to B_2 is given (for arbitrary r_3) by

(B9) $$\frac{B_3}{B_2} = \left(\left(\frac{r_3+1}{r_3+2}\right) \times \left[2 - r_3\sqrt{\frac{3}{(r_3+1)(r_3-1)}}\right]\right)^{1/r_3},$$

which is less than 0.75 for $r_3 < 5.482$ and less than 0.25 for $r_3 < 2.222$.

Biological Reference Points for Canadian Atlantic Gadoid Stocks

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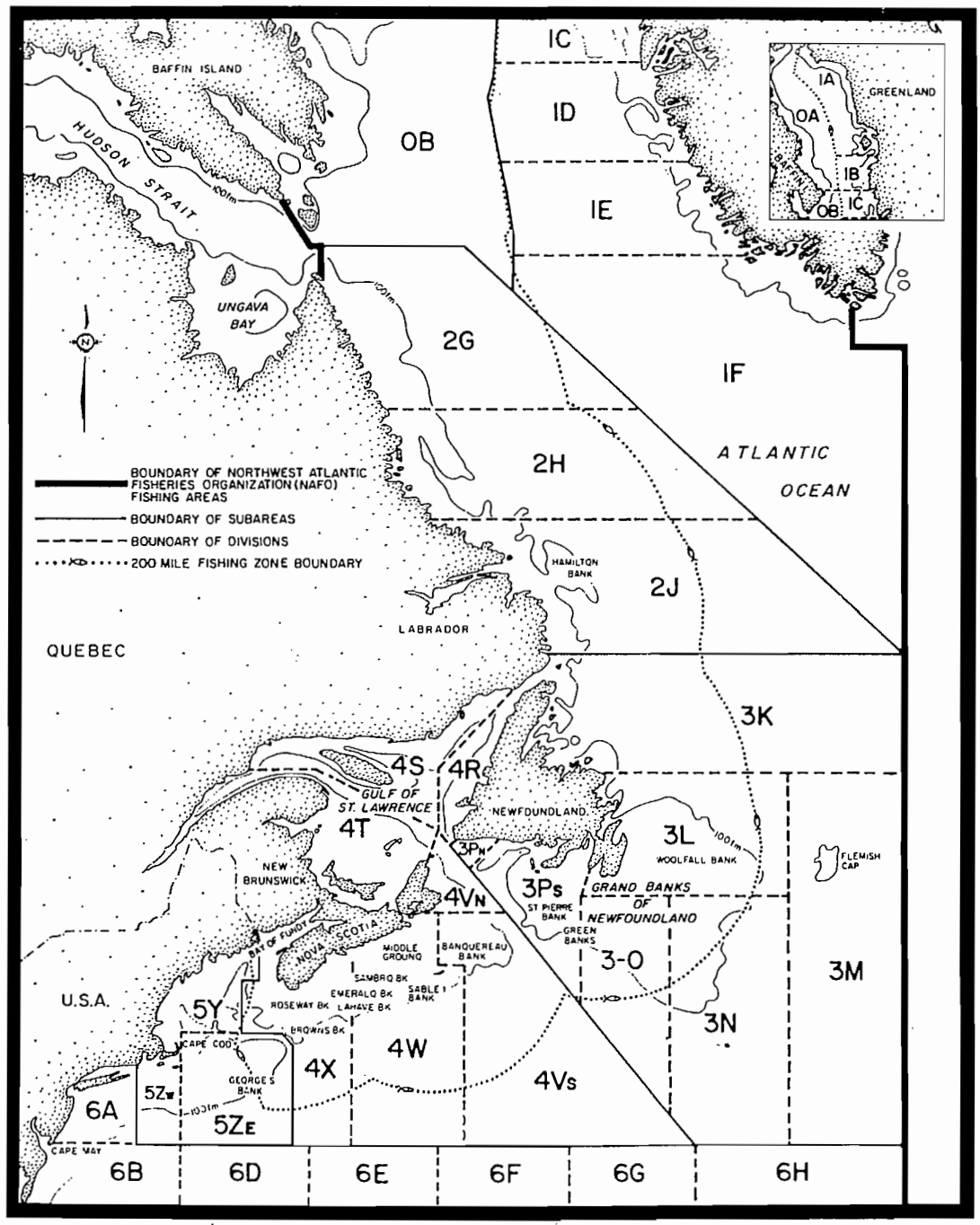
Maguire, J.-J. and P. M. Mace. 1993. Biological reference points for Canadian Atlantic gadoid stocks. p. 321–331. *In* S. J. Smith, J. J. Hunt and D. Rivard [ed.] Risk evaluation and biological reference points for fisheries management. Can. Spec. Publ. Fish. Aquat. Sci. 120.

Various biological reference points are used to guide the management of fisheries. They are usually expressed as fishing mortality rates or critical levels of biomass (spawning or other), and may be derived from yield-per-recruit (YPR), spawning-per-recruit (SPR), stock-recruitment (*S-R*), stock production relationships, or on species interaction considerations (e.g., cod-capelin). Reference points from YPR, SPR and *S-R* relationships were described, computed and compared for 11 Canadian Atlantic coast gadoid stocks (cod: 2J3KL, 3Ps, 3Pn4RS, 4T-Vn(Jan.–Apr.), 4VsW, 4X, 5Zj,m; haddock: 4TVW, 4X, 5Zj,m; pollock: 4VWX5Zc). In all cases, recent fishing mortality rates have exceeded $F_{0.1}$, and in most cases they have exceeded the level that has been identified with recruitment overfishing. It is suggested that fishing mortality rates need to be reduced by at least 50% for most stocks. The results also indicate that several of the stocks are depleted, and that some are near or below a critical threshold spawning biomass.

Différents points de référence biologiques sont utilisés pour orienter la gestion des pêches. Ils sont en général exprimés sous forme de taux de mortalité par pêche ou de niveaux critiques de la biomasse (de reproducteurs ou autre), et peuvent être dérivés des relations rendement par recrue (YPR), ponte par recrue (SPR), stock-recrutement (*S-R*), stock-production, ou à partir de considérations sur l'interaction entre les espèces (par exemple, entre la morue et le capelan). Les points de référence découlant des relations rendement par recrue, ponte par recrue et stock-recrutement ont été décrits, calculés et comparés pour 11 stocks de gadidés de la côte atlantique canadienne (morue de 2J3KL, de 3Ps, de 3Pn4RS, de 4T-Vn (janvier à avril), de 4VsW, de 4X, de 5Zj,m; aiglefin de 4TVW, de 4X de 5Zj,m; goberge de 4VWX5Zc). Dans tous ces cas, les taux de mortalité par pêche récents dépassaient le niveau de référence $F_{0.1}$, et dans la plupart des cas, ils étaient supérieurs au niveau identifié comme la surpêche du recrutement. Les résultats de notre analyse portent à croire qu'il faut réduire d'au moins 50 % le taux de mortalité par pêche de la plupart des stocks. Les résultats indiquent également que plusieurs stocks sont affaiblis, et que certains sont proches du seuil critique de la biomasse de reproducteurs ou inférieurs à ce seuil.

Prior to the late 1800's–early 1900's, fisheries were seen as inexhaustible resources (Smith 1988). However, with the introduction of ever more efficient harvesting technologies typified by large purse seiners and steam trawlers, intense discussions on the effect of fishing took place which led to the creation of the International Council for the Exploration of the Sea (ICES) in 1902. The history of modern fishery science is therefore less than a hundred years old. In this short time, there have been numerous notable theoretical developments as well as a gradual accumulation of practical experience. The advent of sequential population analysis (Murphy 1966; Pope 1972) enabled routine age-structured assessments to be conducted. In the northwest Atlantic, age-structured stock assessments have now been conducted regularly on several groundfish and pelagic stocks for more than 20 years. Their accuracy and precision remains lower than desirable because of poor catch statistics and highly variable abundance indices, but they provide a useful source of information on population dynamics, recruitment trends and fishing mortalities.

In recent years, management of Canadian Atlantic fish stocks has been based primarily on a target fishing mortality rate of $F_{0.1}$ (Halliday and Pinhorn 1985), and has not explicitly included thresholds in the form of either maximum fishing mortality rates or minimum acceptable biomass levels. Rivard and Maguire (1993) reviewed the status of exploitation of several Canadian Atlantic coast groundfishes and they concluded that all were exploited substantially above the $F_{0.1}$ target. To attain fishing mortality rates of $F_{0.1}$, fishing effort and presumably fishing capacity may need to be reduced substantially. In this paper we evaluate alternative biological reference points (BRP's) and examine their consequences.



Subareas and Divisions of the NAFO Convention Area and limits of the Canadian fishing zone (East coast).

Fig. 1. Map of the east coast of Canada showing the stock-areas used by the Northwest Atlantic Fisheries Organization (NAFO).

Material and Methods

Various biological reference points have been used for the management of fisheries. They are usually expressed as fishing mortality rates (e.g., F_{msy}, $F_{0.1}$, F_{max}) or critical levels of spawning or recruited biomass (e.g., 20% of virgin biomass, Beddington and Cooke 1983), and may be derived from yield-per-recruit (YPR), spawning-per-recruit (SPR), stock-recruitment (S-R), stock production relationships, or based on species interaction considerations (e.g., capelin off Newfoundland and Labrador are harvested at a low exploitation rate in order to avoid depleting the food available to cod). Reference points considered here are those from YPR, SPR and S-R relationships. Each of the selected reference points was calculated for 11 Canadian Atlantic coast gadoid stocks (Cod: 2J3KL, 3Ps, 3Pn4RS, 4T-Vn(Jan.–Apr.), 4VsW, 4X, 5Zj,m; Haddock: 4TVW, 4X, 5Zj,m; Pollock: 4VWX5Zc), with stock areas based on the Northwest Atlantic Fisheries Organization (NAFO) system (Fig. 1).

Yield per Recruit (YPR): $F_{0.1}$ and F_{max}

Thompson and Bell (1934) provided the foundations for calculating the effects of different fishing intensities on yield, but the most comprehensive treatment was by Beverton and Holt (1957). The concept of yield-per-recruit was proposed because recruitment was unknown and unpredictable. By making the calculations on a per-recruit basis, it was possible to estimate the relative yields that could be obtained from various fishing mortalities. However, the lack of appropriate data to estimate current fishing mortality, combined with the belief in the 1940's and 1950's that controlling the size of fish caught was sufficient to regulate fisheries, delayed the application of YPR concepts to fisheries management. The first reference point used from yield-per-recruit calculations was F_{max}, the fishing mortality that maximises YPR, but $F_{0.1}$ was soon proposed as a more prudent, preferred reference point (Gulland and Boerema 1973). The calculations presented here used the implementation of the Thompson and Bell (1934) YPR described by Rivard and Joly (1984). A plus group was included by extrapolating the data for the last age included in the assessment out to about 200 ages (to ensure convergence). The stock specific data on weights-at-age and partial recruitment vectors were taken from the 1991 Canadian Atlantic Fisheries Scientific Advisory Committee (CAFSAC) Advisory Document (Anon. 1991). Generally, the weights were averaged over the most recent three years while the partial recruitments (PR's) were those used to make catch forecasts.

$F_{0.1}$ and F_{max} are often used as targets or goals of fisheries management strategies, although for some stocks it may be reasonable to use F_{max} as a threshold or upper limit (i.e., a reference point to be avoided rather than achieved).

Stock-Recruitment (S-R) Observations

The relationship between the amount of spawners and the abundance of the subsequent brood was researched intensively in the late 1960's and early 1970's (e.g., Murphy 1966; Cushing 1971, 1973; Cushing and Harris 1973; Garrod and Jones 1974). In many of these and subsequent studies, the abundance of recruits appeared to be independent of the size of the spawning stock over the range of observations, although most researchers believed that recruitment was bound to be positively correlated with stock size at low stock sizes. Recruitment is notoriously variable and it is difficult to obtain a statistically significant fit to a traditional stock-recruitment (S-R) relationship such as those of Ricker (1954) and Beverton and Holt (1957). An alternative approach is to consider the probability of obtaining strong, average or poor recruitment at various spawning biomass levels (e.g., Hennemuth et al. 1980; Sissenwine et al. 1988). Serebryakov (1990) has proposed an interesting variation on this theme which explicitly recognizes that variable environmental conditions affect recruitment. Serebryakov estimated the effect of the environment by examining survival (R/S) ratios. He originally proposed definitions in terms of fecundity but they can be translated into terms of biomass as:

- **Safe Spawning Stock Biomass** is the biomass at which average survival gives a strong year-class. It is calculated by dividing the abundance of a strong year-class by the mean survival rate (R/S).

- **Minimum Spawning Stock Biomass** is the biomass at which average survival gives an average year-class, high survival gives a strong year-class and low survival gives a poor year-class. It is estimated by dividing the abundance of an average year-class by the average survival.

- The Critical Population Biomass of Serebryakov would probably be more appropriately called the **Dangerous Spawning Stock Biomass** and it is the biomass at which high survival is required to produce an average year-class. It is estimated by dividing the abundance of an average year-class by the highest survival rate.

Estimates of spawning biomass and recruitment were obtained from the researchers at the Department of Fisheries and Oceans, Canada, who were involved in the assessment of the stocks (2J3KL and 3Ps cod — C. Bishop; 3Pn4RS cod — D. Gascon; 4T-Vn(J.–A.) cod — M. Hansen; 4X cod — J. Hamel; 5Zj,m cod — J. J. Hunt; 5Zj,m haddock — S. Gavaris; 4VWX5Zc pollock — C. Annand. pers. comm.). These estimates were unavailable for 4VsW cod, 4TVW haddock and 4X haddock because analytical assessments were not presented in 1991. In those cases, virtual population analysis (VPA; Pope 1972) was run with an arbitrarily high fishing mortality in the terminal year and a PR vector was obtained by applying separable VPA (Pope and Shepherd 1982) or by iteratively averaging PR at age for the last five years. For these three stocks, only those estimates that were relatively insensitive to the terminal fishing mortality were retained (i.e., estimates from the most recent few years were discarded).

In order to calculate the three biomass thresholds of interest, it is necessary to know the average abundance of a year-class, the average survival rate (recruits/spawning stock biomass), the highest survival rate and the abundance of a strong year-class. All these quantities are straightforward to identify except the abundance of a strong year-class which was defined to correspond approximately to the 75th percentile of the distribution of year-class sizes.

The safe spawning stock biomass could be considered a target or goal of management. The minimum spawning biomass should be a threshold at which caution needs to be exercised. The dangerous spawning biomass is a critical threshold at which remedial action is urgently needed.

Spawning Stock Biomass Per Recruit (SPR)

There are two usages that can be made of recruits: they can be fished or they can be allowed to become spawners (Sissenwine and Shepherd 1987). The concept of SPR is analogous to YPR but examines the question of reference points from a different perspective (how much is left rather

than how much is taken). Whatever the number of recruits, if they are fished too hard few will survive to become spawners, regardless of the relationship between stock and recruitment. SPR is at a maximum when fishing mortality is zero and decreases monotonically as fishing mortality increases. Reference points from SPR can be expressed as a percentage of the maximum SPR (%SPR) and used as targets or thresholds in fisheries management strategies (Mace and Sissenwine 1993).

However, until Shepherd (1982) there was no basis for selecting one level of %SPR over another. Shepherd developed a method for combining stock-recruitment data with SPR relationships and suggested that this method could be used to derive biological reference points. The concept was further detailed and expanded in Sissenwine and Shepherd (1987) who introduced the concept of the replacement fishing mortality rate, F_{rep}:

> ... [*S-R*] data points that lie on a replacement line correspond to year-classes that are the exact size to replace the spawning biomass of their parents when subjected to a [constant] level of fishing pressure that corresponds to the line. Points above the replacement line are year-classes that produce a surplus spawning biomass; points below the line are year-classes that do not produce enough. For a population to persist, the points below the line must be balanced by the points above it. (Sissenwine and Shepherd 1987, p. 916.)

Sissenwine and Shepherd (1987) proposed estimating F_{rep} from the median survival ratio (the straight line through the origin that bisects the *S-R* observations), and referred to this estimate as F_{med}. The computational details required to obtain F_{med} are specified in Gabriel et al. (1989) and a detailed discussion of the replacement concept is given both there and in Mace and Sissenwine (1993).

Two other percentiles of the survival ratios are also calculated routinely by at least one stock assessment group (ICES): the lower 10th percentile and the 90th percentile (corresponding respectively to F_{low} and F_{high}; ICES 1983, 1984). The rationale for use of these two points is that F_{low} represents a fishing mortality rate at which the population is likely to increase, while F_{high} represents a fishing mortality rate that is expected to drive the stock down. These two reference points are not discussed in detail here since if F_{med} is a reasonable estimate of F_{rep} (and conditions of stationarity apply), any fishing mortality rate lower than F_{med} will result in a tendency for stock size to increase relative to the average of the observations (although not without limit since $F = 0$ corresponds to the equilibrium virgin condition and F cannot be negative), and any fishing mortality rate exceeding F_{med} will result in a tendency for stock size to decline.

Estimates of recruitment, spawning stock biomass, partial recruitment and weights-at-age were obtained as described in the previous sections. Maturity ogives were provided by the researchers involved in the stock assessments. The spawning biomass per recruit calculations were based on the standard model (Beverton and Holt 1957), and followed the computational procedure outlined in Gabriel et al. (1989) except for the use of mean annual spawning biomass rather than beginning of year or spawning season projections. A plus group was included in the SPR computations using the same method as that outlined above for YPR.

Sissenwine and Shepherd (1987) recommended that F_{rep} (and its estimate, F_{med}) be treated as a recruitment overfishing threshold and, in most cases, it has been used in this way rather than as a fishing target (particularly in U.S. Fishery Management Plans). Mace and Sissenwine (1993 considered F_{med} to be an estimate of F_{τ}, the fishing mortality rate associated with extinction. However, if the *S-R* observations used to calculate F_{med} are taken from a new fishery, F_{med} may be an overly-conservative overfishing threshold. If the *S-R* observations used to calculate F_{med} are more or less restricted to a range of stock sizes considered "near-optimal", F_{med} may even be a valid fishing mortality target.

Results

F and %SPR Reference Points

Wide variation in partial recruitment patterns (e.g., the steepness of the ascending limb and whether PR is domed or flat-topped) and differences in both the age and size used for the first age of recruitment make it difficult to compare fishing mortality rates across stocks. For flat-topped PR's, it is usual to assign all the fully-recruited ages a PR of 1, and scale the PR's for younger ages accordingly; but for domed PR's there does not appear to be a standard convention. In the latter case, we calculated a reference F based on the unweighted average of the age classes represented most strongly in the commercial catch (usually starting near the age of 50% recruitment and continuing to the age where the numerical contribution to the exploited stock would be less than 5%, about 5–7 age classes later, Table 1), and normalised all other PR's to this reference. However, this only partially alleviates the problem of a lack of comparability of fishing mortality rates among stocks.

The replacement SPR (the level of SPR corresponding to F_{rep}) as estimated by the median survival ratio (Fig. 2) is independent of PR; however, replacement SPR is dependent on the assumed first age of recruitment. It is common to divide the replacement SPR (and any other level of SPR) by its maximum ($SPR_{F=0}$) in order to "normalise" it for comparison among stocks, the result being referred to as the replacement %SPR (Mace and Sissenwine 1993).

Here, we focus on within-stock comparisons of alternative fishing mortality reference points and between-stock comparisons of %SPR levels.

There was only one case (5Zj,m haddock) where F_{med} was less than $F_{0.1}$ but 5 of 10 cases where F_{med} was less than F_{max} (Table 1). Thus, if F_{med} is used as a recruitment overfishing threshold, it would not always be wise to use F_{max} as a fishing target. However, F_{med} may sometimes be a conservative estimate of the recruitment overfishing threshold, particularly when the *S-R* observations exhibit pronounced

Table 1. Reference levels of fishing mortality, associated levels of % spawning per recruit (%SPR) and recent fishing mortalities for 11 gadoid stocks. For each stock, fishing mortality rates have been normalised relative to the unweighted average partial recruitment (PR) over the reference ages (last column). Reference ages expressed as a range indicate that the PR was dome-shaped, whereas a "+" indicates it was flat-topped.

Stock	$F_{0.1}$	F_{max}	F_{med}	approx. F'_{91}	YPR at $F_{0.1}$	YPR at F_{max}	%SPR at $F_{0.1}$	%SPR at F_{max}	%SPR at F_{med}	reference ages
2J3KL COD	0.17	0.41	0.38	*	0.63	0.69	28.7	9.2	10.6	5–11
3Ps COD	0.20	0.34	0.53	0.4–0.5	0.77	0.82	29.2	15.2	8.6	5–11
3Pn4RS COD	0.23	1.10	0.40	0.3	0.62	0.70	30.9	7.1	18.8	6–12
4T-Vn (J.–A.) COD	0.14	–	0.5	0.24	0.36	–	35.8	–	22.7	9+
4VsW COD	0.14	0.42	0.91	1.0	0.46	0.51	37.9	13.7	6.6	5–11
4X COD	0.14	0.26	0.48	0.44	1.16	1.25	41.4	26.1	15.0	5+
5Zj,m COD	0.15	0.28	0.36	1.0	1.60	1.73	38.6	22.5	17.4	3+
4TVW HADDOCK	0.21	0.44	0.48	0.7–0.8	0.50	0.55	36.9	20.2	18.8	5–10
4X HADDOCK	0.25	0.74	0.55	1.0	0.55	0.63	43.1	22.6	27.1	6+
5Zj,m HADDOCK	0.25	0.72	0.16	1.0	0.69	0.79	34.8	13.1	47.4	3+
POLLOCK	0.22	0.61	0.55	0.35	1.06	1.18	34.8	15.6	17.2	4–11

* A large decrease in abundance between 1990 and 1991 implies that total mortality (Z) exceeded 1.0. It is believed that natural mortality (M) probably increased in 1991, but the 1992 assessment assumed $M = 0.2$, therefore assigning the remaining mortality to F.

concavity (Sissenwine and Shepherd 1987). Visual examination of Fig. 2 suggests that 3Pn4RS cod and 4T-Vn cod have the highest degrees of concavity; these two stocks also have the highest levels of replacement %SPR (%SPR at F_{med}) of all the cod stocks (and amongst the lowest values of F_{med}).

Yield per recruit for cod followed the expected north to south cline with the largest YPR obtained for the fast-growing 5Zj,m cod (Table 1). The difference would be even more pronounced if a standard age of recruitment was used (age of recruitment also decreases in a north-south direction). There was also a general trend of declining median survival (R/S) going from north to south. When a correction for age of recruitment was made, it was apparent that early-maturing cod stocks (4X cod and 5Zj,m cod) had the lowest survival ratios.

There is little increase gained in YPR by fishing at F_{max} compared with $F_{0.1}$ but a large increase in fishing mortality (and presumably a proportional increase in fishing effort) and a considerable decrease in %SPR (Table 1). Estimates of %SPR at $F_{0.1}$ and F_{max} were much more consistent (ranging from about 29–43% and 7–26%, respectively) than the replacement %SPR (which ranged from 7–47%). The mean replacement %SPR (based on arcsine transformed data) for the cod stocks was 13.8% ($N = 7$) compared to 30.5% ($N = 3$) for haddock. This suggests that cod may be more resilient to fishing than haddock; however, the haddock results have high variance.

With the exception of pollock, recent fishing mortality levels have been near or above F_{med}.

Biomass Reference Points

S-R plots were divided along both axes into weak, average and strong classes (Fig. 3). Relationships with a positive correlation between recruitment and spawning biomass can be identified for 2J3KL cod, 3Ps cod, 4VsW cod, 4TVW haddock and pollock. The most striking example is 2J3KL cod, but the relationship is dependent on the first seven points in the series (1962–68), and may therefore be spurious due to inadequate sampling of several of the important fleets in those early years. The number of recruits appears to be negatively correlated with spawning biomass for 3Pn4RS cod, but there are few data and the quality of the catch data is known to be poor in some of the years. For 4T-Vn (J.–A.) cod, recruitment seems to be correlated with spawning biomass, but the interpretation of the time series as a whole is complicated by the substantial improvement in survival rates that apparently occurred during the period 1973–80 (Fig. 4). All stocks (except 4X cod) for which the time series are reasonably long show a higher frequency of average to strong year classes at intermediate or high levels of spawning biomass (Fig. 3). This suggests that one advantage of avoiding low levels of spawning biomass is improved recruitment.

The S-R plots (Fig. 3) show that the values calculated for safe spawning biomass and minimum spawning biomass (Table 2) are generally supported as producing good recruitment. For most stocks, dangerously low levels of spawning biomass have been approached or breached during the last 30 years. Comparison with recent estimates of biomass suggests that six stocks (2J3KL cod, 3Pn4RS cod, 4VsW cod, 4TVW haddock, 4X haddock and 5Zj,m haddock) may be dangerously low, two (3Ps cod and 5Zj,m cod) are at minimum levels, and the rest are "safe". Comparisons of year-class survival between neighbouring stocks suggest that similar factors may be affecting the recruitment of cod in those stocks in the same years (Fig. 4). For example, the St. Pierre Bank (3Ps), Northern Gulf of St. Lawrence (3Pn4RS), Southern Gulf of St. Lawrence (4T-Vn), and the Eastern Scotian shelf (4VsW) cod stocks all experienced above-average survival during the

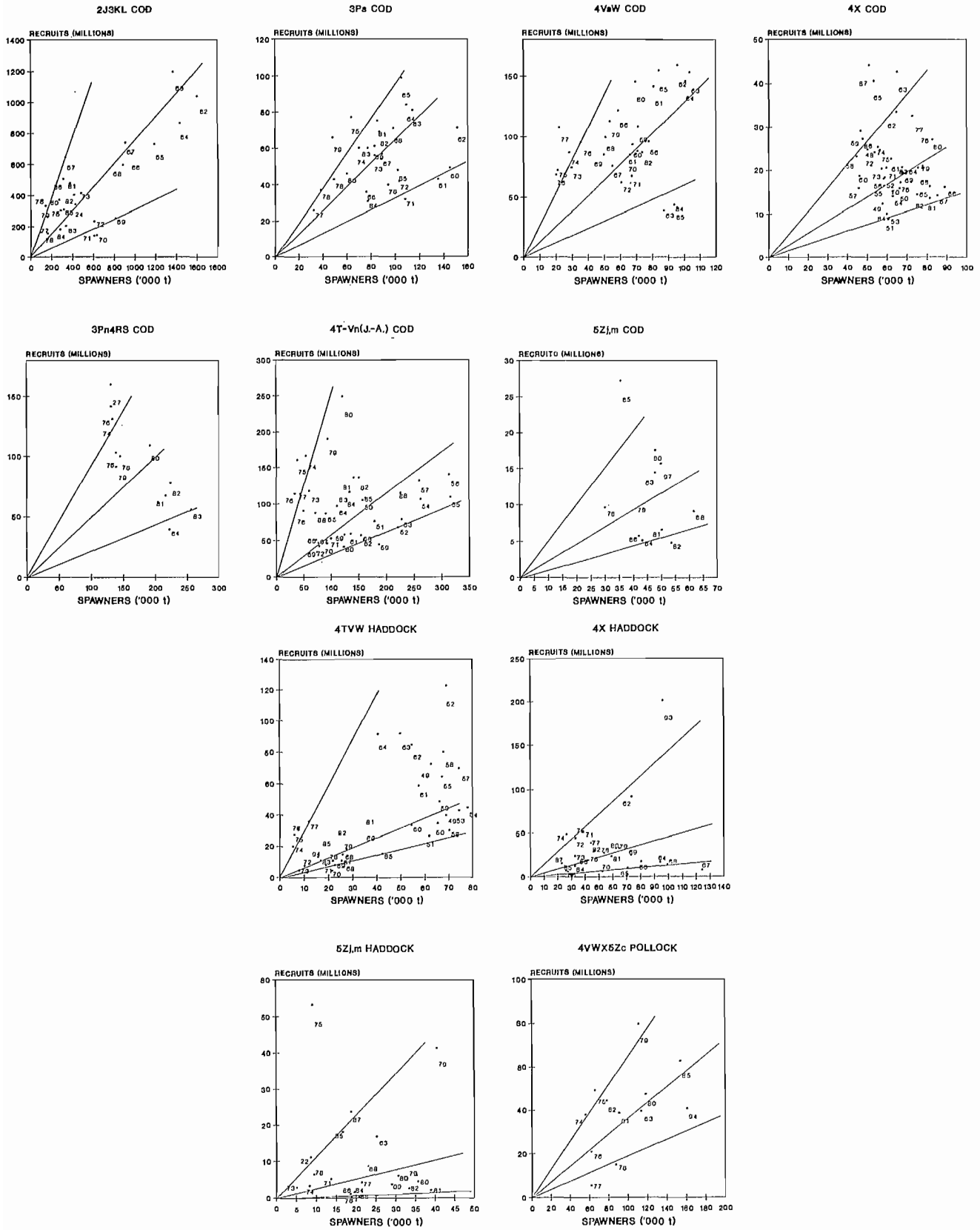


Fig. 2. Stock-recruitment (S-R) scatterplots for 11 Canadian Atlantic gadoid stocks. Lines passing through the origin correspond to F_{high} (upper line, representing the 90th percentile of the survival ratios), F_{med} (middle line, representing the median survival ratio) and F_{low} (bottom line, representing the 10th percentile of the survival ratios).

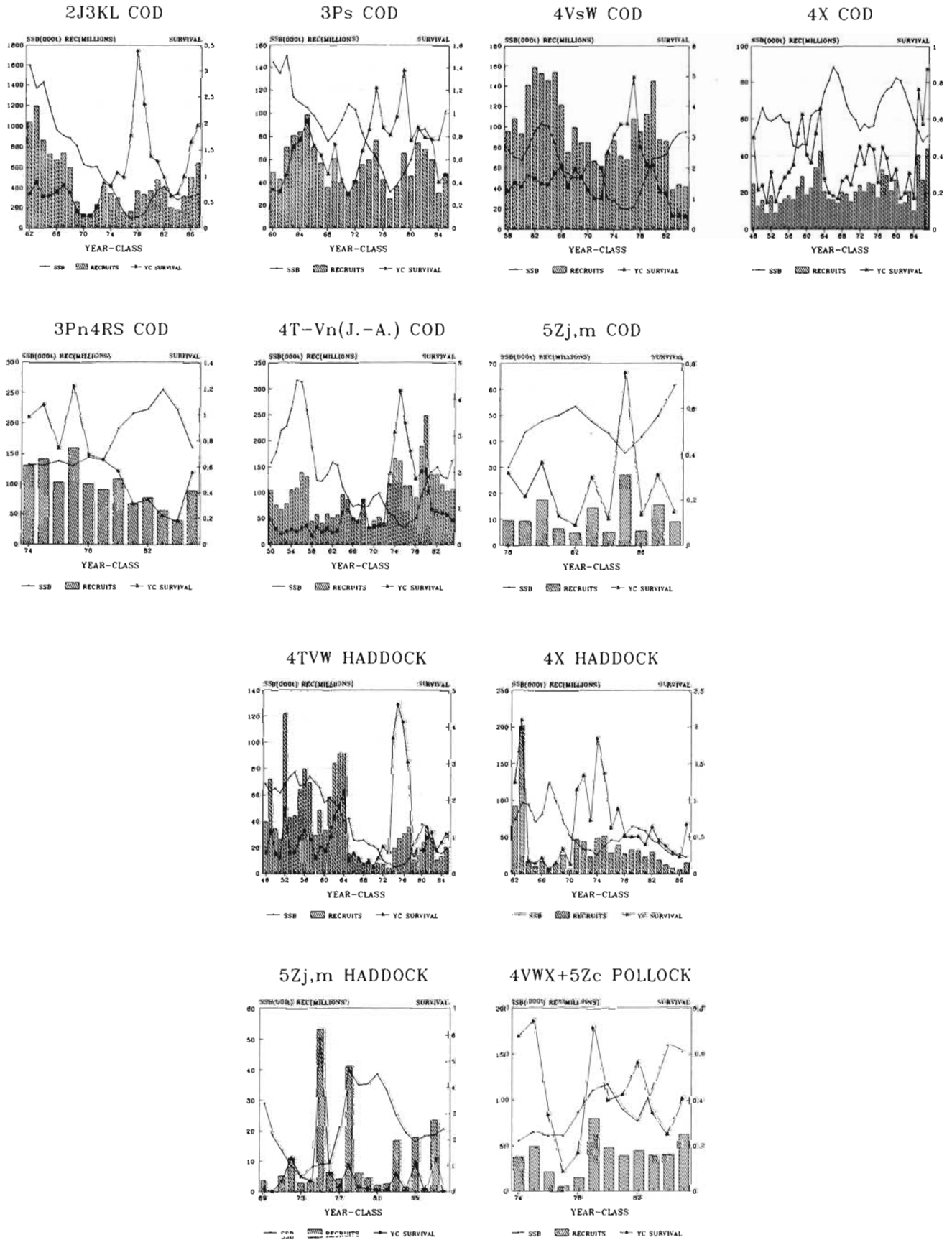


Fig. 3. Trajectories of spawning stock biomass (SSB), recruitment (REC) and survival (R/S) for 11 Canadian Atlantic gadoid stocks.

Table 2. Critical levels of spawning stock biomass ('000 t) derived by the method suggested by Serebryakov (1990) compared to the estimated spawning stock biomass for the most recent year available (year in brackets).

	Spawning stock biomass			
STOCK	Safe	Minimum	Dangerous	Recent
2J3KL COD	388	291	89	70–110 ('91)
3Ps COD	100	80	41	76 ('90)
3Pn4RS COD	175	154	80	55 ('92)
4T-Vn (J.–A.) COD	116	93	23	114 ('92)
4VsW COD	61	53	20	27 ('91)
4X COD	70	59	24	84 ('91)
5Zj,m COD	57	42	14	41 ('91)
4TVW HADDOCK	53	33	8	19 ('85)
4X HADDOCK	61	52	16	23 ('87)
5Zj,m HADDOCK	28	14	2	8 ('92)
POLLOCK	117	93	54	164 ('91)

years 1974–79. Similarly, survival of the 1985 year-class of cod was exceptionally high on both Browns Bank (4X) and Georges Bank (5Zj,m).

Discussion

The problem with fishing strategies that are based on target biological reference points but do not specify thresholds is that the consequences of not achieving the target in a given year may be difficult to evaluate, and the need and degree of urgency for remedial action may not be clear. For example, fishing mortalities that exceed a target such as $F_{0.1}$ for a few years may not jeopardize the stock during periods of favourable environmental conditions resulting in high survival and high recruitment, but may seriously deplete the stock if conditions are poor. The advantage of specifying thresholds (either as maximum F or minimum biomass or both) in addition to targets is that the degree of urgency for action can be judged by the proximity of the current stock condition to the threshold relative to the target.

If $F_{0.1}$ is used as the target and F_{med} as a threshold, then it would appear that all stocks except pollock are closer to the threshold than the target (Table 1). In fact, pollock is the only stock where the current F is substantially below the threshold; in all other cases, the current F is near or above F_{med}. The most extreme case is 5Zj,m haddock, where the current F is estimated to be about six times F_{med} (and four times $F_{0.1}$). For half of the remaining stocks, fishing mortality needs to be reduced by about 50% to bring it down to the level of F_{med}. For the other half, current F is apparently quite similar to F_{med}; however, it must be kept in mind that thresholds should be treated as reference points to avoid, not as targets. The latter point is probably even more important for thresholds based on biomass levels. The worst situation is that where both the biomass threshold and the fishing mortality threshold have been breached. If the "dangerous spawning biomass" (Table 2) is used as the threshold, then it appears that 2J3KL cod, 3Pn4Rs cod, 4VsW cod and all three haddock stocks are in this situation. If the "minimum spawning biomass" (Table 2) is used, then 3Ps cod and 5Zj,m cod would be added to the list. These stocks urgently require reductions in fishing mortality so that they can rebuild.

In situations where the threshold F has been exceeded but current biomass is at a "safe" level, it may be possible to spread the necessary reductions in fishing mortality over several years. Alternatively, if biomass is critically low but fishing mortality is below the threshold, the stock should eventually rebuild even without reductions in fishing mortality, although it may be prudent to reduce F so as to hasten the rebuilding.

Of course, the validity of these conclusions is contingent on the validity of the estimates of thresholds. The validity of using F_{rep} as a recruitment overfishing threshold has already been discussed by Mace and Sissenwine (1993) who concluded that, under conditions of stationarity, F_{rep} was more likely to be a conservatively-biased estimate of F_r, the extinction threshold, than a liberal estimate (due to the fact that most sets of S-R observations exhibit some degree of concavity). However, an estimate of F_{rep} based on observations made during one time period may not necessarily apply to a future time period with a different environmental regime that results in markedly higher or lower survival rates, and the actual bias could be in either direction. There is also the question of the validity of F_{med} as an estimate of F_{rep}; although the question has been mentioned (e.g., Sissenwine and Shepherd 1987), it has never been fully addressed.

It was suggested above that the estimates of F_{med} for 3Pn4RS cod and 4T-Vn cod may be conservative estimates of overfishing thresholds due to the pronounced concavity in the S-R data (Fig. 2). Closer examination of the data shows that there are time trends in survival (Fig. 4) that could bias the estimates in the opposite direction. Survival has declined almost monotonically throughout the time series available for 3Pn4RS cod and for short-term projections it may be better to base the estimate of F_{med} on the last few observations only. For 4T-Vn cod, the period of high survival during 1973–80 brings the overall estimate of F_{med} up, possibly biasing it high

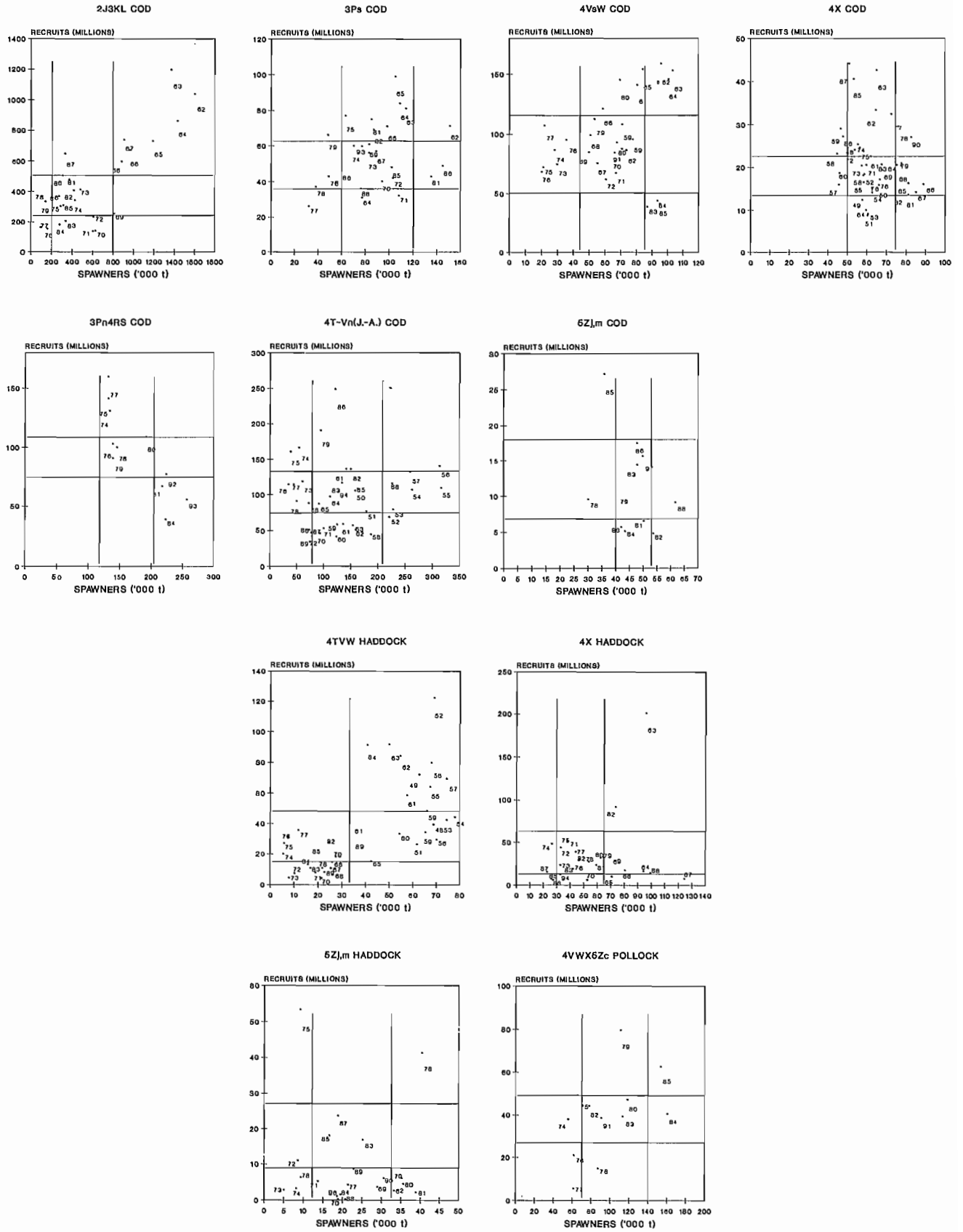


Fig. 4. Stock-recruitment (S-R) scatterplots for 11 Canadian Atlantic gadoid stocks. The recruitment axis has been partitioned into strong, average and weak year-classes (see text for method).

relative to current or future short-term environmental conditions. Similarly, the overall estimate of F_{med} is extremely high for 4VsW cod because survival has been high throughout most of the period of observation, but survival rates for the three most recent years (1983–85) will not allow stock replacement even with F as small as F_{low} (Fig. 2).

The time trend in estimates of F_{med} is both a curse and a blessing. It means that the estimates are never stable, and particularly for small datasets, they could change substantially from one stock assessment to the next. However, since there is no way of avoiding the fact that survival is autocorrelated, the data can be partitioned to make different estimates of F_{med} for different purposes. For example, an estimate based on the entire dataset might be used for long-term projections or simulations, and one based on recent years alone might be used for short-term projections. This is likely to be an improvement over using an alternative, stable F that ignores trends in survival and recruitment.

Another way of judging the degree of conservatism of the estimates of F_{med} is to compare them with the corresponding estimates of %SPR for similar stocks. Mace and Sissenwine (1993) had few northwest Atlantic cod stocks in their database, so we did a comparison with northeast Atlantic stocks (ICES areas). The mean replacement %SPR (based on arcsine transformed data) for NW Atlantic cod stocks was 13.8% ($N = 7$; ±2SE = 9.6–18.6%) compared to 6.4% ($N = 12$; ±2SE = 4.4–8.7%) for NE Atlantic cod; for haddock the corresponding values were 30.5% ($N = 3$; ±2SE = 15.3–48.3%) and 22.1% ($N = 4$; ±2SE = 13.5–25.4%). Differences between the two groups of cod stocks were significant ($p < 0.01$), while those for haddock were not. This could either mean that NW Atlantic cod stocks are less resilient than NE Atlantic cod stocks or else that, as a group, they have been exploited less heavily so that the estimates of F_{med} are more conservative estimates of overfishing thresholds. In any case, the mean for NW Atlantic cod is still well below the mean across all stocks of about 19% (Mace and Sissenwine 1993), reflecting the fact that cod generally have high resilience to fishing. Estimates for haddock are highly variable on both sides of the Atlantic, reflecting the fact that haddock have high variance in recruitment and survival.

The validity of Serebryakov's (1990) method for estimating critical levels of biomass has not been examined in detail but it will obviously also be influenced by the range of stock sizes covered by the observations, and it is likely that this influence will be even stronger and more problematic. "Safe" could be "dangerous" if the observations are all taken from a depleted or collapsing stock, and "dangerous" could be "safe" if the observations are all taken from a new fishery. To obtain the most robust and meaningful estimates for Serebryakov's method requires the largest possible contrast in spawning stock size, whereas the validity of F_{med} as a recruitment overfishing threshold is generally greater for cases where observations are restricted to low stock size where the S-R data can be adequately represented by a straight line through the origin (the range where Serebryakov's reference levels of biomass may all be overly optimistic). These contrasting data requirements must be kept in mind when applying the methods and interpreting the estimates. Based (subjectively) on the degree of curvilinearity in the S-R observations (Fig. 3), it appears that Serebryakov's estimates of critical biomass levels may be the most optimistic for 2J3KL cod, 3Ps cod, 4VsW cod, 4TVW haddock and pollock. This list includes four of the six stocks classified as near or below the dangerous spawning biomass (Table 2).

Another problem is that estimates of current fishing mortality and current biomass are often very imprecise. This means that a strategy of "close the fishery immediately if the threshold is violated" would probably not be practical because some individuals would use the uncertainty to argue against closure, and in some cases they may be justified since the estimates could go up or down retrospectively. It may be necessary to invoke a criterion such as "the fishery will close when the probability of violating the threshold exceeds X", or to use a running average of the most recent few years, or to adopt a strategy that gradually reduces F towards zero as the threshold is approached and crossed.

For the stocks considered in this paper, estimates of yearly recruitment from the early 1960's or before are available from stock assessments based on sequential population analysis. Assuming that the past variability in recruitment is a good indication of future variability, and provided that human activities or natural changes do not interfere to any great extent with productivity, it is easy to provide upper and lower bounds on likely future catches as well as the relative frequencies of various catch, F and biomass levels. The probability of violating biomass or fishing mortality-based thresholds can then be evaluated and compared for a range of alternative management strategies. Knowing the biological limits of the system, management bodies and the fishing industry would be in a better position to make decisions on long term strategies with regards to vessel size, fleet size, number of plants, production schedules and related factors. Decisions on the targets or goals of fisheries management strategies can be based on sociological, economic and political criteria on a case by case basis taking into account the risks that the people involved in the industry and government are willing to take for each resource.

However, biological criteria should take precedence when stocks become severely stressed, as indicated by a fishing mortality rate that exceeds a threshold maximum, or a spawning biomass that has fallen below a threshold minimum, or both. For most of the stocks considered here, the indications are that fishing mortality needs to be decreased substantially to rebuild depleted stocks or to prevent further declines from occurring.

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Stochastic Locally-Optimal Harvesting

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Newly advanced theory is presented for the optimal regulation of discrete-time, non-linear, stochastic systems. The theory relies on local, second order approximations of the dynamic programming equation, trials of which have proved satisfactory in all tests to date. Certainty equivalence is established and an optimal stochastic control rule is derived to determine fishing mortality rates. Extreme or "bang-bang" solutions exist only for special cases and they are unrepresentative of the generality of optimal solutions. The theory is applicable to the harvesting of single species, or of multiple interacting species with one or more fishing fleets and to pest control. Examples are provided of optimal regulation of a) a population with logistic dynamics, b) a multi-cohort fish stock and c) a predator-prey-pest complex. The concept of risk-sensitive optimal control is introduced.

Nous présentons une nouvelle théorie pour la régulation optimale des systèmes stochastiques non linéaires à temps discret. La théorie se fonde sur les approximations locales de second ordre de l'équation de programmation dynamique, dont tous les essais réalisés jusqu'à maintenant ont donné des résultats satisfaisants. Nous établissons l'équivalence de certitude, et nous en tirons une règle de contrôle stochastique optimal pour calculer les taux de mortalité par pêche. Les solutions extrêmes ou «tout ou rien» n'existent que pour les cas spéciaux, et ne sont pas représentatives de l'ensemble des solutions optimales. La théorie s'applique à l'exploitation d'une seule espèce, ou d'espèces multiples en interaction avec une ou plusieurs flottilles de pêche, et à la lutte contre les nuisibles. Nous donnons des exemples de régulation optimale pour (a) une population à dynamique logistique, (b) un stock de poissons à plusieurs cohortes et (c) un complexe prédateur-proie-nuisible. Nous présentons le concept de contrôle optimal sensible au risque.

A problem posed in harvesting studies is, given the known dynamics of a population or complex of interrelated populations, how should it or they be annually harvested so as to maximize some bioeconomic criterion?

Various approaches have been adopted to either solving or providing realistically sensible solutions to that problem. The maximization of yield per recruit (Beverton and Holt 1957) provides an answer to a special case: that of a constant recruitment to a multi-cohort stock, harvested at a constant rate. For many fisheries this still provides perhaps the most useful perspective for management, but it largely neglects the inherent properties of dynamic systems. A constant recruitment model was considered by Clark et al. (1973) who calculated the pattern of fishing mortalities necessary to maximize an integrated, time-discounted yield: the results indicated that a pulsed, rather than a constant, fishing gave optimal yields. Reed (1980) considered the case in which recruitment into the population was density dependent with subsequent survival rates being independent of density. Maximization of the integrated discounted yield was shown to be by harvesting at most two age-classes. Such a result was anticipated by Beddington and Taylor (1973).

Use of the simple, continuous-time, logistic population model, with a linear harvesting and linear costs, shows the general characteristics of the optimal solutions of many real harvesting problems. The solutions tend to be to take the stock or harvesting rate to the optimal equilibrium as quickly as possible. The equilibrium is sensitive to the discount rate. If no discount rate is assumed then the equilibrium of this model is at MEY (maximum economic yield). As the discount rate increases, so the optimal equilibrium is found at a lower stock size and higher fishing rate. At an infinite discount rate the optimum is found at the point of zero marginal return, which for this linear model is at zero current profit. Clark (1973, 1976) showed that in cases wherein the discount rate was large relative to gross rates of profit (a function of population growth rate and economics) the best return could be achieved by exterminating the resource and placing the implied, realized profit elsewhere.

The introduction of stochastic elements adds realism but considerably complicates the mathematics. Before one considers optimal solutions to the stochastic problems the consequences of the introduction of stochasticity warrant a mention. The presence of such noise introduces a risk into any managed system. Constant catch policies, at anything other than relatively low catches, result in risks of depleting the stocks to levels from which it may not recover (Beddington and May 1977). Constant effort policies are generally much safer, but even so relative variance of stock and yield are likely to increase with increasing effort (May 1973, May et al. 1978), although this relationship is sensitive to the structure of the dynamic population processes and the source of the noise (Shepherd and Horwood 1980; Horwood and Shepherd 1981; Horwood 1983; Getz and Haight 1989). Reed

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(1978) confirmed that for a density-dependent, single cohort model, subjected to a multiplicative noise, the expected yield was less than that from the equivalent deterministic model; the converse follows that a yield as high as the deterministic MSY (maximum sustainable yield) cannot be sustained.

Many of the deterministic optimization problems can be solved analytically because they are linear or else are special cases, and their solution usually relies upon recourse to the Pontryagin maximum principle. No such general principle exists for stochastic problems, although Kushner (1965) developed solutions for a class of diffusion processes which have been extended by others. Particular stochastic problems in fisheries have been addressed in several studies (e.g.,Ludwig 1979; Mendelssohn 1978, 1980; Reed 1979, 1988; Abakukus and Prajneshu 1981; Charles 1983). The solutions all tend to be of a bang-bang character (i.e., a switching between maximal controls such as zero and maximum fishing effort), however this is more a feature of the restrictive nature of the tractable problems than a general property of optimal solutions. The above problems and solutions were reviewed by Clark (1976, 1985), Mangel (1985), Getz and Haight (1989), and Williams (1989). Not covered in the above, or in this article, is the problem of control or management of systems with imperfect information on the underlying dynamic structures. Although not covered here, it is evidently an important component of the management of real fisheries; May et al. (1978) concluded that "... what seems really needed (for management) is not further mathematical refinement, but rather robustly self-correcting strategies that can operate with only fuzzy knowledge about stock levels and recruitment curves". This problem has been considered by, for example, Walters (1986), Ludwig (1989) and Hilborn and Walters (1992).

In light of the obvious mathematical difficulties, more realistic problems of determining optimal fishing strategies have been solved numerically (e.g., Horwood 1987) or else control laws have been intuitively constructed and their properties analysed (e.g., Shepherd 1981; Magnusson 1991; Pelletier and Laurec 1992). Studies of the application of standard control theory to the management of fish stocks (Horwood et al. 1990, Jacobs et al. 1991) illustrate the importance of choosing a suitable control law and of tuning the control appropriately; the following example clearly demonstrates this point. The dynamics of a multi-cohort "mackerel" stock was computationally simulated in discrete time, assuming an increasing, monotonic stock and recruitment relationship with multiplicative noise and with errors in estimates of numbers at age. In this example, the objective chosen was to minimize, over long-time, the summed weighted relative variation in yield and in fishing effort (J) about a pre-chosen target equilibrium; in this example an equal weighting was given to relative fluctuations in catch and in fishing mortality or effort

$$J = \sum_t \left[\left(\frac{\Delta C}{C^*} \right)^2 + \left(\frac{\Delta F}{F^*} \right)^2 \right],$$

the implied management philosophy being to minimize year-to-year fluctuations in the catching and marketing sectors of the industry. A standard engineering control rule is the linear PID law, in which the control (e.g., fishing mortality rate) is determined from an appropriate combination of terms of **P**roportional, **I**ntegral and **D**erivatives of the instantaneous deviations in the state variable (e.g., the stock size relative to its ideal level) (Dorf, 1986). The approximate result of using this form of control on the mackerel model is illustrated in Fig. 1, and compared with others it was very ineffective, giving much higher costs (J) than other control systems. A non-linear control was developed as a minimization of a single-step process, and this involved a single parameter, μ, used to tune the weighting given to deviations in catch and $(1-\mu)$ to deviations in fishing effort; the objective still being to minimize J with an equal weighting given to deviations in catch and in effort. Figure 1 shows the cost, J, for various target fishing mortality rates (F^*) and values of the tuning parameter μ. For each target F^* a value for μ can be found that minimizes J. Two features are revealed by Fig. 1 and the envelope of optimal values of μ. First, that if the target F^* is high then, even to minimize an equal weighting of variance in catch and in effort, a constant effort policy is required (i.e., $\mu = 0$), and even at low values of F^* the optimal value of μ was about 0.75. This is because the effect of constant catches can be destabilizing, driving the stock to low levels (Beddington and May 1977). Second, that the difference in costs between the PID and the sub-optimal, non-linear control rules is substantial, as is the difference between costs of the best and worse choice of μ. It demonstrates that it may well be profitable to seek more completely optimal solutions.

The mathematics of the full deterministic bioeconomic problems are sufficiently difficult that general solutions are not to be expected, and this is even more true of the related stochastic problems. Clark (1985) stated that "... multidimensional dynamic optimization problems are notoriously difficult to solve and no complete solution has yet been obtained", he further speculated that if solutions arose then " ... few additional bioeconomic insights can be anticipated". Cohen (1987) stressed the need for greater realism in the biology and economics considered in optimization studies, but did not appear optimistic about the prospect of further advance in the theory to accommodate such realism. Presented below are results of an attempt to solve the more general non-linear problem; it is believed that in this respect a substantial advance has been made. The results allow recognition of a natural extension to the case of additive Gaussian process and measurement noise. Examples are provided.

The Problem

Assume that the fishery, or fishery complex, can be described by a discrete-time, non-linear, multi-age model, and that fishing occurs with one or several fleets. The underlying population dynamics are assumed to be known, but the recruitments are subject to a natural variability and the population sizes are estimated with some random error. The problem addressed is to find an operable control law that will determine

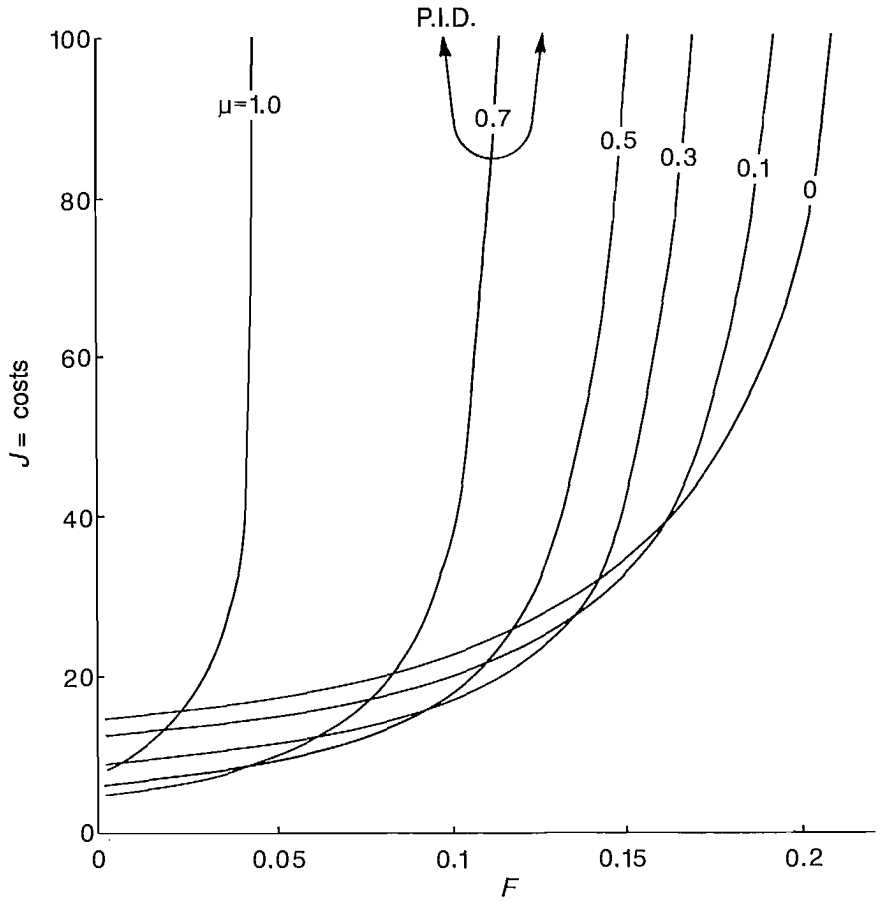


Fig. 1. Average long-term costs (J) from a model of a mackerel stock at different target fishing mortality rates (F) of an optimized one-step, non-linear control law regulating equally deviations in fishing effort and in catch with a tuning parameter μ. The approximate cost of using the well known PID controller is also shown.

the magnitude of annual fishing mortalities so as to maximize an expected discounted, bioeconomic return over an infinite horizon, from a single or multispecies fishery.

Define the following terms:

$\boldsymbol{X}(t)$: the k column vector of population numbers at age at time t, which can be composed of one or several species (this definition is open to extension, see example 2),

$\boldsymbol{X}^*$: the optimal equilibrium of the populations,

$\boldsymbol{U}(t)$: the vector of j control variables, being the set of fishing mortality rates or effort expended by the different fleets $(F_1, F_2, \ldots, F_j)$,

$\boldsymbol{U}^*$: the optimal equilibrium of the controls,

$a(\boldsymbol{X}(t), \boldsymbol{U}(t))$: the vector of the age-specific and density-dependent reproduction and survival functions,

$\boldsymbol{x}(t)$ and $\boldsymbol{u}(t)$: the vectors of deviations from the optima, $\boldsymbol{X}(t) - \boldsymbol{X}^*$ and $\boldsymbol{U}(t) - \boldsymbol{U}^*$,

$x_1(t)$, $u_1(t)$ etc.,: the elements of $\boldsymbol{X}(t)$ and $\boldsymbol{U}(t)$,

$\boldsymbol{e}(t)$: a vector of random additive noise,

β: the discount factor,

$g(\boldsymbol{X}(t), \boldsymbol{U}(t))$: the scalar reward from the fishery at time t,

$\Gamma(\boldsymbol{X})$: is the maximal value of the reward function for a given value $\boldsymbol{X}$ of $\boldsymbol{X}(0)$.

For convenience the following terminology will be used: e.g., $\boldsymbol{g}_{ux}$ is the matrix with j, kth element $\partial^2 g / \partial u_j \partial x_k$ etc. $\boldsymbol{A}^T$ represents the transpose of $\boldsymbol{A}$ and $\boldsymbol{I}$ the unit matrix.

The reward function is taken to have the form,

$$(1) \qquad \sum_t^{\infty} \beta^t \, g\,(\boldsymbol{X}(t), \boldsymbol{U}(t))\,,$$

and the dynamics can be described by the Markovian equation,

$$(2) \qquad \boldsymbol{X}(t+1) = a(\boldsymbol{X}(t), \boldsymbol{U}(t)) + \boldsymbol{e}(t)$$

The problem is to maximize equation 1 subject to the constraint on the dynamics of the system given by equation 2.

The Approach and Solution

The work was originally undertaken by Horwood and Whittle (1986a,b) and proofs of the assertions are given therein. The approach adopted is based upon the dynamic programming equation (e.g., Whittle 1982) to the deterministic problem. Once formulated the dynamic programming equation is expanded, as a perturbation about an assumed optimum equilibrium $(\boldsymbol{X}^*, \boldsymbol{U}^*)$ in $\boldsymbol{x}$ and $\boldsymbol{u}$, to include, crucially, second order terms. The existence of a stable control matrix, defined later, implies the valid existence of a stable optimal equilibrium, which is a positive attractor for the perturbed deterministic or stochastically driven system (Whittle 1982; Horwood and Whittle 1986a). A natural elimination of linear terms leaves a quadratic relationship that can be recognised as the Bellman solution to the optimizing problem of a system with linear dynamics and quadratic costs. The extension to include an additive stochastic element is relatively trivial. Several other studies have considered solutions to approximations of either the initial problem or, and better, the optimal reward function (e.g., Burt and Cummings 1977; Yakowitz and Rutherford 1984; Stengel 1986; Kolberg 1992). However, full advantage has not been taken of the general theory that can be invoked through a translation of the solution to LQG form.

The approach is not the same as taking an initial linear approximation to the dynamics of the system, and in fact the quadratic terms of the dynamics are very important. The problem has not been reduced to the standard LQG (Linear dynamics, Quadratic costs, Gaussian noise) model, only the solution is the same.

By definition, $\Gamma(\boldsymbol{X})$ obeys the dynamic programming equation,

$$\Gamma(\boldsymbol{X}(t)) = \max_{\boldsymbol{U}} [g(\boldsymbol{X}(t), \boldsymbol{U}(t)) + \beta\Gamma(a(\boldsymbol{X}(t), \boldsymbol{U}(t)))] \tag{3}$$

From equations 2 ($\boldsymbol{e} = 0$) and 3 the optimal equilibrium, assuming its existence at this stage, can be found from the simultaneous relationships,

$$\boldsymbol{X}^* = a(\boldsymbol{X}^*, \boldsymbol{U}^*) \text{ and} \tag{4a}$$

$$\boldsymbol{g}_u + \beta \boldsymbol{g}_x (\boldsymbol{I} - \beta \boldsymbol{a}_x) - \boldsymbol{1}_{a_u} = 0 \tag{4b}$$

Multiple solutions may exist, only one of which is the global optimum, but for regular functions, and in all examples studied personally to date, a unique optimum has been found. Equation 3 can be expanded to second order in $\boldsymbol{x}$ and $\boldsymbol{u}$. It can be seen (Horwood and Whittle 1986a) that linear terms in $\boldsymbol{x}$ and $\boldsymbol{u}$ cancel, to leave the Riccati equation,

$$\boldsymbol{P} = \boldsymbol{R} + \beta \boldsymbol{A}^T \boldsymbol{P} \boldsymbol{A} - (\boldsymbol{S}^T + \beta \boldsymbol{A}^T \boldsymbol{P} \boldsymbol{B}) \times (\boldsymbol{Q} + \beta \boldsymbol{B}^T \boldsymbol{P} \boldsymbol{B})^{-1} (\boldsymbol{S} + \beta \boldsymbol{B}^T \boldsymbol{P} \boldsymbol{A}) \tag{5}$$

where,

$$\begin{aligned} \boldsymbol{A} &= \boldsymbol{a}_x, \\ \boldsymbol{B} &= \boldsymbol{a}_u, \\ \boldsymbol{R} &= -\boldsymbol{g}_{xx} - \beta \sum_i \left(\frac{\partial \Gamma}{\partial x_i} \boldsymbol{a}_{i,xx} \right) \\ \boldsymbol{S} &= -\boldsymbol{g}_{ux} - \beta \sum_i \left(\frac{\partial \Gamma}{\partial x_i} \boldsymbol{a}_{i,ux} \right) \\ \boldsymbol{Q} &= -\boldsymbol{g}_{uu} - \beta \sum_i \left(\frac{\partial \Gamma}{\partial x_i} \boldsymbol{a}_{i,uu} \right) \text{ and} \end{aligned}$$

$\boldsymbol{P} = -\Gamma_{xx}$, and where $\Gamma_x = \boldsymbol{g}_x (\boldsymbol{I} - \beta \boldsymbol{a}_x)^{-1}$.

The matrices $\boldsymbol{R}$, $\boldsymbol{S}$ and $\boldsymbol{Q}$ represent the quadratic "costs" of deviating from the optimum. In the standard LQG problem the summed terms of $\boldsymbol{R}$, $\boldsymbol{S}$ and $\boldsymbol{Q}$ are absent, and it is here that the curvatures of the dynamics are seen to be significant.

In the neighbourhood of the optimal equilibrium the optimal control is given by the linear relationship,

$$\boldsymbol{U}(t) = \boldsymbol{U}^* + \boldsymbol{K}(\boldsymbol{X}(t) - \boldsymbol{X}^*), \tag{6}$$

where the vector $\boldsymbol{K}$ is a constant, depending upon the dynamics and economics, given by,

$$\boldsymbol{K} = -(\boldsymbol{Q} + \beta \boldsymbol{B}^T \boldsymbol{P} \boldsymbol{B})^{-1} (\boldsymbol{S} + \beta \boldsymbol{B}^T \boldsymbol{P} \boldsymbol{A}). \tag{7}$$

Equations 6 and 7 represent the locally optimal solution to the posed deterministic problem and to the stochastic problem with $\boldsymbol{e} \neq 0$. The stochastic elements of recruitment and measured stock sizes are replaced by their best estimates. The solution for the matrix $\boldsymbol{P}$, from equation 5, can be found numerically, and substituted into 7. It can be recognised that the optimal control varies linearly as the state variable deviates from its desired location.

Character of the Solution

The approach adopted leads to a familiar intermediate result, that of solving the LQG problem of standard control theory. This system has been much examined and the strong properties of the system are well defined (e.g., Whittle 1982). It is stressed that the solution has not arisen from an initial linearization and that the curvatures of the population dynamics appear in the "cost" function of the LQG problem. The redefinition of the problem allows incorporation of an additive, stochastic noise, for which a certainty equivalence relationship can be proved for the perturbed system.

The main feature of the optimal control is that it (in this case the fishing mortality rate) is linear with respect to departures from the optimum equilibrium of the state variable $(\boldsymbol{X})$, or stock size at age. An attractive feature of such a regulatory mechanism is that it is plausible and comprehensible and hence "sellable" without recourse to the mathematics. Management schemes that have to be adopted by a variety of interested parties, such as technicians, managers, fishermen

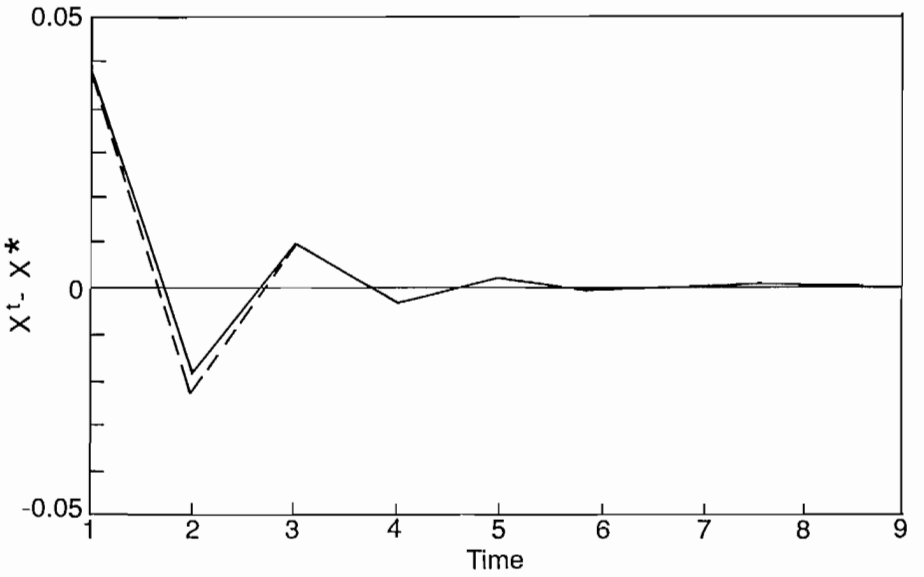


Fig. 2. Optimal trajectories of stock size deviations, from a logistic model, against time, t, calculated numerically (bold line) and from the locally optimal solution (broken line).

etc., should ideally have such a simple final form. The dynamic programming approach gives rise to a control rule of "closed-loop" form. That is, the decisions made are dependent on the current state of the fish stocks and, importantly, are not dependent on previous decisions being optimal.

One should however ask, is the solution valid, is it stable and how good is the approximation? It will be noted that the results are presented conditional upon there being such a thing as a valid optimal equilibrium.

Stability of the controlled system is determined through examination of the roots of the stability matrix $(\boldsymbol{A}+\boldsymbol{BK})$. For stability, the modulus of the roots have to be less than unity. Convergence of the Riccati equation itself implies stability, and if $\boldsymbol{P}$ is obtained from iteration of equation 5 the resulting control should be stable. However, this is not strictly true if $\beta \neq$ unity, and there may be a small region of parameter space that allows convergence but not stability; only stable results are valid. Sufficient conditions for a valid solution are positive definiteness of $\boldsymbol{P}$, $\boldsymbol{Q}$ and $\begin{bmatrix} \boldsymbol{R} & \boldsymbol{S}^T \\ \boldsymbol{S} & \boldsymbol{Q} \end{bmatrix}$, but in examples considered this can be violated and it appears that it is only necessary that $(\boldsymbol{Q}+\beta \boldsymbol{B}^T \boldsymbol{P} \boldsymbol{B})$ be positive definite (see also Whittle 1990). Stability of the system ensures both the existence of an underlying optimal equilibrium and the controllability of the system in relation to variability.

The lack of a solution conveys a real meaning. A solution implies that an optimal equilibrium exists; it is of a stationary form and the deterministic optimum is a constant $(\boldsymbol{X}^*, \boldsymbol{U}^*)$. If no solution is obtained then it can be concluded that the solution is not stationary and it can be periodic or pulsed; examples are described below.

The validity of the approximation can only be investigated through examples. Several examples are provided below which illustrate that important qualitative features of the system are retained and that the approximate and exact solutions are very close in the examples so far studied. Exact solutions were obtained by computationally expensive, finite-time numerical methods.

Examples

Example 1 is from Horwood and Whittle (1986a), example 2 from Horwood and Whittle (1986b), and example 3 from Horwood (1990).

1. Simple Logistic Model

The dynamics and reward function are given by the familiar, scalar form,

$$\begin{aligned} \boldsymbol{x}(t+1) &= \boldsymbol{x}(t)\,(2-\boldsymbol{x}(t))-\boldsymbol{u}(t), \text{ and} \\ g\,(\boldsymbol{x}(t),\boldsymbol{u}(t)) &= \boldsymbol{u}(t)-c\frac{\boldsymbol{u}(t)}{\boldsymbol{x}(t)}. \end{aligned}$$

The cost function assumes that costs are proportional to harvesting effort and that yields generate a unit return per unit harvest. Taking arbitrary values for the cost parameter, c, of 0.1 and for β of 0.9 equations 4a,b give an optimal equilibrium at $(\boldsymbol{x}^*, \boldsymbol{u}^*)$ of (0.505, 0.250). The resulting optimal control from equations 6 and 7 gives $K = 1.401$ and the optimal control law of $\boldsymbol{u}(t) = 0.250 + 1.401(\boldsymbol{x}(t) - 0.505)$. Investigation of the stability matrix shows an oscillatory convergence.

Figure 2 shows the time trajectory given by the optimal control starting with the stock at 0.555 ($x(0) = 0.05$) and no further perturbations. It is compared with the numerical solution obtained from the full model. It is seen that there is a good correspondence between the two results, both qualitatively in the nature of the solution, and quantitatively in that

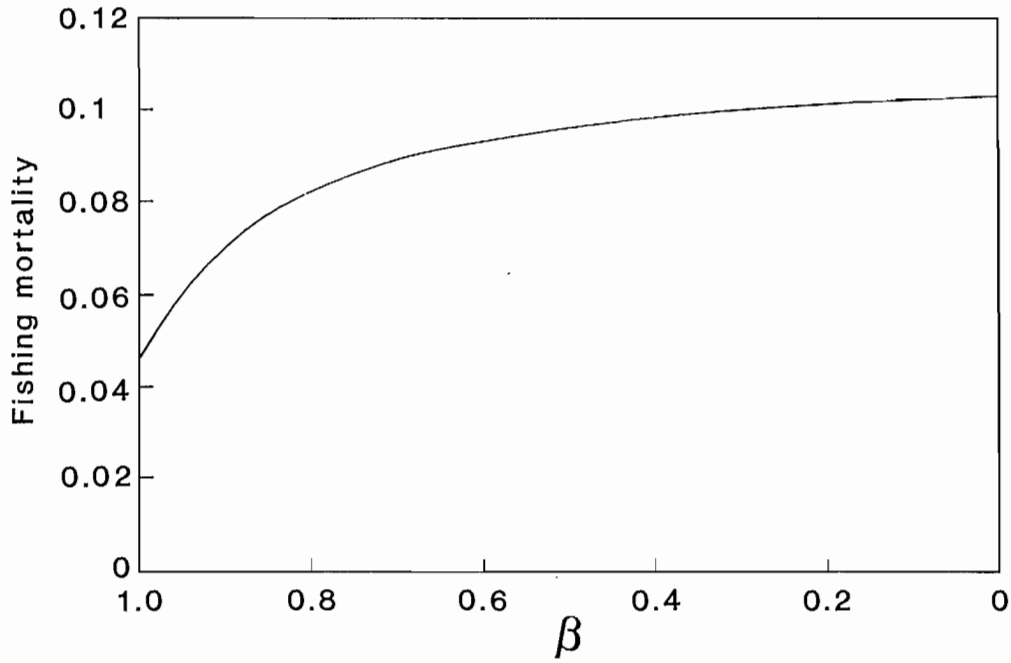


Fig. 3. The locus of optimal equilibria fishing mortality rates (F) against discount factor, β, for a multi-cohort model of a stock of mackerel.

the values are very similar. The richness of the dynamics in these simple, discrete time models cannot be anticipated from the continuous time models. The character of the solution is influenced by the value of the discount factor β. Solutions can be found for $\beta > 0.815$. For values less than that the Riccati equation fails to converge and a solution cannot be found. Numerical examinations show that indeed there is a change in the nature of the solution at this critical value and no optimum equilibrium exists if $\beta < 0.815$. For $c = 0$, then $\boldsymbol{R} = \boldsymbol{S} = \boldsymbol{0}$ and no optimal equilibrium exists; Clark (1971) showed that for this model a bang-bang control was optimal.

2. Harvesting a Multi-Cohort Stock

The following example is based on a caricatured study of the western stock of mackerel (Horwood and Whittle, 1986b). The dynamics are assumed to follow a typical fisheries model that incorporates a Beverton-Holt recruitment relationship (f(SSB)) for the number of age 1 (x_1) mackerel against the biomass of the spawning stock (SSB). The model is,

$$\begin{aligned} x_1(t+1) &= f[\mathrm{SSB}(X(t))] \\ x_{i+1}(t+1) &= x_i(t)\exp(-M_i - q_i F(t)) \\ x_{10}(t+1) &= x_9(t)\exp(-M_9 - q_9 F(t)) \\ &\quad + x_{10}(t)\exp(-M_{10} - q_{10}F(t)), \end{aligned}$$

where $i = 1, \ldots, 8$ and a plus group is assumed for age 10 and over. The age specific natural mortality rate (M) and selectivity rate (q) are constant over time, and the control is the annual fishing mortality rate, $F(t)$. The reward function was assumed to be,

$$g(t) = \text{yield}(t) - 2.5F(t),$$

that is, the revenue is proportional to yield in weight and the costs are linear in F.

The parameters of the model gave MSY at $F = 0.086\text{y}^{-1}$ and MEY at $F = 0.046\text{y}^{-1}$. Figure 3 gives the locus of the optimal equilibrium as β is varied. For $\beta = 1.0$, F^* is at 0.046y^{-1}, as anticipated from the simpler models, and for $\beta = 0$, $F^* = 0.103\text{y}^{-1}$, the position of zero marginal return (not necessarily zero profit). The control, F, is scalar, and the optimal control is given by,

$$F(t) = F^* + \boldsymbol{K}\left(\boldsymbol{X}(t) - \boldsymbol{X}^*\right), \tag{8a}$$

where, for the example of $\beta = 0.86$, the control matrix

$$\boldsymbol{K} = [-1.69, 0.91, 0.44, 1.44, 0.76, 1.57, 2.24, 2.11, 2.08, 2.72], \tag{8b}$$

values have been multiplied by 10^4.

If all age groups of 2 and older were at their equilibrium, then it can be seen that a relatively high year-class, entering as 1 year-olds, would cause the fishing mortality to be reduced (because of the -1.69×10^{-4}); i.e., $F(t) = F^* - 1.69 \times 10^{-4}(x_1(t) - x_1^*)$. This implies that there must be too small a mesh size and the optimal control is allowing the good year-class to be conserved until later.

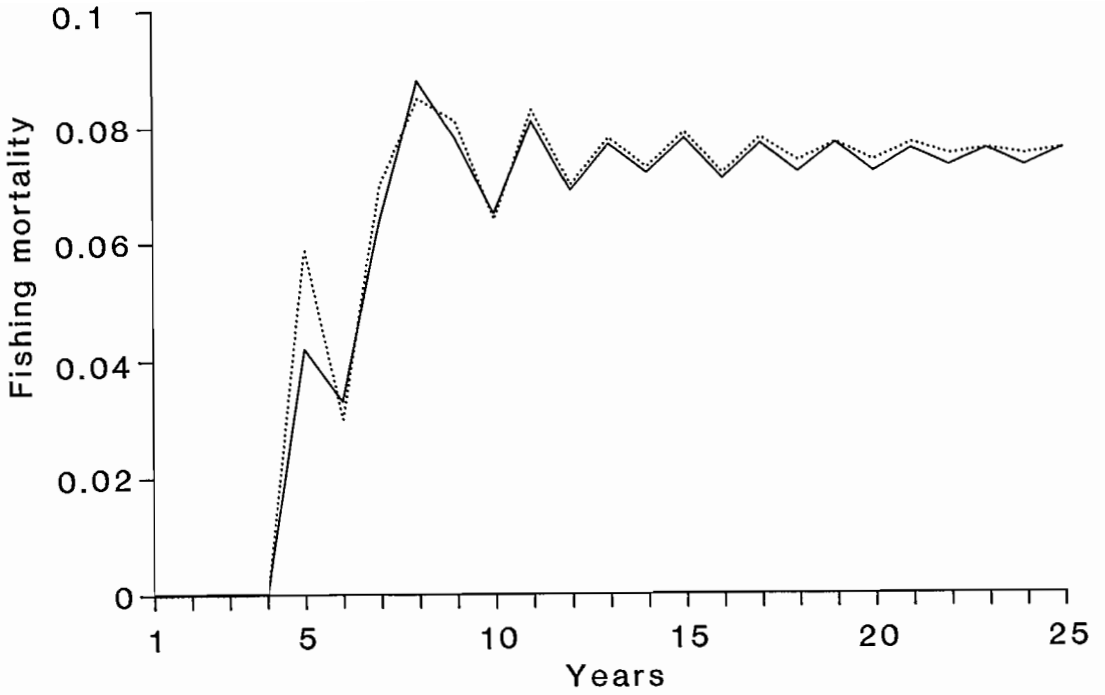


Fig. 4. Comparison of numerically calculated, truly optimal, fishing mortality rates (bold line) and those derived from the locally optimal control (broken line) against time for the modelled mackerel stock started at a depleted level and with $\beta = 0.86$.

Table 1. Control matrix $\boldsymbol{K}$ for a model of the North Sea haddock incorporating a stock and recruitment function and quadratic costs of change in fishing effort. Numbers in parentheses give the percentage increase in $F(t)$ due to a 10% increase in the state variables $(\boldsymbol{X}(t), F(t-1))$. Elements of $\boldsymbol{K}$ have been multiplied by 1,000 (Horwood, 1991).

$\boldsymbol{K}$	Increase in $F(t)$
−0.01	(−1.7%)
−0.24	(−7.6%)
−0.07	(−1.2%)
0.27	(2.6%)
0.66	(3.3%)
0.93	(2.7%)
0.76	(1.4%)
1.40	(1.6%)
2.10	(1.6%)
2.30	(2.7%)
460.00	(4.6%)

The converse follows, if there is a poor year class then the effort is increased to minimize the loss of older fish to natural mortality.

This important feature, which is applicable in multispecies fisheries, allows a more efficient use of variable year classes than current management methods which are based upon more static considerations.

A similar, but stronger, feature is seen in control laws proposed for the North Sea haddock (Horwood 1991). Table 1 shows the control matrix $\boldsymbol{K}$ for a particular realization of the North Sea haddock models. The last term is associated with a year to year change in F, such that,

$$F(t) = F^* + k_1\left(x_1(t) - x_1^*\right) + k_{10}\left(x_{10}(t) - x_{10}^*\right) + \cdots + k_{11}\left(F(t-1) - F^*\right),$$

and it follows from ascribing a cost to annual changes in effort. Table 1 shows that, in this example, a 10% deviation in any of the state variables (stock size and previous fishing effort) causes about a 2–8% change in $F(t)$.

The non-linear mackerel model is mathematically complicated and one should inquire as to how good are the locally optimal results. Figure 4 illustrates the pattern of optimal $F(t)$ values, calculated with the above optimal control rule (equations 8a,b), needed to bring a much depleted mackerel stock to its optimum level. The results are compared with those from a numerical maximization and it can be seen that the agreement is very good.

As demonstrated by the previous example not all regions of parameter space allowed a solution. For $\beta > 0.9$ the Riccati equation did not converge and the optimal equilibria are then not valid. The solution in these cases take the form of a pulsed fishing, as shown in Figure 5 (see also Horwood 1987, 1991). The locally optimal approach does not help in these cases, but nevertheless it does show that the optimal solution is not of a stationary form. In practice, small costs associated with changes in the rate of fishing mortality will tend to give stationary solutions.

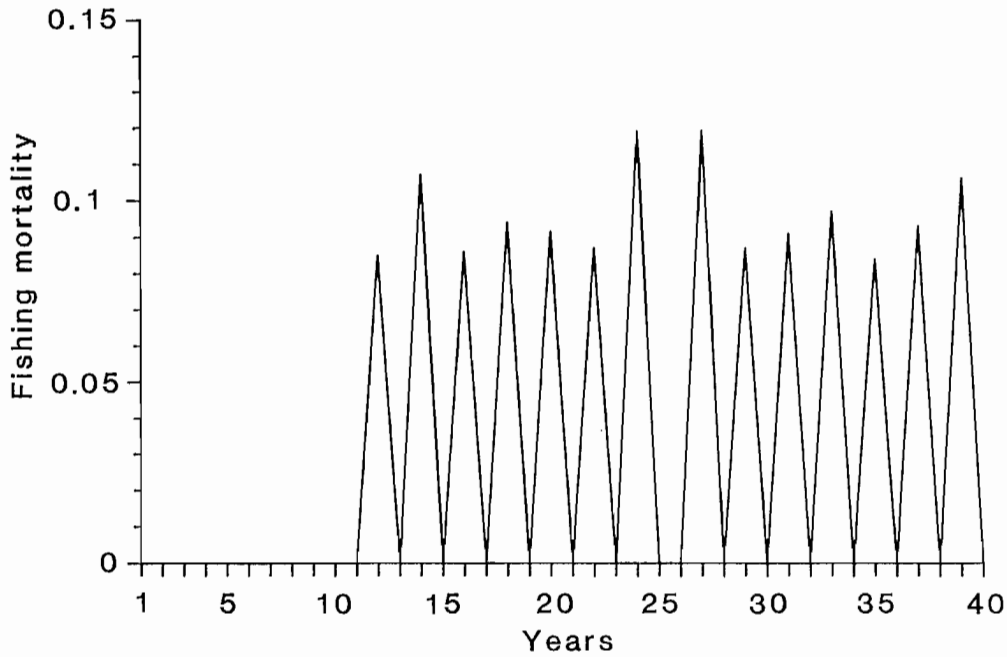


Fig. 5. The pulsed, optimal fishing pattern for the mackerel model with $\beta = 1.0$, no optimal equilibrium exists and no local control law can be constructed.

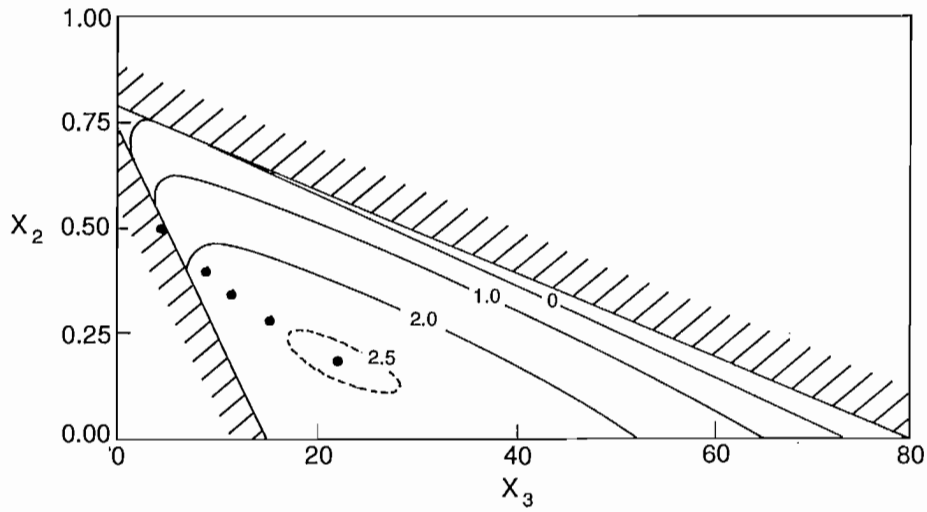


Fig. 6. Contours of equilibria values of the reward function from the "pest" model described in the text, for numbers of species x_1 and x_3. Hatched zones show invalid parameter spaces. The dots indicate the locus of the optimal equilibria for values of $\beta = 1.0$ (at MEY) , 0.9, 0.8, 0.7 and 0.5.

3. A Predator-Prey-Multifleet Model

This example is included to demonstrate the multispecies, multi-fleet character of the problem posed and of the solutions, since descriptions of the models in matrix notation can hide their full scope. The example is of two species harvested by two methods. The first species is of one age-class, with the dynamics described by the logistic model as above. The second species is of two age-classes, involving a stock-recruitment relationship and a plus-group. The first species is a predator on the juvenile age-class of the second species having an effect m per encounter. The two species are subject to independent harvesting or control regimes. The scalar dynamics are given by (values of x and u on the **rhs** are evaluated at time t),

$$\begin{aligned} x_1(t+1) &= x_1 + x_1(1 - x_1) - u_1 x_1 \\ x_2(t+1) &= 20x_3/(1 + x_3) \\ x_3(t+1) &= 0.8x_3 + 0.8x_2 - mx_1x_2 - u_2x_3 \end{aligned}$$

The reward function is assumed to take the following form,

$$g(t) = u_2x_3 \exp(-0.1u_2x_3) - u_2 - (u_1)^2 .$$

No reward accrues from harvesting species one, only costs, and consequently we can recognise species one as a "pest". It may be seals eating fish, oystercatchers eating cockles or insects attacking crops. A quadratic cost of harvesting the pest is assumed, which may well be sensible if social costs are considered. The adults of species two are harvested with a reward but its value is subject to a decreasing *per capita* return and costs are linear in u_2.

Figure 6 shows the equilibria values of g of the above system for the predation rate $m = 1.0$. Hatched areas show invalid regions of parameter space ($u_2 < 0.0$, $u_2 > 0.8$). The locus of the optimum equilibria are shown as β varies, at $\beta = 1.0$ the point lies at the joint MEY. For $m = 1.0$ and $\beta = 0.9$ the optimum equilibrium is found at $\boldsymbol{X}^* = (0.28, 18.8, 15.1)$ and $\boldsymbol{U}^* = (0.72, 0.44)$ and, after solving for $\boldsymbol{K}$, the optimal control law is given by,

$$\begin{aligned} u_1(t) &= 0.715 + 3.212\,(x_1(t) - 0.28) \\ &- 0.019\,(x_2(t) - 18.8) - 0.025\,(x_3(t) - 15.1)\,, \\ u_2(t) &= 0.441 - 0.095\,(x_1(t) - 0.28) \\ &+ 0.003\,(x_2(t) - 18.8) - 0.021\,(x_3(t) - 15.1)\,. \end{aligned}$$

Figure 7 shows a comparison of numerically derived optimal trajectories and those given by the locally optimal control law. Again good agreements are achieved.

Conclusion

The above reviews the development and applicability of a stochastic, locally optimal control of a general, non-linear harvested complex. The approach offers a significant increase in the range of models that can be solved, previous solutions have been obtained for either very simple forms or else special cases. The inclusion of stochastic effects is of particular use in fisheries, because of the large natural variations in recruitment and the probably large errors in measurement of stock size. The solutions have been presented for the case of white (uncorrelated) noise, the solutions need amendment to incorporate any serial correlation in recruitments. The examples show that, when a solution is obtained, the approximate solution is very near to that given by an exact numerical solution. (Numerical solutions cannot be used more generally. For deterministic problems they are costly in time, but more importantly do not allow the construction of a general management control. The stochastic laws cannot be derived). Even when a solution is not forthcoming, that itself is informative even though the precise control is unknown.

The control law is linear, the magnitude of the parameters are determined from the dynamics of the system and the costs. Given this, one might question why *ad hoc* controls are proposed; at least such controls should be tested against the locally optimal control law. Figure 1 shows that poor controls can easily be constructed. Of course one answer is that the reward function is unspecified for most fisheries and so investigators do not know what to maximize, consequently they

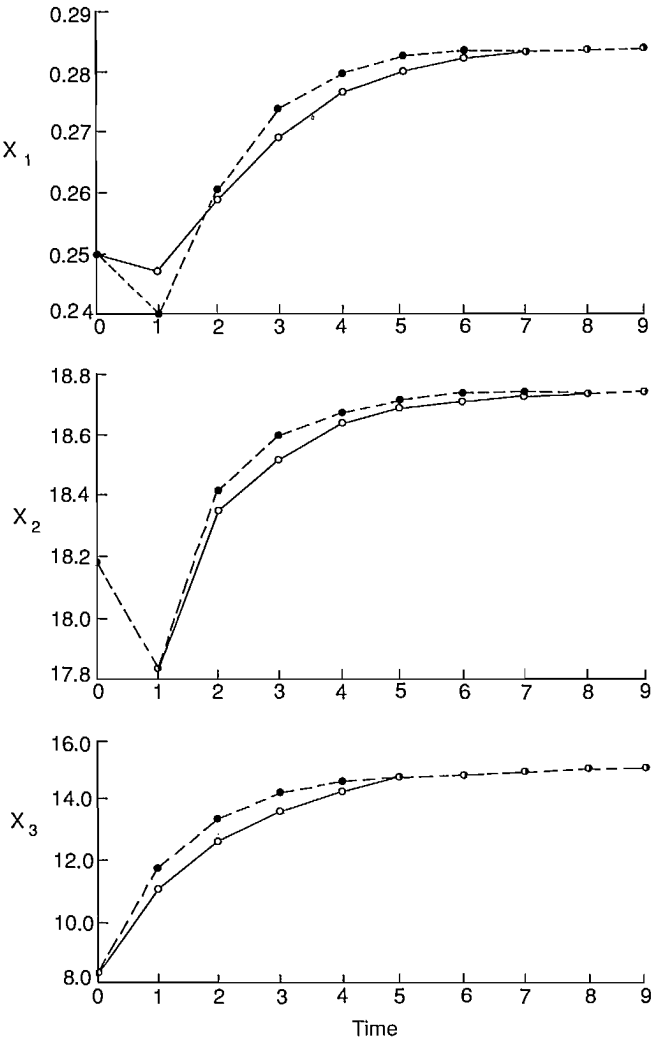


Fig. 7. The numerically calculated (bold line) and locally optimally derived trajectories (broken line) for the "pest" model, with $\beta = 0.9$ and $m = 1.0$, and started at substantial deviations from the optimum equilibrium.

attempt to develop controls with generally sensible properties. A conclusion would seem to be that much more attention should be paid to defining *operable objectives* of fisheries management by the *managers* — rather than by the scientists. In the case of the European Community objectives are specified in a "motherhood" form — high yield, high value, high profit, high employment, high stocks etc., the real decisions of management appear to occur in a capricious manner, frequently after a crisis.

The approach outlined above has further extension. More general non-approximate solutions can be found if we assume that rewards can be given by an expectation (E) of the exponential form of the more usually recognised reward function g,

$$\gamma(\theta) = \log\left[E\,(\exp\,(\theta g))\right]/\,\theta \qquad (9)$$

This may seem to be a mathematical trick to obtain some solution to a problem that no one wanted solved; this is not unknown! However, this is very far from the case. Equation 9 can be identified as a risk sensitive criterion, the risk sensitive parameter being θ, and as $\theta \to 0$ the model reduces to the usual risk neutral case. If $\theta < 0$ the objective is risk averse (since we are seeking to maximize 9). The parameter can be more easily understood when it is recognised that this approach requires extremization of the function $(g + D/\theta)$, where D is related to the variance of $g(t)$ and D describes the improbability of any particular trajectory over time. A negative value of θ will result in a control that makes D (which is necessarily positive) small. However, a more specific relationship between values for θ and population criteria is still to be established. If the criterion is accepted then strong consequences follow. In the, perhaps unlikely, situation when we have a LQG problem then a stochastic maximum principle (SMP) is available and a certainty equivalence principle (CEP) is established. Of greater importance is that, under conditions appropriate to large deviation theory, (when the model is close to determinism in that the increment of the variable in a small time interval can be regarded as the average of a large number of statistically independent increments), a SMP and CEP are also proved; details are found in Whittle (1982, 1990).

Returning to more general issues, two aspects appear important. First, it is assumed that management strategies can be developed — this implies a time scale of greater than one or two years, and at least five, it being the typical generation time of most exploited stocks. We are then forced to model the population dynamics. In doing this we cannot neglect the problem that the underlying functional forms are poorly known and that any management scheme must be robust to that lack of knowledge. Second, is that we need to know better the variability in assessments. Much attention has been devoted in recent years to deriving very much improved procedures for estimating stock sizes. Much less attention has been devoted to estimates of variance of these estimates. Any competent management scheme must use fully such information.

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ITQ Based Fisheries Management

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Arnason, R. 1993. ITQ based fisheries management. p. 345–356. *In* S. J. Smith, J. J. Hunt and D. Rivard [ed.] Risk evaluation and biological reference points for fisheries management. Can. Spec. Publ. Fish. Aquat. Sci. 120.

Traditional methods of fisheries management including licences, effort restrictions, catch quotas etc., make enormous demands on a central fisheries manager. The fisheries manager is relied on to collect the necessary biological and economic data for determining the optimal levels of the relevant control variables, perform the calculations and enforce the results. In most cases these demands greatly exceed the capabilities of any realistic fisheries manager. This paper shows that a particular institutional framework for fisheries, namely the Individual Transferable Share Quota (ITSQ) system, makes it possible to utilize market forces to generate maximum economic rents from a fishery with minimal centralized intervention and data collection. The paper goes on to show that this result also applies to multispecies fisheries or, more generally, to the ecological context. Finally, the paper examines to what extent these results may also apply in fisheries subject to uncertainty.

Les méthodes traditionnelles de gestion des pêches, qui font appel à des permis, des restrictions sur l'effort, des contingents de prises, etc., imposent des demandes énormes à un gestionnaire des pêches investi d'un rôle central. Le gestionnaire des pêches a pour tâche de recueillir les données biologiques et économiques nécessaires pour fixer les niveaux optimaux des variables pertinentes pour la régulation, d'effectuer les calculs et de faire appliquer les résultats. Dans la plupart des cas, ces demandes dépassent nettement les capacités d'un gestionnaire réaliste.

La présente étude montre qu'un cadre institutionnel particulier, le système des contingents individuels transférables (CIT), permet d'utiliser les forces du marché pour tirer une rente économique maximale de la pêche avec un minimum de centralisation dans l'intervention et la collecte des données. Nous montrons ensuite que ce résultat s'applique aussi aux pêches plurispécifiques ou, de façon plus générale, au contexte écologique. Enfin, nous examinons dans quelle mesure ces résultats peuvent aussi s'appliquer dans des pêches sujettes à l'incertitude.

It is now firmly established that competitive utilization of ocean fisheries is generally economically wasteful[1]. The fundamental reason is that fishing firms, by their harvesting activity, impose production externalities on each other. By removing fish from limited stocks, each fishing firm reduces the harvesting possibilities of other firms. In competitive ocean fisheries these externalities are so pervasive that the equilibrium outcome is one of a near or complete dissipation of economic rents irrespective of the underlying bioeconomic productivity of the fisheries.

This observation has prompted fisheries authorities worldwide to search for management measures to remedy the situation. Many direct management methods have been tried. Some are biologically oriented designed to preserve the biological resource. Others are economically oriented designed to restore economic rents in the fisheries. Among the more recently suggested (and tried) economic management methods are entry limitations, effort restrictions and taxes on fishing activity. The aim of these measures is to interfere in the decision making process of fishing firms so as to elicit socially optimal behaviour. In fact, given the appropriate conditions, each of these management measures can be shown to be theoretically capable of restoring economic efficiency in common property fisheries.

The problem is that these direct management methods, although efficient on paper, are not very helpful in practice. One fundamental reason has to do with their informational requirements. In most ocean fisheries immense amounts of data are needed for the calculation of optimal entry licences, effort restrictions, taxes, etc. In order to do these calculations correctly, the fisheries manager must know virtually every detail about the fishery at each point of time. Clearly, this requirement exceeds of the capacity of any real world fisheries manager. It follows that optimal management of fisheries based on these direct methods is not feasible and the methods are therefore of limited practical use.

The problem of ocean fisheries may be viewed from a different angle, however. More precisely, it may be regarded as one of inappropriate institutional organization. On land, competition between economic agents within the framework of the market system normally yields great economic benefits. Due to the pervasive production externalities in fisheries, this institutional arrangement is not suitable for guiding private enterprise toward the optimal utilization of ocean resources. In fact, as already mentioned, it usually generates the opposite result. This failure, however, does not preclude the existence of a modified institutional framework that allows the fisheries manager to exploit the basic advantages of the market system in order to solve the fisheries problem.

[1]The first known derivation of this result is was by Warming (1911). In the English speaking world the result appears to have been first demonstrated by Gordon (1954).

This paper describes a particular institutional arrangement of this nature. The crucial element of this arrangement is the establishment of permanent property rights in the fishery via individual transferable share quotas (ITSQ). Given this institutional framework and the usual assumptions regarding the rationality of economic agents, the requirements for centralized fisheries management are minimal. In particular, the fisheries authorities do not have to engage in extensive measurements of the relevant biological and economic relationships. All they need to do is to observe and respond in a relatively simple way to private decisions as they appear in the market for share quotas.

The Basic Fisheries Model

The analysis of this paper is based on a multi-species, multi-firm fisheries model. To facilitate the presentation a highly aggregative representation of key variables is employed. Thus, fisheries inputs are represented by a single variable called fishing effort. More restrictively perhaps, the biological characteristics of each species of fish are also represented by a single variable, namely its biomass. Although standard in analytical fisheries economics (see e.g., Clark (1976), Conrad and Clark (1987) and Neher (1990), this particular simplification has the disadvantage of preventing the study of the rather important issues of cohort fisheries management (see e.g., Helgason (1989)).

Consider a fishery in which a number of fishing firms exploit several stocks of fish belonging to a common ecology.

Let there be M species in the ecology, where $M > 0$. Also, let the variable $x(j, t)$ represent the aggregate biomass of species j at time t. The state of the complete ecology at time t is described by the vector[2]

$$\boldsymbol{x}(t) = [x(1,t), x(2,t), \ldots, x(M,t)].$$

Let the fishing industry consist of N fishing firms where $N > 0$. All these firms are assumed to be profit maximizers. At a given point of time some of these N firms may choose not to be operating in the industry. Thus N refers to potentially active fishing firms.

Each fishing firm is assumed to be able to exert fishing effort toward one or more of the species in the ecology. Refer to the fishing effort by firm i toward species j at time t by $e(i, j, t)$. The corresponding vector of fishing efforts is:

$$\boldsymbol{e}(i,t) = [e(i,1,t), e(i,2,t), \ldots, e(i,M,t)].$$

Normally, one expects that only one of the elements of the vector $\boldsymbol{e}(i, t)$ is nonzero at a given point of time.

Suppressing the time indices, t, the firms' instantaneous harvesting functions for each of the species is given by:

$$(1) \qquad y(i,j) = Y(\boldsymbol{e}(i), x(j); i, j), \text{ all } i \text{ and } j.$$

[2]Vectors will be indicated by boldface in this paper

The functions $Y(.,.;.)$ are assumed to be increasing in both arguments. For mathematical convenience they are also assumed to be concave and to exhibit sufficiently high degree of smoothness. More precisely, they are taken to be twice continuously differentiable. Moreover,

$$Y(\boldsymbol{0},.;.) = Y(.,0;.) = Y_{e(i)}(.,0;.) = Y_x(\boldsymbol{0},.;.) = 0,$$

where the subscripts indicate derivatives, i.e.,

$$Y_{e(i)} \equiv \frac{\partial Y(t)}{\partial e(i)}.$$

It should be noted that the above specification of the harvesting functions allows for bycatch. The vector of catches by firm i is denoted by:

$$\boldsymbol{y}(i) = [y(i,1), y(i,2), \ldots, y(i,M)].$$

Due to bycatch several elements of the vector $\boldsymbol{y}(i)$ may be positive at the same time.

The harvesting cost functions which are also assumed to vary over firms and species are given by:

$$(2) \qquad c(i,j) = C(e(i,j); i, j), \text{ all } i \text{ and } j,$$

where the functions $C(.;.)$ are assumed to be twice continuously differentiable, increasing and convex in all $e(i, j)$.

While inoperative fishing firms do not incur costs, fishing firms currently operating in the industry are assumed to experience costs even if they do not exert any fishing effort. Thus, $C(0; i, j) \geq 0$ with the strict inequality applying to operative firms. This specification has the effect of keeping the number of operative fishing firms finite.

Biological growth is defined by the differential equations

$$(3) \qquad \boldsymbol{x}' \equiv \frac{\partial \boldsymbol{x}(t)}{\partial t} = \boldsymbol{G}(\boldsymbol{x}) - \sum_i \boldsymbol{y}(i).$$

The $(1 \times M)$ vector $\boldsymbol{x}'$ represents net instantaneous biomass growth for each of the M species. $G(x) = [G(\boldsymbol{x}, 1), G(\boldsymbol{x}, 2), \ldots, \boldsymbol{G}(\boldsymbol{x}, M)]$ is the vector of natural growth functions one for each species and $\sum_i \boldsymbol{y}(i)$ is the corresponding vector of biomass extraction by harvesting. Note that each natural growth function depends on all biomass levels in the ecology. The Jacobian matrix of $\boldsymbol{G}(\boldsymbol{x})$, i.e., its matrix of first derivatives, $[\partial G(h)/\partial x(k)]$, is often referred to as the community matrix.

The natural growth functions, $G(\boldsymbol{x}, j)$'s, are assumed to be twice continuously differentiable and strictly concave in their own biomass levels, $x(j)$. They are moreover assumed to have a maximum at a strictly positive level of their own biomass irrespective of the biomass levels of the other species. More formally:

$$\partial G(\boldsymbol{x}, j)/\partial x(j) = 0, \text{ for some } x(j) > 0 \text{ and all } j.$$

Furthermore, it is assumed that there exists a strictly positive vector of biomass levels at which growth is positive for all the species simultaneously, i.e.,

$$\boldsymbol{G}(\boldsymbol{x}) > \boldsymbol{0}, \text{ for some } \boldsymbol{x} > \boldsymbol{0}.$$

Finally, let the vector $\boldsymbol{p}$ and the scalar r refer to the market prices of catch and the discount rate respectively. It is assumed that the elements of $\boldsymbol{p}$ are finite, $\boldsymbol{p}, r > 0$ and remain constant over time. In what follows it will be taken for granted that these prices coincide with social shadow prices.

Given these specifications, the instantaneous profit function for a representative fishing firm i may be written as:

$$(4) \quad \pi(\boldsymbol{e}(i), \boldsymbol{x}, \boldsymbol{p}; i) = \sum_j p(j)y(i, j) - C\left(e(i, j); i\right),$$

where $\pi(.,.,.;.)$ is twice continuously differentiable and jointly concave in $\boldsymbol{e}(i)$ and $\boldsymbol{x}$.

The present value of firm's i future profits from the fishery is defined by:

$$(5) \quad \mathrm{PV}\left(\{\boldsymbol{e}(i)\}, \{\boldsymbol{x}\}, \boldsymbol{p}, r; i\right) = \int_0^\infty \pi\left(\boldsymbol{e}(i), \boldsymbol{x}, \boldsymbol{p}; i\right) \times \exp\left(-rt\right) dt,$$

where the braces, $\{\cdot\}$, indicate that the time path of the respective variable is involved.

Provided that the economic prices involved in the profit functions, (4) and (5), accurately reflect the respective social values, these functions may be taken as measures of social benefits. Notice, however, that they do not necessarily measure resource rents since some of the profits may be intramarginal ones (for a discussion of resource rents see Copes (1972)).

The efficiency properties of the single species version of this kind of fisheries model have been extensively investigated in recent years and are now well established. The fundamental result is that competitive utilization will be economically inefficient. In fact, given a sufficient number of fishing firms, competition for catch will normally all but eliminate economic rents obtainable from the fishery[3].

The Individual Transferable Share Quota System (ITSQ)

Let us now consider a fishery regulated by means of individual catch quotas. Many variants of this system are conceivable. Here we restrict our attention to the following basic system:

1. The individual catch quotas are shares in the total allowable rate of catch (TAC). These quotas are referred to as share quotas.
2. The share quotas stipulate the firm's permitted rate of catch.
3. The share quotas are permanent in the sense that they allow the holder the stated share in the TAC's in perpetuity.
4. The share quotas are transferable and perfectly divisible.
5. A central authority, which may be referred to as the fisheries manager, issues the initial share quotas and subsequently decides on the TAC for each species in the ecology at each point of time.

We refer to this system as the individual transferable share quota system or, in short, ITSQ.

The significance of a ITSQ system compared with the individual transferable quantity quota system (ITQ) is primarily in terms of the effect of variations in TAC on the economics of the firms. Under ITSQ system, changes in TAC are automatically reflected in uncompensated quota increases or decreases for individual firms. Under the ITQ system, on the other hand, the fisheries manager can only implement TAC adjustments by trading in the quota market. Hence, in the ITQ system, individual firms are always compensated for variations in TAC's in both directions. This suggests that under the ITQ system quota holders have a somewhat reduced interest in the fate of the fish stocks compared to the ITSQ system. This clearly has important implications for fisheries management (for a further discussion of this see Arnason (1990)).

The ITSQ system described above requires that all quotas be met exactly. This important point warrants an explanation. If a particular fishery, j, say, is profitable, profit maximizing firms will obviously not choose to hold unused share quotas. In the ecological context, however, optimal fisheries management will typically require the reduction of the stock of some species that are themselves not valuable but compete with or prey on valuable species. Such fisheries will not be privately profitable and the respective quota holders would like to avoid filling their quotas. For this reason the requirement that all quotas be exactly met is needed[4].

The ITSQ system described above is continuous in the sense that the quotas stipulate instantaneous rates of catch. Discrete quota systems, on the other hand, constrain the volume of catch over a period of finite length. The significance of this should not be missed. The crucial point is that fisheries management on the basis of a discrete quota system is, in general, not capable of generating full efficiency. The reason is that if the quota period is finite, different exploitation paths will satisfy the quota constraint. Generally, not all of these paths can be optimal.

[3]Recent references for single species fisheries (easily extended to the multi-species case) are provided by Clark and Munro (1982) and Arnason (1990).

[4]Actually, the requirement could be relaxed to state that the share quotas stipulate the upper limit on the permitted rate of catch for profitable fisheries and a lower limit for unprofitable fisheries.

Continuous quota systems in the above sense are usually not possible in practice. It follows that actual quota management systems cannot be fully efficient. This result, of course, applies not only to catch quotas but management by quantity restrictions in general. Note, however, as the quota periods are reduced the discrete quota system converges to the continuous one[5].

Within the institutional framework of the ITSQ system, individual quota holdings at time t are given by:

$$q(i, j, t) = \alpha(i, j, t)Q(j, t), \text{ all } i, j \text{ and } t, \tag{6}$$

where $1 \geq \alpha(i, j, t) \geq 0$, $q(i, j, t)$ stands for the volume of quota and $\alpha(i, j, t)$ the quota share in species j held by firm i at time t. $Q(j, t)$ represents the total quota or TAC for species j at time t.

According to the ITSQ system, the share quotas are transferable and perfectly divisible. The share quotas thus constitute a homogeneous tradeable commodity. Consequently we can safely assume that a market for share quotas will emerge. To simplify the discussion we, moreover, assume that this market continuously equilibrates supply and demand for share quotas and thus generates a continuous path of temporary equilibrium share quota prices over time. Let the equilibrium quota price for species j at time t be denoted by $s(j, t)$.

Since, under the ITSQ system, quota trades are possible, the instantaneous profit function of a representative firm i, defined in equation (4) above, must now be modified accordingly:

$$\begin{aligned} \pi(\boldsymbol{e}(i), z(i), \boldsymbol{x}, \boldsymbol{p}, \boldsymbol{s}; i) &= \sum_j p(j)y(i, j) - C(e(i, j); i) \\ &\quad - s(j)z(i, j), \end{aligned} \tag{7}$$

where, as before, $p(j)$ represents the catch price for species j, $y(i, j)$ the catch and $e(i, j)$ the fishing effort for species j by firm i. $\boldsymbol{x}$ represents the vector of species biomasses, $s(j)$ the share quota price for species j, $z(i, j)$ firm i's trades in share quotas for species j and $z(i)$ the vector of all such trades.

Now, according to the ITSQ system:

$$Y(\boldsymbol{e}(i), x(j)) = q(i, j), \text{ for all } i, j, \tag{8}$$

where, it will be recalled, $\boldsymbol{e}(i)$ represents firm i's vector of fishing effort for all the species.

Therefore, given the properties of the harvesting functions[6], the vector of fishing effort must satisfy:

$$\boldsymbol{e}(i) = \boldsymbol{E}(\boldsymbol{q}(i), \boldsymbol{x}), \text{ for all } i, \tag{9}$$

where $\boldsymbol{q}(i)$ is firm i's vector of share quotas of all the species.

Given (8) and (9), the instantaneous profit function for firm i can be written in a more convenient form as:

$$\begin{aligned} \pi(\boldsymbol{q}, \boldsymbol{x}, \boldsymbol{p}, \boldsymbol{s}; i) &= \sum_j p(j)q(i, j) - C(E(\boldsymbol{q}(i), \boldsymbol{x}); i) \\ &\quad - \sum_j s(j)z(i, j), \end{aligned} \tag{10}$$

where

$$\begin{aligned} C(E(\boldsymbol{q}(i), \boldsymbol{x}), i) &= \sum_j C(E(\boldsymbol{q}(i), \boldsymbol{x}; j); i). \\ \boldsymbol{q}(i) &= [q(i, 1), q(i, 2), \ldots, q(i, M)], \\ q(i, j) &= \alpha(i, j)Q(j). \end{aligned}$$

Now, under the ITSQ system the fisheries management problem is to select total quotas and allocate share quotas to individual firms so as to maximize economic benefits from the fishery. More precisely:

$$\begin{aligned} \underset{\text{all } \{\alpha(i, j)\}, \{Q\}}{\text{Max}} J &= \sum_i \int_0^\infty \sum_j (p(j)\alpha(i, j)\boldsymbol{Q} \\ &\quad - C(E(\boldsymbol{q}(i), \boldsymbol{x}); i)) \exp(-rt)\, dt, \end{aligned} \tag{I}$$

Subject to

$$\begin{aligned} &\text{(a)} \quad \boldsymbol{x}' = G(\boldsymbol{x}) - \boldsymbol{Q}, \\ &\text{(b)} \quad \sum_i \alpha(i, j) = 1, \\ &\text{(c)} \quad 1 \geq \alpha(i, j) \geq 0, \text{ all } i, \end{aligned}$$

To solve this problem requires full instantaneous knowledge of the biological growth functions as well as all individual harvesting and cost functions in every fishery. Clearly these requirements are beyond the capabilities of any real fisheries manager.

The problem of individual fishing firms under the ITSQ system is much simpler than the general problem. Most importantly, individual fishing decisions cannot influence the path of the ecology as this is completely determined by the natural growth functions and the TAC's, i.e., the vector $\boldsymbol{Q}$, determined by the fisheries manager. Consequently, each firm's decision problem is limited to selecting its share quotas on the basis of its own economic conditions and market prices. More formally, the problem of the representative fishing firm, i, is:

$$\begin{aligned} &\underset{\text{all } \{z(i)\}}{\text{Max}} J = \\ &\int_0^\infty \sum_j \left(p(j)\alpha(i, j)Q(j) - C(E(\boldsymbol{q}(i), \boldsymbol{x}); i) \right. \\ &\left. \quad - \sum_j s(j)z(i, j) \right) \exp(-rt)\, dt, \end{aligned} \tag{II}$$

[5] In the limit, as the quota periods become infinitely short, the discrete quota system becomes identical to the continuous one.

[6] That satisfy the conditions of the implicit function theorem. See e.g., Takayama (1974, p. 406).

Subject to

$$\text{(a)} \quad \alpha'(i, j) = \partial\alpha(i, j)/\partial t = z(i, j),$$
$$\text{(b)} \quad 1 \geq \alpha(i, j) \geq 0.$$

It should be noted that the control variables, $z(i, j)$, appear linearly in this problem. Consequently, the optimal control will be of a bang-bang character. Since each $z(i, j)$ is unbounded, this means that the actual share quotas of the firms will be instantaneously adjusted to desired levels. Thus, quota holdings will at all times be at their optimal long run level given the variables that are exogenous to the firms, i.e., $\boldsymbol{Q}$, $\boldsymbol{x}$, $\boldsymbol{s}$, $\boldsymbol{p}$ and r. When these variables change, however, quota holdings must be adjusted.

Now, individual fishing firms, of course, hold the most complete knowledge of their own economic conditions. Moreover, it is in their own interest to keep track of all other data, including market prices, relevant to their profits. Hence they can be relied on to choose their share quotas, $\alpha(i, j)$ in the quota market, so as to maximize instantaneous profits. To see this, consider for instance the case where firm i's holdings of quota share for species j, i.e., $\alpha(i, j)$, is excessive. This means that some other firm, firm k, say, can make more profits utilizing this quota share than firm i. This means that its $\alpha(k, j)$ is too small. This implies that firms i and k can engage in mutually profitable trades in the quota market. This argument shows that no firm will maximize its own profits unless its quota holdings are socially optimal[7].

It is convenient for later reference to state this result formally:

Lemma 1 *Given the time path of total quotas, $\{\boldsymbol{Q}\}$, quota trades will ensure that the share quotas, $\alpha(i, j)$'s, are socially optimal at each point of time.*

According to Lemma 1 the fisheries manager does not have to worry about the allocation of share quotas under the ITSQ system. Hence, it should be clear that under the ITSQ system the fisheries management problem is reduced to selecting the optimal time path of the vector of total quotas, $\{\boldsymbol{Q}\}$.

Single Species Fisheries Management

Although the reduction of the fisheries management problem under the ITSQ system to that of determining the optimal time path of total quotas, $\{\boldsymbol{Q}^*\}$, say, constitutes a great advantage, the problems of determining $\{\boldsymbol{Q}^*\}$ should not be underestimated. In fact, since $\{\boldsymbol{Q}^*\}$ depends on all the biological and economic relationships of the fisheries, the fisheries manager has to know every detail about these same relationships in order to calculate the optimal total quotas.

The ITSQ system, however, offers an alternative approach that efficiently exploits market information thus enabling the quota authority to identify the optimal total quota path with minimal information. The fundamental idea is that the prices in the share quota market prices reflect all relevant information about current and future conditions in the fishery available to the fishing firms or, for that matter, any participant in the quota market. It follows that the quota authority only has to monitor the quota market price to become privy to the same information.

We will now proceed to clarify this idea. In order keep the discussion as simple as possible we restrict the analysis to a single species fishery.

The previous profit maximization problem of a representative fishing firm, restated in terms of a single fishery may be written as:

$$\text{(III)} \quad \underset{\{z\}}{\text{Max}} \int_0^\infty (p\alpha Q - C(E(\alpha Q, \boldsymbol{x})) - sz)\exp(-rt)\,dt,$$

Subject to

$$\text{(a)} \quad \alpha' \equiv \partial\alpha/\partial t = z$$
$$\text{(b)} \quad 1 \geq \alpha \geq 0.$$

Assuming that the upper bound on α is not binding[8] the current value Hamiltonian for this problem may be written as:

$$(11) \qquad H = p\alpha Q - C(E(\alpha Q, \boldsymbol{x})) - sz + \sigma z,$$

where σ is the shadow value of an additional unit of share quota to the firm.

The necessary conditions for a solution to problem (II) include:

$$(11a) \qquad s = \sigma, \text{ for active firms,}$$
$$s \geq \sigma, \text{ for inactive firms,}$$
$$(11b) \quad \sigma' - r\sigma = \left(p - C_E E_q\right) Q \equiv -\pi_q Q = -\pi_\alpha.$$

According to (11a) firms should purchase additional units of share quotas in the market as long as their shadow value exceeds their market price and vice versa. Profit maximization may also require some firms to sell all their share quotas and come to rest with no quota holdings at a point where $s > \sigma$. Equation (11b) is the equation of motion for the shadow value of additional quotas.

Combining (11a) and (11b) yields the time path of quota prices.

$$(12) \qquad rs - s' = \pi_\alpha.$$

Equation (12) may be regarded as the fundamental dynamic demand function for share quotas[9]. The LHS of this

[7]This theorem can, of course, be proven rigourously. See Arnason (1990).

[8]If it were, the firm would be a sole owner and the efficiency problem would disappear.

[9]This equation may be recognized as a generalization of the famous arbitrage rule for asset holdings due to Hotelling (1931).

equation may be interpreted as the cost of holding a share quota. The term rs represents the opportunity cost of holding a share quota compared to investing its market value. The term s' measures the capital gain/loss of holding a share quota due to its instantaneous price changes. The sum of these two terms is the total cost of holding a share quota. The RHS of equation (12), on the other hand, is the marginal profits of share quota holdings. It represents the economic benefits to the firm of utilizing an additional unit of share quota for fishing.

The dynamic demand function for share quotas, equation (12) above, holds for all active firms at all times. Provided the fishery is pursued at all, only the price path defined by (12) is compatible with private profit maximization and, consequently, equilibrium in the quota market. Multiplying this equation for each firm i by its private profit maximizing quota level, $q^*(i)$, we find:

$$(s' - rs)\, q^*(i) = -\pi_\alpha q^*(i), \text{ all } i \text{ and } t.$$

Summing over all active firms and remembering that $\sum_i q^*(i) = Q$, yields:

$$(13) \qquad (s' - rs) = -\sum_i (p - C_E E_q)\, q^*(i), \text{ all t,}$$

where the RHS represents total resource rents generated in the fishery.[10]

Solving this differential equation for an arbitrary initial time $t = 0$, we obtain the fundamental share quota price equation (remember that $r > 0$):

$$(14)\ s(0) = \int_0^\infty \sum_i \left[\left(p - C_E\left(E_q\left(q^*(i), x\right)\right)\right) q^*(i)\right] \times \exp(-rt)\, dt.$$

Now, $s(0)$ is the market price of share quotas at time 0. The RHS is the present value of all future resource rents in the fishery. More precisely, as the future is unknown, $(p - C_E E_q)q^*(i)$ must be expected resource rents. In fact, since (14) is based on private profit maximizing behaviour, the variables on the RHS of (14) must represent the expectations of the firms in the fishing industry.

Expected rents, however, are not necessarily equal to actual rents. On the other hand, future conditions in a fishery are normally unknown and, consequently, expectations have to be relied upon. The crucial point is that the best available predictor of future conditions in the fishery is likely to be the expectations of the fishing firms. This statement can be justified on a number of grounds[11]. First, the fishing firms are generally more knowledgeable about their own economic conditions than anyone else. Second, accurate predictions are vital to their profitability and commercial survival. Third, the expectations of the firms are revealed by their trades in the quota market. By these very same trades the firms are really waging their money on the quality of their expectations.

Equation (14) thus establishes the following fundamental proposition of the ITSQ system:

Proposition 1 ***(Share quota values equal resource rents)*** *Under the individual transferable share quota system, ITSQ, defined in section 3, the market value of outstanding quotas equals the present value of expected future resource rents generated in the fishery.*

Now at a point in time, total catch equals total catch quotas, i.e., Q. Therefore, according to the biological growth function, equation (3), the future path of biomass depends only on the initial biomass level, $x(0)$ and the path of total quotas, $\{Q\}$. More formally:

$$x(t) = X\left(x(0), \{Q\}\right).$$

Moreover, by the ITSQ system

$$q^*(i) \equiv \alpha^*(i) Q.$$

Finally, as expressed in Lemma 1, the q^*'s are determined optimally by individual firms as a function of $x(0)$, $\{Q\}$ and the exogenous variables p and r. Thus, it is readily seen that the RHS of (14) depends only on the path of total quotas, $\{Q\}$, the initial biomass level, $x(0)$, and the exogenous variables p and r. It follows that adjusting the total quota so as to maximize the value of outstanding quotas is equivalent to maximizing the present value of expected future resource rents in the fishery. This establishes the following proposition:

Proposition 2 ***(Single species fisheries management)*** *Under the ITSQ system, adjusting total quotas so as to maximize the value of outstanding share quotas is equivalent to maximizing the present value of economic rents in the fishery.*

$$\operatorname*{Max}_{\{Q\}} s(0) \Leftrightarrow \operatorname*{Max}_{\{Q\}} \sum_i \int_0^\infty \left(p - C_E\left(E_q\left(q^*(i), x\right)\right)\right) \times q^*(i) \exp(-rt)\, dt.$$

The objective of fisheries management, however, is not to maximize resource rents. The management objective is to maximize aggregate profits in the fishery. The difference between profits and resource rents are the so-called intramarginal rents that are earned by the more efficient firms. Only when the firms are equally efficient will profits coincide with resource rents. On the other hand, at least when the firms have access to the same technology, there are grounds to expect equally efficient firms and thus equality between

[10] The concept of resource rents was employed by Gordon in his seminal paper on the fisheries problem in 1954. For a fuller discussion of resource rents in fisheries see Copes (1972).

[11] Fundamentally, however, it is based on the principle of rational expectations. See e.g., Lucas and Sargent (1981).

resource rents and profits. A simple argument is sufficient to establish this.

Recall from equation (13) that instantaneous resource rents equal the opportunity cost of holding quotas, i.e.,

$$\sum_i \pi_q q^*(i) = (rs - s').$$

If now profits are greater than rents, i.e.,

$$\sum_i \pi > \sum_i \pi_q q^*(i) = (rs - s')$$

it would be profitable for new firms enter the fishery by purchasing quotas. But then quota price would rise and this situation cannot be an equilibrium. Conversely, if profits were less than economic rents the opportunity cost of holding quotas would exceed the profits of utilizing these quotas. Hence firms would sell quotas and the quota price would decrease. For these reasons, one would expect resource rents to approximately coincide with profits at least in competitive equilibrium. On this basis we have the following assumption:

$$\pi = \pi_q q^*(i), \text{ all i.} \tag{15}$$

It may be noticed that condition (15) implies that the each firm minimizes the cost of filling its quota. This attribute is often associated with the catch quota system.

The above arguments have established the third fundamental proposition of this paper.

Proposition 3 ***(Single species fisheries management)*** *In the individual transferable share quota system, ITSQ, defined above and given the assumption expressed in equation (15) adjusting current total quotas to maximize the market value of total outstanding quotas at each point of time is equivalent to the maximization of expected profits attainable from the resource. Formally:*

$$\underset{\{Q\}}{\text{Max}}\ s(0) \quad \Leftrightarrow \quad \underset{\{Q\}}{\text{Max}} \sum_i \int_0^\infty (pq^* - C(E(q^*, x))) \times \exp(-rt)\, dt.$$

The practical implications of Propositions 2 and 3 are obvious. To identify the optimal total quota at a point of time, the quota authority does not have to collect data about the fish stocks and the economics of the harvesting firms. It has merely to monitor the share quota price in the quota market and adjust the total quota so as to maximize the total value of the share quotas. For obvious reasons, we refer to this management method as the Minimum Information Management method (MIM).

Notice that, the quota authority does not have to announce the future time path of total quotas in order to locate the optimal current one. Relying on the expectations of the quota market players, the quota authority only has to announce its long term objective in setting future total quotas and then select the current total quota.

For given expectations about future total quotas we may illustrate the relationship between the current total quota and resource rents as presented in Fig. 1.

There appear to be many ways of carrying out the search for the optimal total quota required by the Minimum Information Management method. One is to imitate the normal market adjustment processes as for instance exemplified by the Walrasian tatonnement process[12] by iteratively announcing the total quota[13] and checking the response of the quota price. Presumably, for most fisheries, this could be carried out in a modified stock exchange setting. In any case, compared to solving the individual optimization problems directly this particular search for the optimal TAC appears relatively tractable.

Propositions 2 and 3 depend heavily on the rationality of the fishing firms and the quota market players. Notice, however, that if the firms are not rational and do not formulate rational expectations it is hard to imagine any decentralized arrangement of the fishery that will attain economic efficiency. This means that even when the fishing firms are less than rational the best management procedure may still be to follow the prescriptions of the MIM procedure.

Ecological Fisheries Management

The MIM fisheries management technique described above can be easily extended to the ecological context. As explained in section 3, the private profit maximization problem in the ecological context is:

$$\underset{\text{all } \{z(j)\}}{\text{Max}}\ J = \int_0^\infty \left(\sum_j p(j)\, \alpha(i, j) Q - C(E(q(i), x); i) - \sum_j s(j) z(i, j) \right) \times \exp(-rt)\, dt$$

Subject to

$$\text{(a)} \quad \alpha'(i, j) = \partial \alpha(i, j)/\partial t = z(i, j),$$
$$\text{(b)} \quad 1 \geq \alpha(i, j) \geq 0.$$

The necessary conditions for solving problem (IV) yield the following dynamic share quota demand equations for firms active in a particular fishery:

$$rs(j) - s' = \pi_\alpha(i, j), \text{ all } j, \text{ and active } i. \tag{16}$$

Proceeding as in the single species case we obtain the fundamental quota share price equations at an arbitrary initial time 0:

[12] On the Walrasian tatonnement process see e.g., Nagatani (1981) p. 57.

[13] To facilitate this process, the "auctioneer" may choose to follow the prescriptions of a numerical search algorithm

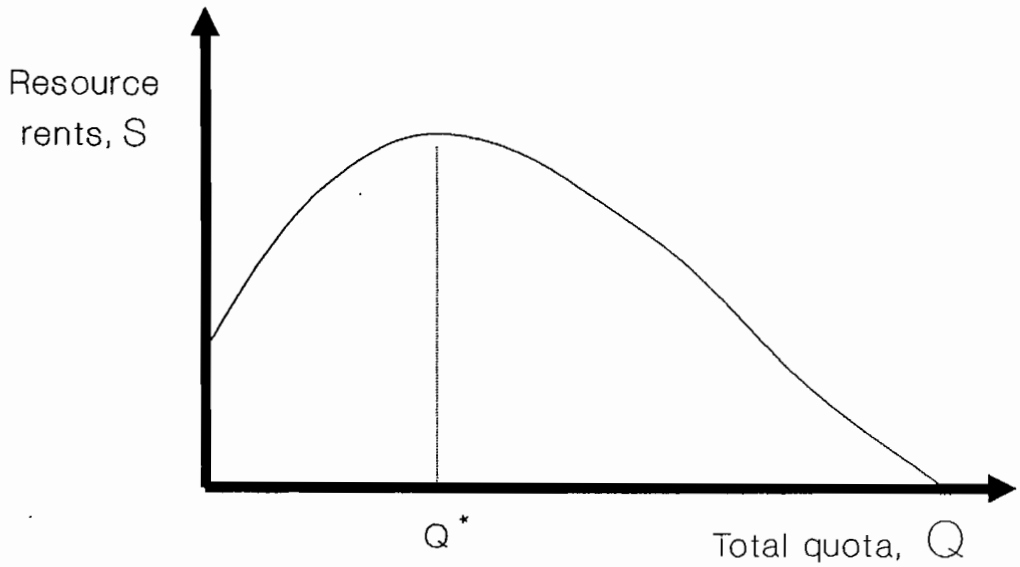


Fig. 1. Resource Rents and Total Quotas.

$$s(0, j) = \int_0^\infty \sum_i \left[\left(p - C_{\boldsymbol{E}}\left(\boldsymbol{E}_{q(i,j)}\left(\boldsymbol{q}^*(i), \boldsymbol{x}\right)\right)\right) \times q^*(i, j)\right] \exp(-rt)\, dt, \text{ all j.}$$

This equation has the same economic interpretation as the corresponding one in the single species case. In particular, the RHS of (17) represents the expected present value of resource rents from fishery j. Moreover, as in the single species case, if the firms have access to the same technology, equilibrium resource rents coincide with profits at least approximately. In that case, the RHS of (17) may be identified with the present value of expected future profits in the fishery.

Notice that ecological interdependence between the species is represented in the quota price equation by the vector of future stock sizes, $\boldsymbol{x}$. This, as is made clear in the biological growth function, equation (3), depends on current stock sizes and the future path of harvesting. In the ITSQ system, the expectations of fishing firms concerning future total quotas as well as their perceptions of ecological relationships are reflected in quota prices. It should be noted, that ecological interdependence is not the only connection between different fisheries in equation (17). As discussed in section 3, bycatch provides another connection.

Summing over fisheries yields the present value of expected economic rents in all the fisheries

$$R(0) = \sum_j s(0, j) = \sum_j \int_0^\infty \sum_i \left[\left(p - C_E\left(E_{q(i,j)}\left(\boldsymbol{q}^*(i), \boldsymbol{x}\right)\right)\right) \times q^*(i, j)\right] \exp(-rt)\, dt.$$

Now, as mentioned, future biomass levels only depend on initial biomass levels and the future path of total quotas, $\{\boldsymbol{Q}\}$. Moreover, by the structure of the ITSQ system, individual quotas are defined by $q^*(i, j) = \alpha^*(i, j)Q(j)$. As discussed earlier, however, profit maximizing quota trades ensure that the $\alpha(i, j)$'s always coincide with the socially optimal allocation given $\{\boldsymbol{Q}\}$. It follows that $R(0)$, the present value of total expected rents in the fisheries, is a function only of the future path of total quotas, $\{\boldsymbol{Q}\}$, initial biomass levels, $\boldsymbol{x}(0)$, and the exogenous prices $\boldsymbol{p}$ and r. More formally,

$$R(\boldsymbol{Q}) = R(\{\boldsymbol{Q}\}, \boldsymbol{x}(0), \boldsymbol{p}, r). \tag{17}$$

This observation establishes the multi-species versions of Propositions 2 and 3 as follows:

Proposition 4 ***(Ecological fisheries management)***
In the ITSQ system defined in section 3 adjusting current total quotas so as to maximize the current total market value of quota share prices is equivalent to maximizing resource rents in all the fisheries. Formally:

$$\operatorname*{Max}_{\{\boldsymbol{Q}\}} R(\{\boldsymbol{Q}\}, \boldsymbol{x}(0), \boldsymbol{p}, r) \Leftrightarrow$$

$$\operatorname*{Max}_{\{\boldsymbol{Q}\}} \sum_j \int_0^\infty \sum_i \left[\left(p - C_{\boldsymbol{E}}\left(\boldsymbol{E}_{q(i,j)}\left(\boldsymbol{q}^*(i), \boldsymbol{x}\right)\right)\right) \times q^*(i, j)\right] \exp(-rt)\, dt$$

Proposition 5 ***(Ecological fisheries management)***
In the ITSQ system, if all fishing firms maximize average profits of quota holdings, adjusting current total quotas to maximize the market value of total outstanding quotas at each point of time is equivalent to the maximization of expected profits attainable from the resource.

Propositions 4 and 5 extend the Minimum Information Management (MIM) principle to the ecological context. To achieve maximum benefits from the fisheries, the fisheries manager should simply impose the system of individual transferable share quotas (ITSQ) for all relevant species and then set total catch quotas so that the aggregate value of all quota prices is maximized. Instead of announcing a total quota for a single species, however, the fisheries manager must now announce a vector of total quotas for all relevant species in the ecology. As before, profit maximizing fishing firms may be relied on to gather and interpret the pertinent biological and economic information in the most efficient manner. This in formation will be reflected in market quota prices. Given this, there will be a vector of total quotas that maximizes aggregate resource rents from utilizing the ecology. On the assumptions discussed above, this vector constitutes the best fisheries policy.

In the ecological context, the task of locating the maximum of the $R(\boldsymbol{Q})$ surface will be substantially more demanding than in the case of a single species. First, the search takes place over a possibly high dimensional vector of total quotas. Secondly, ecological relationships may involve complexities including multiple biomass equilibria and bifurcations. It stands to reason that these ecological complexities will be reflected in the shape of the $R(\boldsymbol{Q})$ relationship. Hence, this function may exhibit nonconcavities and discontinuities. It must be emphasized, however, that these difficulties reflect fundamental ecological complexities and the Minimum Information Management technique still appears to offer the most efficient approach to deal with these.

An interesting feature of ecological fisheries management within the ITSQ framework is that some share quota prices may well be negative. The reason is that the optimal ecological fisheries policy will usually require the reduction in the stock size of some species of fish that are themselves not valuable but prey on or compete with commercially valuable species. The quota price for these species would be negative representing harvesting subsidies. In this case, contrary to the one of positive quota prices, profit maximizing quota holders would prefer not to spend economic resources catching their share quota. Therefore, in the case of negative quota prices, the ITSQ requirement that quotas be fulfilled must be imposed[14]. Clearly, profit maximizers will only assume such an obligation for a payment. This is represented by the negative quota price or harvesting subsidy mentioned. Initially therefore the negatively priced share quotas will normally be allocated to the firm requiring the lowest subsidy. It may be noted that a negative quota price with the requirement of fulfillment corresponds to a payment for the eradication of pests and predators that is commonplace in traditional agriculture.

Another interesting feature of ecological fisheries management with the help of the ITSQ system is that optimal total quotas for some species might be negative. This means that the corresponding share quotas holders would be under the obligation of enhancing the species in question to the extent stipulated by their quota holdings. Thus, it appears that ecological fisheries management with the help of share quotas naturally accommodates fish stock enhancement as a dual to harvesting. Again, if a quota price for a negative total quota is negative it represents a subsidy for fish stock enhancement. That would occur in the case of socially optimal but privately profitable stock enhancement. Alternatively, stock enhancement may be privately profitable. A case in point is ocean ranching of valuable species. For ecological reasons the quota price would normally be positive indicating that the ocean ranching firms would pay for the privilege of releasing fish into the ocean. These polar cases of ecological fishes management are summarized in Table 1.

Table 1. Total Quota and Share Quota Price: Polar Cases.

	Total Quota, $\boldsymbol{Q}$	
Quota Price, s	Negative	Positive
Negative	Unprofitable Stock Enhancement	Unprofitable Fishery Predator/competitor stock reduction
Positive	Profitable Stock Enhancement (Ocean ranching)	Profitable Fishery (Commercial Fishery)

Uncertainty and Risk

Uncertainty enters the fisheries problem in many ways. Fisheries production and cost functions are subject to uncertainty due to stochastic movements in prices and productivity of economic inputs. Current fish stocks are measured with errors and future ones are uncertain due to biological randomness. The fishing firms, moreover, are subject to additional uncertainty concerning the future course of total quotas.

In general, the existence of uncertainty affects the nature of the optimal fisheries policy[15]. In the context of this paper the question is what impact uncertainty has on the performance of the Minimum Information Management (MIM) method described in the previous two sections.

Notice first that individual fishing firms will of course respond to the uncertainty they are faced with. They will, in other words, modify their behaviour accordingly. In particular, this will be reflected in their demand for share quotas and, consequently, in the share quota price which, in turn, transmits management signals to the fisheries manager. From this, it follows that when the fisheries manager maximizes the share quota price by adjusting the total quota he is in fact taking account of the uncertainty in the fishery. This is, of course, most encouraging. The disadvantage, however, is that the uncertainty perceived by the fishing firms may not coincide with the socially appropriate one.

[14]Moreover, for a similar reason it may become necessary to keep track of who actually holds negatively priced share quotas.

[15]There are special cases under which the optimal fisheries policy under uncertainty is identical to the one under certainty.

In order to make these ideas a little more precise let us examine briefly the single species case. Under uncertainty the representative firm seeks to maximize the expected value of the present value of future profits:

$$\text{(IV)} \quad \underset{\{z\}}{\text{Max}} \quad \Phi\left[\int_0^\infty (p\,\alpha Q - C\,(E\,(\alpha Q, x)) - sz) \times \exp(-rt)\,dt\right].$$

Subject to

$$\text{(a)} \quad \alpha' \equiv \partial\alpha/\partial t = z$$
$$\text{(b)} \quad 1 \geq \alpha \geq 0,$$

where Φ denotes the expectations operator.

The solution to problem (IV) is quite hard to characterize in general. It depends on the probability distributions of the uncertain variables and the functional forms involved. However, on simplifying assumptions it is possible to make some progress.

From the point of view of the individual firm, all price variables in this problem, i.e., p, s and r, are uncertain. So also are future quotas, $\boldsymbol{Q}$, and fish biomass, x. On the other hand, trade in quotas, z, being a control variable is not stochastic. Neither is the quota share, α, since it is completely controllable by z.

It follows that under the ITSQ system the firm is only subject to exogenous uncertainty or risk (See Dasgupta and Heal (1979, p. 416–17). It can do nothing to influence its risky environment. Given this, assuming independence and stationarity of the respective distributions and making certain other simplifications it is possible to represent the risk as an additional discount factor: More precisely,

$$\Phi\,(\pi^\circ) = \pi \exp(-mt), \tag{18}$$

where π° represents uncertain income t periods hence. π represents a certain income and m is the risk factor.

Thus, on these simplifying assumptions, problem (IV) can be rewritten as a non-stochastic control problem with a new discount rate, $i = r + m$, representing the sum of the time discount rate, r, and the risk factor, m:

$$\text{(V)} \quad \underset{\{z\}}{\text{Max}} \int_0^\infty (p\,\alpha\boldsymbol{Q} - C\,(E\,(\alpha\boldsymbol{Q}, \boldsymbol{x})) - sz) \exp(-it)\,dt.$$

Subject to

$$\text{(a)} \quad \alpha' \equiv \partial\alpha/\partial t = z$$
$$\text{(b)} \quad 1 \geq \alpha \geq 0,$$

Now, the risk factor, m, may be either negative or positive depending on the probability distributions and functional forms. If it is positive, fishing firms will value current profits more highly than the present value of more uncertain present values of future profits. If that is the case, fishing firms will prefer higher quotas now than would otherwise be the case. Consequently, the equilibrium biomass level will be less than under certainty. If m is negative, the opposite result holds.

Solving problem (V) for all firms yields the same equations as earlier. In particular, the quota market price is given by:

$$\text{(19)} \quad s(0) = \int_0^\infty \sum_i \left[\left(p - C_E\left(E_q\left(q^*(i), \boldsymbol{x}\right)\right)\right) q^*(i)\right] \times \exp(-it)\,dt.$$

This expression demonstrates the earlier assertion that the quota market price will indeed depend on the risk facing private fishing firms.

Now, referring to our earlier analysis, maximizing the quota price by adjusting total quotas is equivalent to maximizing the RHS of (19). The social fisheries problem under uncertainty, however, is to maximize the expected value of economic rents, namely:

$$\text{(VI)} \quad \underset{\{Q\}}{\text{Max}} \quad \Phi\left[\sum_i \int_0^\infty (p\,\alpha Q - C\,(E\,(\alpha Q, x)) - sz) \times \exp(-rt)\,dt\right].$$

It follows that maximizing $s(0)$ only solves the social fisheries problem under uncertainty if the social risk of the fishery can be expressed in the same way and is identical to that of individual firms.

There are several reasons why this may not be the case. First, part of the risk facing individual firms concerns the future path of total quotas. For the fisheries manager, total quotas are control variables and therefore not stochastic, at least not to the same extent. This suggests that individual fishing firms may regard the fisheries situation as more risky than the fisheries manager. Second, society is a bigger entity than individual fishing firms. Therefore, for the usual reasons it will tend to be less risk averse than individual firms.

This suggests that the private risk factor would tend to exceed that of the fisheries manager. In that case, quota prices would tend to be maximized at a total quota in excess of the socially optimal level. If this is actually the case, the quota authority should select total quotas below the level that maximizes quota prices. This case is illustrated in Figure 2.

On the other hand, it should be noted that the share quotas will tend to be held by the fishing firms that are most optimistic about future conditions and least risk averse. These firms will, *ceteris paribus*, be willing to pay higher prices for share quotas than more pessimistic firms. This suggests that the share quota price may reflect more optimism and less risk

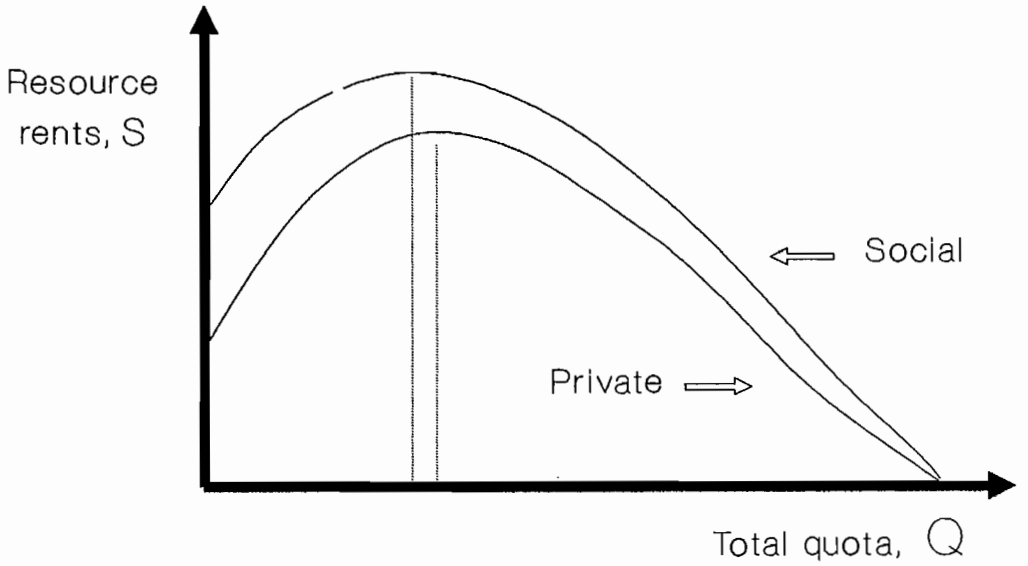


Fig. 2. Resource Rents and the Total Quota: The Uncertain Case.

aversion than would hold for society as a whole. Hence, the risk factor, m, would be lower and the MIM method will result in lower quotas than would otherwise be the case.

These considerations show that, in terms of the very simple risk model defined above, it will generally not be the case that the private risk factor equals that of the fisheries manager. It follows that the MIM method as described above will generally not result in the optimal stochastic fisheries policy. However, given that the quota price will reflect the risk perceptions and attitudes of private fishing firms, it may well be the case that the difference is not economically significant.

Conclusion

The individual transferable share quota system (ITSQ) offers an extremely simple way to manage ocean fisheries with a minimal centralized collection of information. In the non-stochastic case the fisheries manager only has to manipulate total current quotas until aggregate quota values are maximized. Under fairly nonrestrictive assumptions this method can be shown to maximize aggregate resource rents both in the single-species and the multi-species case.

Under uncertainty, moreover, this result must be modified. Social and private perceptions of and attitudes to risk are generally not identical. For this reason maximization of quota values will in general not correspond to maximization of social benefits from the fisheries. The difference, however, is likely to be relatively small.

It is interesting to notice that the Minimum Information Management (MIM) method greatly reduces the need for centralized research into the conditions of the fishery. Once the ITSQ system has been installed, the fishing industry finds itself in an operating environment very similar to that of traditional land-based industries. Consequently, the fishing firms will find it in their interest to collect necessary data and do the appropriate research for the maximization of their profits. In fact, those firms that fail to do so will be subject to comparatively poor economic results and ultimately leave the fishery. All of the information collected by the fishing firms that is relevant for management purposes will be communicated to the fisheries manager via the quota prices. Hence, under ideal conditions, there is little reason to engage in costly centralized research.

This implication of the MIM method, however, is of course too simplistic. Partly it is due to the aggregative nature of the model used. In a more realistic setting, optimal fisheries management would not only be concerned with the aggregate biomass but also cohort management, spawning stocks, nursery grounds etc. These added complications do not, in practice, appear amenable to market based management techniques. In addition, as has been suggested above, knowledge about the conditions of the fishery will greatly facilitate the search for the optimal total quota, especially under conditions of uncertainty.

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Bioeconomic Methods for Determining TAC's

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The paper reviews the literature on bioeconomic reference points on the setting of annual total allowable catches (TAC's). Alternative strategies for determining TAC's are compared with respect to their objectives (e.g., maximization of resource rents, harvesting profits), the mechanism used to describe population dynamics (e.g., aggregate production models, or age-structured models), the temporal treatment of the problem (e.g., static or dynamic, short or long term), and consideration of risk and uncertainty (e.g., deterministic or stochastic modelling). The focus of the review is on conceptual and applied research that has made an impact on actual management policies in fisheries. Difficulties associated with the standardized biological reference point approach that forms the basis of current scientific advice are discussed. A more general conceptual framework would enable managers to take direct account of biological constraints and economic objectives. A comparative analysis of alternative TAC schedules is developed for the case of 2J3KL Northern cod. Alternative schedules incorporate the most recent bioeconomic data and permit the evaluation of alternative management strategies, allowing for a long run strategic view of the resource and the fishery.

L'article examine les travaux publiés sur les points de référence bioéconomiques touchant l'établissement des totaux des prises admissibles (TPA) annuels. D'autres stratégies d'établissement des TPA seront comparées en termes d'objectifs (p. ex. maximisation de la rente de la ressource, excédents sociaux, profits d'exploitation), de mécanisme servant à décrire la dynamique de la population (p. ex. modèles de la production globale, ou modèles de la structure d'âge), du traitement temporel du problème (p. ex. statique ou dynamique, à court ou à long terme), et d'examen des risques et de l'incertitude (p. ex. modélisation déterministe ou stochastique). L'examen sera axé sur la recherche conceptuelle et appliquée qui a eu un effet sur les politiques réelles de gestion dans le domaine des pêches. Nous développons une analyse comparative de différents régimes de TPA dans le cas de la morue du nord. Les résultats montrent les difficultés liées à la méthode étroite et normalisée des points de référence biologiques qui est à la base des avis scientifiques actuels. Un cadre conceptuel plus général appliqué individuellement à chaque pêche permettrait aux gestionnaires de tenir compte directement des objectifs biologiques et économiques, d'intégrer les données bioéconomiques les plus récentes et d'envisager l'élaboration et l'évaluation de stratégies différentes de gestion, et donnerait une vue stratégique à longue échéance de la pêche.

Canadian fisheries policy and management are mandated by the Fisheries Act. The Fisheries Act was first enacted in 1868. It has provided the Government of Canada with policy goals for nearly 125 years. Broadly stated, these goals are: (1) conservation of fish resources; (2) development of commercially viable Canadian fisheries; and (3) distribution of fish resources among Canadians who benefit from these resources (Laubstein 1987). From an operational perspective, the mandate has been implemented through fisheries management policies that incorporate (1) biological assessment of stock status for sustained growth, (2) the determination of seasonal exploitation limits or TAC's (total allowable catches), and (3) the allocation of harvesting rights to competing fishermen.

While effective fisheries management policy ought to integrate all three of these objectives, it is important to note that crises in Canada's ocean fisheries in the 1980's have most often been the result of difficulties in maintaining economic viability of the commercial fishery, and not solely because of deteriorated biological status of stocks. As Mitchell (1981, p. 191) states,

> . . . fisheries problems although due to biological factors are largely socioeconomic in nature. As a result, biological measures alone are incapable

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of solving them and it is now recognized that a bioeconomic approach is necessary for fisheries management.

In a similar vein, Clark(1985) argues that while fisheries biology has developed a "scientific" method of analysis to support its objectives, a similar mechanism is absent from applied fisheries economics. Rather, the dual economic-based issues of overall viable economic performance and allocative equity have tended to divide the economic research community. The domination of the distributional issues faced by the operational management agencies have, in Clark's view, drawn resource economists away from the study of the overall economic performance of the fishery. The result has been a domination of biological-based analyses that fills the apparent policy void.

The purpose of this paper is to (i) review alternative bioeconomic approaches for determining static and dynamic levels of seasonal exploitation levels (TAC's) for commercial fisheries, and (ii) evaluate the impact these approaches have on an actual fishery. Specifically, this paper focuses on methods for establishing annual exploitation limits that take into account economic variables and objectives as well as biological factors in an integrated bioeconomic framework. A comparative analysis of simple biological and economic performance measures of alternative TAC strategies is presented using data from the 2J3KL Northern cod fishery.

In order to put the discussion of alternative bioeconomic approaches in context, a brief overview of harvesting strategies including a sketch of the historical developments in the setting of global annual allocation limits is presented.

Exploitation Regimes and Historical Developments

Policies for setting annual or seasonal exploitation limits can be classified into three general types of catch controls. Annual total allowable catch limits may be based on: (1) a minimum required escapement of the reproductive stock; (2) removal of a fixed fraction of the harvestable stock (fixed fishing mortality); and (3) direct fixing of a constant maximum level of catch. The determination of catch limit values for each of these approaches assumes some underlying reference point or long-term stock size objectives. Moreover, conceptually, feasible long-term stock objectives can be achieved through appropriate manipulation of any of these three exploitation regimes.

The historical development of setting exploitation limits in Atlantic Canada centres around the selection of appropriate biological reference points for determining TAC's. Prior to the 1950's most management measures relied on catch and fishing effort statistics and crude biologically-based models were used to justify management regulations. Operational measures were directed at restricting fishing effort. Gear limitations and time and area closures dominated as management instruments. The mounting patchwork of input based regulatory measures did not work because of "free" entry or "open access" to the fishing grounds and improved efficiency of fishing vessels. The incentive for profit-taking remained and overfishing pressure on fish stocks increased. This necessitated a spiraling of still stricter regulatory measures designed to curtail harvesting activity in order to protect dwindling stocks.

In the early 1960's the International Commission for the Northwest Atlantic Fisheries (ICNAF) responded to mounting pressure on fish stocks by introducing controls on the minimum mesh sizes used for net captures. The theory behind this regulation had been developed by Beverton and Holt (1957) in their analysis of depleted North Sea herring stocks. Beverton and Holt modeled the fish population by considering the abundance of different ages or "cohorts" of the population. Assuming known biological data on growth and natural mortality, led them to the determination of maximum equilibrium catch (yield) from fishing on selected older cohorts through the control of the size of mesh in fishing nets. Their analyses determined fishing mortality levels and corresponding yields at equilibrium from fishing stock cohorts above a specified age (or size). They argued that setting mesh size regulations appropriately would theoretically permit escapement of fish of sizes smaller than the average required to maximize yield.

By the late 1960's however, it became evident that effort restrictions in open access fisheries were not sufficient to curb high catches and halt the depletion of stocks. On the impetus of Gordon's (1954) economic arguments which showed the weaknesses of open access fisheries, many fisheries during this period were "closed" to new entrants and "limited entry" fisheries became commonplace. At the same time, direct catch limits were contemplated. Early in the next decade, ICNAF inaugurated global catch limits for all member countries on some groundfish stocks in its area. Then in 1972, ICNAF set the first national allocations for Canada and the United States on herring stocks. By 1974, most groundfish species were being managed by annual TAC's (Halliday and Pinhorn 1985).

The implementation of TAC schemes was not without difficulties. There were two major issues associated with annual exploitation limits: (i) how to determine the appropriate size of the total quota, and (ii) how to allocate the global quota among mixed fleets. Initially, models of population dynamics were assumed in order to establish specific reference point fishing mortality rates, e.g., to maximize the yield per recruit (F_{max}), or aggregate stock measures, e.g., to maximize the sustainable yield, MSY. Fundamental reliance on the more detailed age-structured model for population dynamics established the "fixed fishing mortality" exploitation regime as the operative foundation for setting annual TAC's.

However, subsequent analyses on generalized growth model and stochastic modelling applications soon affirmed that the static optimum underpinnings of TAC's based on fixed fishing mortality targets (e.g., F_{max}) could give undesirably high levels of actual catch (Doubleday 1976, Beddington and May 1977, Clark and Munro 1978). F_{max} and MSY were ultimately rejected as inappropriate for determining TAC's

and the search was on for a more conservative fixed fishing mortality standard.

In response, John Gulland, as part of an ICNAF working group charged with resolving the issue, put forward a generally applicable fishing mortality reference point for determining TAC's (Gulland 1968). Using Beverton and Holt's yield per recruit analysis, he proposed the "$F_{0.1}$" fishing mortality level for determining annual TAC's. The $F_{0.1}$ reference point is defined as that level of fishing mortality for which the marginal yield (catch) per recruit due to a small increase in fishing activity is 10% of the marginal yield per recruit when the same stock is very lightly exploited over the long run. In his argument for an understandable, standardized, and parsimonious criteria on which to base TAC's, Gulland arrived arbitrarily at the figure of 10% as a more conservative approach than maximizing yield per recruit. Gulland (Gulland and Boerema 1973) assumed that

> ...a marginal yield of 10% of the initial catch [i.e., of an unexploited stock] per unit effort is not worthwhile.

The $F_{0.1}$ standard was introduced to ICNAF in 1972 (Anthony 1982). Annual TAC's were determined directly by applying the fishing mortality by age to the recruited proportion of the assessed stock size. Since 1975 more and more commercial fish stocks have come to be managed through annual TAC's based on the $F_{0.1}$ fishing mortality reference point. Moreover, $F_{0.1}$ values are required for nearly all Canadian Atlantic Fisheries Scientific Advisory Committee (CAFSAC) groundfish and pelagic stock assessments. This standard has also been a regular feature of assessments by working groups of the International Council for the Exploration of the Sea (ICES) and the North Atlantic Fisheries Organization (NAFO).

Current Practices and Issues

The $F_{0.1}$ reference level for annual stock exploitation had immediate operational impact, since it could be applied directly to the current status of the stock. Moreover, it was hoped that the long-run implications of $F_{0.1}$ as a conservative exploitation strategy would permit overexploited stocks to grow in the short-run. This built-in growth would further reduce the risk of declining stock abundance. In recent years, strict adherence to the $F_{0.1}$ fishing mortality rate were observed to lead to "drastic changes in TAC levels between successive years." This was a consequence of the observed year-over-year changes in stock assessment estimates. Recognizing the potentially negative short-term impacts on fishermen due to adjustment costs, a "50% rule" was established "by which the adjustment to $F_{0.1}$ would be phased in over time." The 50% rule used in CAFSAC advice sets the fishing mortality in the next year at a value "halfway between the current fishing mortality and $F_{0.1}$" (Rivard and Maguire 1993). As a decision rule, $F_{0.1}$ changed from being an operational decision value, to a longer run statement of "objective" in stock management.

In the spirit of the times in which $F_{0.1}$ appeared, characterized by over-capitalization of fishing fleets and overexploitation of fish stocks, and prior to the introduction of extended jurisdiction, its supporters lauded the perceived benefits of larger than current biomass levels. These benefits included projections of larger average weight per catch (as a result of more fish present at older age groups), improved rates of catchability, and anticipated reduced cost of fishing effort. Others argued that the reduced fishing mortalities implied by $F_{0.1}$ would lead to a more stable schedule of TAC's — a desirable economic consequence related to random fluctuations in harvestable stock biomass (Shepherd 1990).

The 50% rule was also seen by biologists to be the embodiment of economic principles that promised to "minimize the effects on future recruitment while promoting economic efficiency" (Norris 1991) or "optimize gains to fishermen" (O'Boyle and Koeller 1986). $F_{0.1}$ was widely embraced as the first real attempt to capture bioeconomic principles in an operational way in fisheries management. Recently, however, biological reference points like $F_{0.1}$ or the "50% rule" have come under attack by biologists and economists. The biological and economic shortcomings of $F_{0.1}$ have demonstrated the need for reconsidering the "standardized" approach to determining reference point underpinnings for the calculation of TAC's. In their investigation of the crisis situation in the Newfoundland inshore cod fishery, Alverson, Beamish, Gulland, Larkin, and Pope (1987) state that

> ... $F_{0.1}$ has no particular biological virtue as the target for future policy, although thought to be relatively conservative if achieved. It had the practical advantage in the days of ICNAF that it could be objectively determined from readily observable biological parameters (growth and mortality) and that it could be agreed upon as a target by countries with very different economic and social objectives.

Deriso (1987) elaborates on the fundamental reliance of $F_{0.1}$ on the Beverton and Holt yield per recruit analysis. The independence of yield-per-recruit analyses on underlying stock-recruitment relationships allows Deriso to develop formal conditions in the presence of stock-recruitment dependencies that show $F_{0.1}$ levels may actually exceed the MSY fishing mortality for stocks with low growth rates. Deriso (1987) also shows, given age at first capture, that $F_{0.1}$ can be calculated directly as a function solely of the two biological parameters, M, the constant rate of natural mortality applied to all population cohorts, and K, the von Bertalanffy growth parameter for the stock. Economic inputs are noticeably absent from the calculation. Moreover, the argument that long-run marginal yield from harvesting effort should be maintained above an arbitrary percentage of the unexploited equilibrium stock is not consistent with the basic economic principles of profit maximization in a dynamic environment.

The foundation of $F_{0.1}$ on static equilibrium yield per recruit analysis ignores the reality of dynamic and stochastic

variability both in terms of the stock and economic variables (Clark 1976a). Yield per recruit analyses (Gulland 1983) study the average weight of catch from each young fish entering the system assuming an 'equilibrium state' for the average life time of a fish (which can be 8 or 12 years). However, fish populations, fishing effort, technology and even markets all change over such a time scale and these tend to invalidate the model.

Fisheries management by fixed F emerged as a result of biologically-based science and stock assessment reacting to economic and social crises. It filled the void in operational management by trying to integrate biological and economic objectives. To overcome today's problems, a clearly defined strategic policy founded on integrated bioeconomic targets must be articulated. As Alverson, Beamish, Gulland, Larkin, and Pope (1987) state it:

> Ideally, management actions should flow from clear policies (e.g., "maximize the net economic returns from the fishery,") which can be expressed, following suitable analysis (e.g., of the costs and earnings under differing values of fishing effort,) as quantitative targets of, for example, fishing mortality or spawning stock size, and then, given conditions in the current year, into tactical measures, e.g., the TAC the following year. They should also take account of uncertainties and associated risks that must exist in assessments of every fish stock...While the choice of a strategic target must be a political one, based on the desired balance of social and economic objectives, which will usually be partly conflicting, the choice is likely to be sound only if preceded by an analysis of the impacts of alternative strategies on the relevant characteristics of the fishery, e.g., on the costs and earnings, or the extent of employment.

Recent research that deals with the shortcomings of $F_{0.1}$ has proposed fishing mortality levels at a higher or lower F value. For example, Andrew and Butterworth (1987) propose $F_{0.2} > F_{0.1}$ as a reference point on which to base TAC's for hake in ICSEAF Division 1.6. For the fishery, they show the economic advantages of using $F_{0.2}$ versus $F_{0.1}$. Lane (1989b) uses a simple model to estimate aggregate economic performance for a range of fishing mortality values under equilibrium conditions. Lane concluded that $F_{0.1}$ probably does not have desirable economic characteristics over time.

The objections raised about biological reference points and their current use in management are fundamental to the whole approach to developing bioeconomic TAC schedules. These objections arise from the direct association of biological constraints (e.g., minimum sustainable limits on stock size and number at age) with harvesting strategies without considering value-based objectives of the harvesting and processing activities. In the current management view, the definition of $F_{0.1}$-based TAC's represents the "scientific advice" on which policy decisions are taken. However, in the absence of economic input, and without evaluating other alternative fishing mortalities that take into account current stock status, it is reasonable to assume that there may exist dominating TAC strategies that give improved biological and economic performance, all other things being equal. What is required is a movement away from articulating a narrow, conceptual framework that supports a standardized approach, and toward the development of a more general strategic framework that can be applied specifically to different fisheries.

Static Bioeconomic Models

The work of Gordon and Schaefer is the earliest and most widely employed fisheries bioeconomic model. The economics of the model draws mainly from the work of Gordon (1954), Scott (1955), Crutchfield and Zellner (1962), and Copes (1970). The biological aspects are credited to Schaefer (1954).

Gordon's analysis of open access fisheries was based on Schaefer's simple model of aggregate logistic population growth. Gordon developed a static or equilibrium view of sustained yield from fisheries. The model postulates that the growth in aggregate weight of a fish stock depends on the aggregate weight of the stock at the beginning of each period. This relationship is the logistic growth function shown in Fig. 1. Within the range bounded by zero and the environmental carrying capacity (denoted by K), growth is nonnegative. At stock sizes of 0 and K the growth rate equals zero.

If the stock size is K as fishing begins, the stock initially declines by an amount equal to the catch rate since growth is zero at K. If catch is held constant at or below the maximum net reproductive potential of the stock, then a new equilibrium will be established where the stock size is large enough to allow the biological productivity of the stock to be equal to the catch. Since the stock is constant at this catch level, then theoretically (and, in particular, in the absence of random shocks), it can be maintained indefinitely and the constant catch would be a "sustainable yield". The sustainable yield at each stock size is equivalent to the net growth of the stock at that stock level. The sustainable yield curve in Fig. 2 resembles the biological productivity curve in Fig. 1 with its characteristic inverted parabola shape. However, the origin of the sustainable yield curve corresponds to stock size K on the biological productivity curve. Increased fishing effort is associated with a smaller stock size and vice versa.

The economics of the Gordon-Schaefer model are based on three simplifying assumptions:

1. Constant price. The price for fish at the port market level is exogenously determined and output in the fishery is small enough to have no effect on the overall market for fish. The constant price assumption means that the sustainable yield curve times price gives the total revenue (TR) curve for the fishery, i.e., a curve directly proportional to Fig. 2.

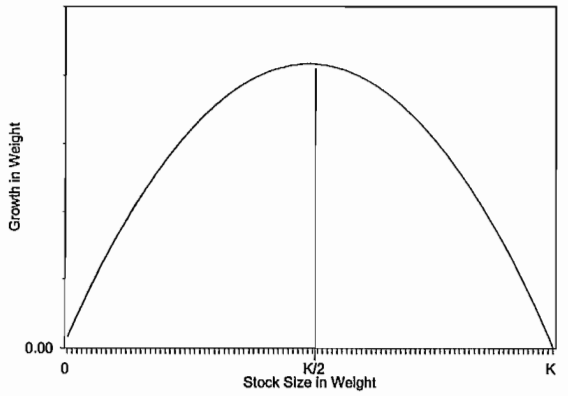


Fig. 1. Logistic model of aggregate population growth. Maximum stock growth occurs at one half the carrying capacity (K) of the stock. Source: Schaefer (1954).

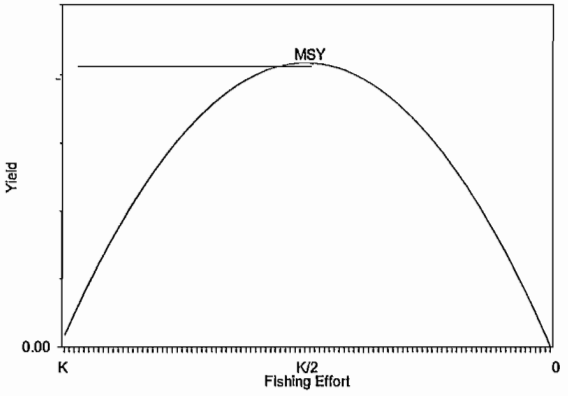


Fig. 2. Sustainable yield curve. MSY (Maximum sustainable yield) is found at fishing effort corresponding maintenance of the stock at its most productive level at one half the carrying capacity (K) of the stock.

2. Fishermen derive no non-monetary benefits from fishing. This assumption emphasizes that revenue and cost have primacy in calculations of wealth for fishermen.

3. Entry and exit of fishermen and boats adjust effort and fixed costs in the long run. In the short run, only variable costs are affected by adjustments in fishing effort. The total cost (TC) of effort is therefore assumed to be a linear function of the total effort.

The static bioeconomic model lends itself to a simple graphical illustration shown in Fig. 3. In an open-access fishery, entry will continue until an equilibrium is established at an effort level where TR=TC. Fishermen enter if their costs are covered even if their marginal contribution to the fishery is negative ($dTR < 0$, $dTC > 0$) and all rents to the resource are dissipated because this aspect of entry is not incorporated into their calculations of profit or loss.

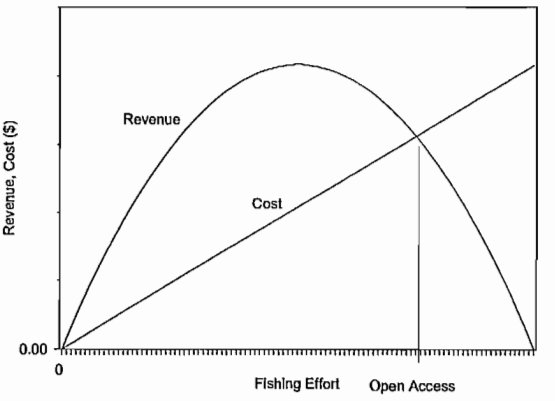


Fig. 3. Static bioeconomic model equilibrium. Open access equilibrium occurs at fishing effort levels where Total Revenue=Total Cost (rent dissipation).

Gordon's model was used as an explanation for the persistent problem of overfishing observed throughput the 1950's and 1960's. The basis for his analysis was the "common property", open-access nature of fisheries (Hardin 1968). Gordon's analysis led to the introduction of limited entry (or "closed access") programs to restrict fishing effort in the late 1960's and 1970's. The properties of the static model have been discussed extensively (Anderson 1977; Cunningham, Dunn and Whitmarsh 1985).

Empirical Applications of the Static Gordon-Schaefer Model

The static model has been most often used by economists to estimate how current effort in a fishery compares to the economically optimal fishery where, for profit maximization, marginal cost is equal to marginal revenue.

Hannesson (1989) applied the Gordon-Schaefer model to the inshore fishery of Cyprus to determine how current levels of catch and effort compared to the economically optimal level for the fishery. Staniford (1988) used the model to quantify how supply and demand conditions impacted on the economic benefits of fisheries management in the rock lobster fishery in South Australia. Stollery (1986) used the static model to investigate empirically how per boat effort in terms of capital employed changed in response to a fixed total quota and shortened seasons in the halibut fishery. In these applications the Gordon-Schaefer model is used as a vehicle to study the optimal level of capital and labour in the industry.

Of particular interest to the issue of setting catch limits are the series of papers by Copes and Cook (1982) and Cook and Copes (1987) on the Pacific halibut fishery. These authors use the static model to compare existing harvest levels to predicted harvest equilibria of the model. Cook and Copes (1987) found that average actual effort in the fishery was 158,600 skate-soaks with catches amounting to 13.7 million pounds per period in an open access equilibrium over the 1968-1980 time period. The model predicted optimal catch was calculated as 9.75 million pounds associated with effort

of 95,200 skate-soaks per period. The open access fishery was calculated to cost society about $1.7 million compared to the model optimal catch.

Despite the empirically verifiable predictions of the Gordon-Schaefer model, the prescriptions for avoiding economic and biological overfishing were less than successful. Serious depletion and even collapse of some fish stocks, and economic difficulties for fishing fleets and processing companies continued in spite of limited entry programs. This is attributed to the inability to limit entry before fleets became severely over-capitalized (and the failure of buy-back programs). In fact, many limited entry fleets had an incentive to continue to invest in harvesting capacity, known as "capital stuffing", because the fishery was transformed into an open access fishery with a fixed number of participants and the "race for fish" and profit remained.

Dynamic Bioeconomic Models

Fisheries economics is a branch of capital theory in economics. Scott (1955) was the first to treat the problem of the management of the fishery in capital theoretic terms. Modern treatments of the subject include Clark and Munro (1975) and Clark (1976a, 1985) and Swierzbinski (1981). The basic model starts with the growth equation for the biomass as:

$$dX/dt = G[X(t)] \tag{1}$$

where $G[X(t)]$ is the continuous growth rate function of the current stock biomass, $X(t)$ at time t. It is assumed that $dG/dX > 0$ for $0 < X < K$ and that $dG(0)/dt = dG(K)/dt = 0$ where K is the environmental carrying capacity of the stock and $d^2G/dX^2 < 0$..

The Schaefer aggregate stock logistic model has dominated as the specification of $G(X)$ in the modelling of population dynamics of fish stocks. The Schaefer model is given by:

$$G(X) = rX(1 - X/K) \tag{2}$$

where r is the intrinsic growth rate of the resource. The Schaefer growth function is illustrated in Fig. 1. (Generalized forms of the logistic function are discussed in detail in Clark 1976a.)

When harvesting is introduced Equation (1) is modified to become:

$$dX/dt = G[X(t)] - H(t) \tag{3}$$

where $H(t)$ is the harvest rate at time t. The sustained yield from the resource occurs where $dX/dt = 0$ and $H(t) = G[X(t)]$ for all $X(t)$ in the domain from 0 to K. The equilibrium sustained yield curve is plotted in Fig. 2 as for the static model.

The harvest production function is typically specified as:

$$H(t) = qE(t)X(t) \tag{4}$$

where $E(t)$ denotes the measure of fishing effort at time t and q is a technological constant called the catchability coefficient. Without loss of generality it is assumed that q is normalized to 1. Furthermore, it is assumed that landed value prices are exogenous and constant over time and that the supply of fishing effort is perfectly elastic. Accordingly, the total cost of fishing effort may be given by:

$$\mathrm{TC}[E(t)] = cE(t) \tag{5}$$

where c is constant. Combining (4) and (5) and expressing TC as a function of both harvest rate and biomass, then

$$\mathrm{TC}(H, X) = cH/(qX) \tag{6}$$

Finally, the net revenue in period t from harvesting can be expressed as:

$$\Pi(t) = [p - c/(qX)]H \tag{7}$$

The goal of management in this framework is to maximize the net present value of the net revenue flow, $\Pi(t)$ over an infinite time horizon. The total present value of $\Pi(t)$ over time is $\int \Pi(t)e^{-\delta t}\,\mathrm{dt}$. The maximization of $\Pi(t)$ is subject to constraints on nonnegative stock size, $X \geq 0$ and capital limits on harvest rates, $0 \leq H(t) \leq H_{\max}$, where $H_{\max}$ is the upper limit on harvesting capacity. It is also assumed that $H(t) \leq X(t)$ and that the initial biomass $X(0)$ is known. Formally, the dynamic optimal control problem is:

$$\text{Max} \int_0^\infty \exp(-\delta t)\,[p - c/(qX(t))]\,H(t)dt \tag{8}$$

subject to:

$$\begin{aligned} dX/dt &= G[X(t)] - H(t) \\ 0 \leq H(t) &\leq H_{\max} \\ X(0) &\text{ is known} \end{aligned}$$

This problem can be solved using the maximum principle or through Lagrange multipliers. Using the control theory methods, the Hamiltonian for this problem is:

$$J = e^{-\delta t}\,[p - c/(qX)]\,H(t) + \lambda(t)\,\{G[X(t)] - H(t)\} \tag{9}$$

where $\lambda(t)$, the adjoint variable, is the shadow price of the resource at time t. (See also Clark 1976a, 1985.)

From the maximum principle the optimal biomass level is determined from the necessary condition for optimality:

$$\frac{(dG/dX^*)\,[p - c/(qX^*)]\,G(X^*)}{p - c/(qX^*)} = \delta \tag{10}$$

The condition (10) is known as the "Golden Rule". It states that the own rate of interest of the resource should be equated to the opportunity cost rate of interest. The denominator $p - c/(qX)$ is the sacrifice required in terms of current

net revenues to make an investment in the resource while the numerator is the marginal sustainable resource rent.

The optimal decision rule is to invest (i.e., allow the stock to grow at a maximum rate) or disinvest (i.e., harvest the stock at a maximum rate) in the resource until the own rate of interest is equal to the opportunity cost rate of discount. At that point of equilibrium, the stock and annual catch is sustainable (Clark 1976).

By substituting (2) in for X in (10) we can solve explicitly for the optimal sustainable stock size X^* from:

$$X^* = 0.25\left[\frac{c}{(pq)} + K\left(1 - \frac{\delta}{r}\right) + \sqrt{\left[\frac{c}{(pq)} + K\left(1 - \frac{\delta}{r}\right)\right]^2 + 8K\left[\frac{c\delta}{(pqr)}\right]}\right] \quad (11)$$

The optimal harvest policy is to set $H^*(t) = G[X^*(t)]$. The specific harvest rate will depend on q and p/c and δ. Special cases of interest (Clark 1985, p. 24) are the equilibrium positions that arise (i) to give "maximum economic yield" or MEY under no discounting ($\delta = 0$), (ii) under no costs ($c = 0$), and (iii) under static "bionomic" (Gordon 1954) assumptions, i.e., no discounting and zero rents. The equilibrium stock size biomass for these special cases are given by:

$$(12)\quad \delta = 0, \qquad X_{\text{MEY}} = 1/2([c/(pq)] + K) > X_{\text{MSY}}$$

$$(13)\quad c = 0, \qquad X_0 = (K/2)[1 - (\delta/r)] < X_{\text{MSY}}$$

$$(14)\quad \delta = 0, \qquad \text{and TC=TR, } X_d = c/(pq)$$

which illustrate that cost effects alone ($\delta = 0$) move the maximum economic yield (MEY) equilibrium stock size, X^* to the right of the maximum sustainable yield stock size, X_{MSY} and discounting alone ($c = 0$) moves X^* to the left. Of particular interest is the static equivalent case where $\delta = 0$ and economic rents are dissipated so that $\text{TR} = pH = \text{TC} = cH/(qX)$ which yields $X_d = c/(pq)$ as in (14). Depending on the value of the c/p ratio, the static equilibrium stock size may occur anywhere in the domain of X between 0 and K. Clark (1985) shows that $X_d < X^* < X_{\text{MEY}}$, i.e., X^* lies between the bionomic equilibrium and the static rent maximizing optimum, X_{MEY}.

The results of the equilibrium harvest levels, H^* follow directly from the results in (11) to (14). For $\delta = 0$, H_{MEY} does not exceed H_{MSY} since the stock size at MSY exceeds that at MEY. Similarly, for $c = 0$, H^* exceeds H_{MSY}. For the static Gordon-Schaefer model annual yields (or catch limits) may vary throughout the range 0 to $X(t)$ depending on the ratio of c to p in the function $H = rc/p[1 - c/(pqk)]$.

A number of significant research studies have extended the basic dynamic model to include stochastic changes in the system, and to include simultaneous control on more than one decision variable. Clark et al. (1979) use optimal control theory to determine simultaneously resource stock and fishing capital stock dynamics to maximize long run economic benefits. Charles and Munro (1985) extend this model to cover stochastic aspects of the problem. Clark et al. (1985) use similar methodology to plan the optimal expansion of developing fisheries. Charles (1987) provides a review of control methods in fisheries management.

Perturbation analysis is used by Ludwig (1979) to determine optimal fishing effort levels for continuous, stochastically fluctuating stocks. He shows that optimal policies are "bang-bang" controls: fishing effort is maximal if the stock exceeds the optimal stock size; if the stock is below the optimal level effort is minimal. These results compare directly to those found under expected value or certainty-equivalent assumptions.

Empirical Application of the Dynamic Model

The empirical applications of the dynamic model are dominated by studies that measure the actual stock of biological capital and harvesting capacity relative to the model predictions. The optimal equilibrium harvest size (for setting the TAC) is a by-product of this analysis. The nature of the time path solution to the equilibrium position in the dynamic model is the most rapid approach path to equilibrium, i.e., if the stock is observed to be below its equilibrium level then cease harvesting; if the stock is above equilibrium levels, then fish at the sustainable rate . This characteristic of the "bang-bang" rule makes direct operational application of the optimal dynamic policies untenable for most established commercial fisheries that have not reached their long-run potential.

Whale fisheries have provided a rich application area for the dynamic model. The relatively slow growth of these stocks and the intense fishing effort and capital directed at these animals necessitates a long term strategic and dynamic outlook. A number of deterministic dynamic model applications to whale fisheries are presented below.

The dynamic model was first applied in fisheries by Spence (1974) to examine the plight of the Antarctic blue whale. The study explored optimum harvest levels for blue whales under a range of parameter assumptions. Clark (1976b) applied the model to calculate optimal escapement for Antarctic fin whales at different assumed discount rates. He also established that the continuous-time non-delay logistic model was a good approximation for practical purposes of whale management to the delay difference population model. Clark and Lamberson (1982) use the basic dynamic model to calculate the "bionomic" equilibrium yield (i.e., the static model, dissipated rents solution) for the Antarctic whaling industry as a yield of 2750 blue whale units compared to an estimated maximum sustainable yield of at least 5000 blue whale units. Conrad (1989) calculates optimal dynamic harvest levels for western Arctic bowhead whale. Conrad and Bjorndal (1991) analyze the case of Northeast Atlantic minke whale and conclude that the optimal stock ranges from 51,000–59,000 units supporting a harvest of 1600–1750 whales by 90–115 catchers. Palsson (1991) used the dynamic model and a delayed logistic growth function to calculate the sole owner

optimum and annual harvest removals for the East Greenland-Iceland fin whale stock. His results were supportive of the Icelandic claim that stocks were healthy and could support a limited and economically viable commercial fishery.

Wilen (1976) applied the deterministic version of the dynamic model proposed by Smith (1969) to the Pacific fur seal. This study was concerned principally with open access investment dynamics and not with setting TAC's. Wilen calculated MSY as 79,000–89,000 animals as a check on his empirical estimation of the biological growth equation. However, the methodology used can determine rent maximizing TAC's.

Andrew and Butterworth (1987) note that $F_{0.1}$ as a strategy attempts to induce stock recovery by keeping annual catches below current sustainable yields while building up biomass to a level higher than the MSY level. They use a discrete version of the bioeconomic model to examine whether the $F_{0.1}$ biomass is the appropriate level at which to stabilize hake stocks (assuming that the lowest recorded catch rate in each hake Division corresponded to zero economic rent). The discount rates for which $F_{0.1}$ is optimal were found to range from 10–81% for the four Divisions. By assuming that 5% is a more realistic discount rate, optimal values under the assumed price/cost ratio and stock dynamics were found to range from $F_{0.15}$ to $F_{0.30}$. Accordingly the authors suggest the hake stock should be larger than the steady state $F_{0.1}$ stock size.

While the method of Andrew and Butterworth is noteworthy, their interpretation of discount rates do not distinguish between private and social discount rates. The high industry discount rates they found to be associated with $F_{0.1}$ in different areas are low on theoretical grounds for a fishing industry competing on an open access basis for a global TAC. Firms are expected to use an infinite discount rate since they have no assured access to future profits. The absence of property rights causes a large divergence between the private and the social rate of discount and it is inappropriate to apply a social discount rate of 5% to the industry to calculate the optimal $F_{0.m}$.

Bjorndal (1988) calculated deterministic optimal stock levels and harvest rates for North Sea herring. Difficulties in modelling the schooling behaviour of herring are avoided through the assumption that costs are not density dependent. Bjorndal concludes that stock extinction is optimal under the dynamic model at a discount rate of 52% whereas the stock would be kept at MSY if the discount rate was zero.

Conrad (1990) applies the "approximately optimal" technique proposed by Burt and Cummings (1977) to simulate adaptive management of the Pacific whiting fishery. Conrad's starting point is that while stochastic optimization results are useful for describing the long-run effects of uncertainty in the bioeconomic parameters, the distributional form of the results does not provide a prescription for short-term management. Evidence of this difficulty is notable by the absence of applied stochastic analyses in the fisheries literature. Decisions in the short run would require that the optimal "approach path" to the long-run equilibrium (found for example from the deterministic certainty-equivalent problem or the finite-horizon stochastic dynamic programming problem) be determined. Since the stock production function $F[X(t), E(t)]$ is likely nonlinear this is a technically difficult problem to solve. Conrad notes that the first order condition (10) can be regarded as defining a relationship between stock, $X(t)$, and effort, $E(t)$, in the vicinity of long-run equilibrium. The assumption of being in the neighbourhood of the equilibrium and condition (10) are used to approximate the optimal short run approach path. Estimates are obtained directly for the coming year to set $E(t)$ and TAC's. For Pacific whiting, Conrad determined that the current stock is within 30% of the long-run equilibrium and therefore the approximately optimal decision procedure can be applied to the fishery. Similarly, in an application to the anchovy fishery in northern California, Kolberg (1990) compared the approximation procedure to the optimal solution obtained via dynamic programming and found that they result in harvesting decisions within 1% of the maximum.

Within the assumptions of the models, the approximate optimal methods circumvent the difficulties of stochastic bioeconomic analyses and make it technically feasible for dynamic bioeconomic considerations goals to be taken into account when setting annual TAC's.

Mathematical Programming Models

Integrated TAC decision problems can be formulated explicitly using operational research-based methods including mathematical programming techniques (Lane 1989a and Williams 1989). These methods are generally designed for specific problems or applications. They provide a great deal of insight and information about alternative options to meet specific goals and offer a efficient alternative strategy development and evaluation framework.

In this approach, biological considerations are formulated as part of the constraint set on feasible TAC decision variables. Socioeconomic considerations are accounted for explicitly in the objective functional and may take any wealth-related form, e.g., profit maximization, consumer surplus, discounted net operating income, etc. Static and dynamic formulations are presented typically in deterministic settings. Stochastic considerations are examined using simulation modelling approaches in tandem with expected "optimal" results from the deterministic analyses.

In general, the mathematical programming problem can be formulated as follows:

$$\max_{F(t)} \sum_{t,i} [p_i(t) - c_i(X,t)]\, H[X(t), F(t)] \tag{15}$$

subject to,

$$\sum_{A_j} b_i X_i(t) \geq B_j(t) \tag{16}$$

$$0 < F(t) < r \tag{17}$$

This formulation is comprised of three major components: (i) the decision variable set, e.g., the schedule of fishing mortality values, F(t), over the discrete time planning period; (ii) the single-valued objective functional (15) that incorporates value-related components of the problem, e.g., landed value prices, p at age i and period t, and unit harvesting costs of fish at age i in time t; and (iii) the system of constraints, j that quantify biological targets over time (16) and for age groupings A_j of the stock, and impose bounds on the decision variables (17) each period.

Depending on the population dynamics model relating stock size, $X(t)$ changes over time as a function of the decision variables, $F(t)$, the mathematical programming problem will typically be nonlinear in the objective function and the constraint set. Solution procedures are computationally tractable using numerical methods for most deterministic data problems of medium to long term duration (5 to 20 years). The key differences in this approach versus fixed fishing mortality or equilibrium search methods is that it incorporates known elements of the current status of all data and variable values in a search over a feasible domain of the decision variables. Depending on updated information, the current and strategic views of the desired stock size and harvesting limits may change. Empirical examples of these alternative strategy development models are discussed below.

Empirical Applications of Mathematical Models

Rothschild and Balsiger (1971) constructed a linear-programming model to allocate the catch of salmon among the days of the salmon run. They considered the possibility of increasing the landed value of the catch if one knows in advance the structure of the run by optimally allocating the catch among the days of the run. Formally, the objective of the model was to derive a management schedule for catching salmon that maximized the landed value given daily constraints on processing capacity and escapement of salmon of both sexes. The dual problem considered the shadow prices of the various sizes of fish, eggs, and cannery capacity thus enabling the manager to view his decisions in light of the marginal values of these entities. The model was applied to the run of sockeye salmon in the Bristol Bay system in 1960.

In a critical analysis of decision making with respect to identifying representative population dynamics models for stocks, Ludwig and Walters (1982) used dynamic programming, statistical analysis, and simulation to demonstrate the significant impacts of unreliable catch and effort data on derived "optimal catch levels with data taken from the Pacific coast salmon fishery. Their results demonstrated that stock probing and experimental management regimes provide valuable information in determining actual stock dynamic behaviour.

Lane and Kaufmann (1993) presented a mathematical programming framework as in (15)–(17) for the determination of TAC strategies for the 2J3KL northern cod fishery. The framework included the development and bioeconomic evaluation of alternative dynamic TAC adjustment strategies to reach prespecified stock targets over time. Their results showed that feasible alternative strategies may exist.

Table 1. Estimated parameter set and values for the 2J3KL Northern cod TAC schedule approaches. Data values for 1991 are approximate actual values only.

Parameter	Description	Value/Units
p	1991 price per unit catch by weight	\$0.50/kg
c	Unit cost of effort per kg times 3+ biomass	\$460,000 t/kg
c_1	1991 variable cost of fishing	\$0.37/kg
c_2	1991 total cost per vessel per year	\$250,000
δ	Continuous annual discount rate	10%
M	Continuous annual rate of natural mortality	0.20
K_1	von Bertalanffy growth term	0.15
K_2	Stock carrying capacity ages 3+	1,600,000 t
r	Year-over-year intrinsic growth rate	0.80
R	Constant annual recruitment at ages 3	300,000 t
α	Harvest catchability factor	0.00078
b	Cost adjustment factor for year over year TAC decreases greater than 25,000 t	\$6.00/(000's t)^2

Comparative Analysis

This section compares bioeconomic measures for annual TAC strategies developed from five different models discussed above. Each are applied to current (start of year 1991) data for the 2J3KL northern cod fishery. Comparisons are made on the deterministic versions of the different strategies. The five approaches include:

1. the constant annual fishing mortality strategy, $F_{0.1}$, where $F_{0.1} = 0.25$ for 2J3KL Northern cod (Gulland 1983);
2. the 50% adjustment rule to the $F_{0.1}$ fishing mortality strategy (Rivard and Maguire 1993);
3. the maximum economic yield strategy based on the Golden Rule (10) (Clark 1976a);
4. the mathematical programming model based on (15–17) (Lane and Kaufmann 1993); see also Appendix A; and
5. the approximate optimal strategy based on annual re-estimation of (10) (Conrad 1990).

The annual TAC schedules for northern cod are derived for each of the five strategies over a 10 year time horizon beginning in 1991 and ending in the year 2000 according to the deterministic bioeconomic model described in Appendix B. Data for the initial year (1991) status of the 2J3KL northern cod stock and projected annual stock-recruitment and growth

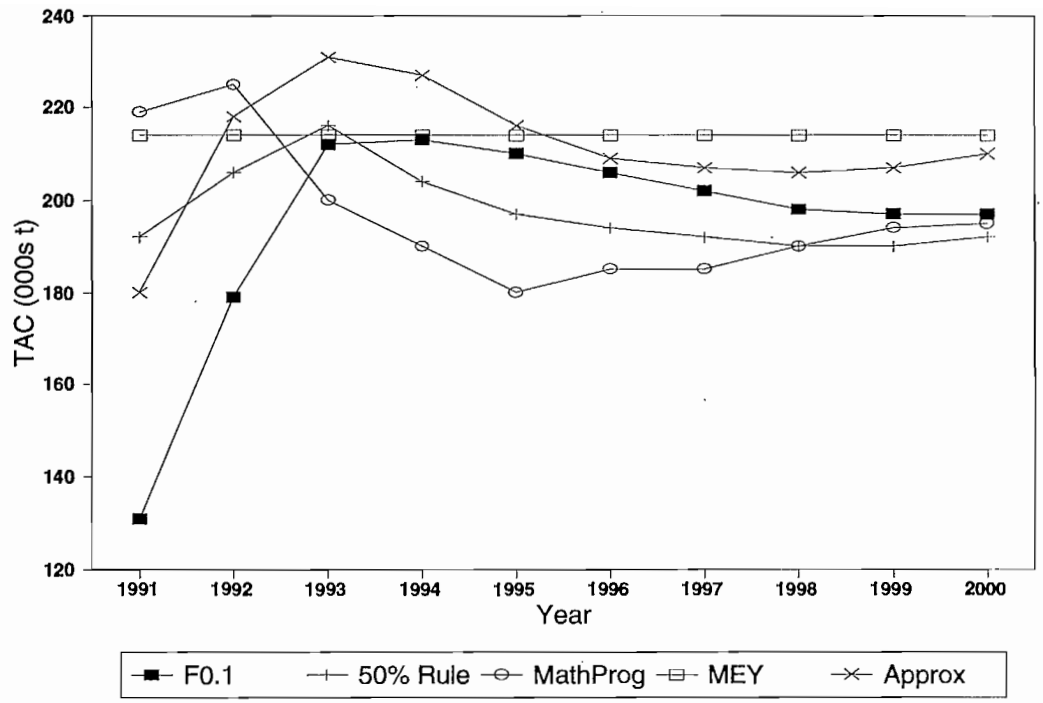


Fig. 4. Alternative annual TAC schedules for 2J3KL Northern cod for 1991 to 2000.

Table 2. Required parameter sets by TAC schedule strategy.

TAC Schedule	Parameters Required
1. F	M, K_1
2. 50% Rule	M, K_1
3. Maximum Economic Yield	$p, c, \delta = 0, K_2, (r, R)$
4. Math Programming	$p, c_1, \delta, K_1, K_2, M, R$
5. Approximate Optimal	$p, c_2, \delta, r, K2, M, R, \alpha$

Table 3. Alternative TAC schedules for 2J3KL Northern cod.

Year	$F_{0.1}$	50% Rule	MathProg	MEY	Approximate
1991	131	192	219	214	180
1992	179	206	225	214	218
1993	212	216	200	214	231
1994	213	204	190	214	227
1995	210	197	180	214	216
1996	206	194	185	214	209
1997	202	192	185	214	207
1998	198	190	190	214	206
1999	197	190	194	214	207
2000	197	192	195	214	210
TOTAL	1945	1973	1963	2140	2111

used for each strategy are provided in Appendix C. A summary of the northern cod parameter set and estimated data values used in all TAC strategies development are provided in Table 1. Table 2 presents the table of parameter sets required for each of the five TAC strategies.

The 1991–2000 annual TAC strategies for Northern cod were projected using the current (start of year 1991) age structured assessment, partial recruitment at age function, and average weight at age for cod (Appendix C). The TAC schedules for each approach are presented in Table 3 and Fig. 4.

Summary biological (stock), and economic output measures for each approach are calculated and compared. Summary measures include: (i) the estimated size of the harvestable cod stock biomass (ages 3 and over) at the end of the planning period (year 2000), and (ii) the adult cod stock biomass (ages 7 and older) at year 2000, and (iii) the estimated total discounted (at 10% annually to 1991) net revenue from harvesting. Annual net revenue is calculated from annual landed value, $p\text{TAC}(t)$ less total annual operating costs of fishing effort as a function of harvestable (age 3 and older cod) biomass, $c\text{TAC}(t)/X(3+, t)$ over the planning period 1991 to 2000. Adjustment costs are also calculated when the year over year change in the TAC for any strategy schedule decreases by more than 25,000 t of cod. Adjustment costs are calculated as $b = 6$ times the square of TAC decrease measured in thousands of metric tonnes (i.e., a decrease of 30,000 t in TAC from one year to the next results in an adjustment cost to net revenue for the year of $6 \times (30 \times 30) = 5400$ thousand dollars). Table 4 presents the biological and economic output measures for start of year 2000 spawning stock and the discounted total net revenue to 1991. Percentage differences

Table 4. Summary stock and value output measures for the alternative TAC schedules. Discounted Net Revenue values are net of adjustment (large negative TAC decreases) costs.

TAC Schedule	Year 2000 3+ Biomass (000's t)	Year 2000 7+ Biomass (000's t)	Discounted Net Revenue[1] ($M 1991)	Diff. Disc. versus F (%)
$F_{0.1}$	1314	704	142.9	—
50% Rule	1274	665	147.9	+3.50
MEY	1123	526	135.7	−5.04
Math Prog	1298	682	143.1	+0.14
Approx.	1152	551	142.5	−0.28

[1] Discounted Net Revenue results are net of adjustment costs.

in the discounted net revenue measures are relative to the $F_{0.1}$ strategy results.

The results of Table 4 show marked similarity for the summary values for the economic output, discounted net revenue after adjustment costs. Percentage differences vary only between −5 and +4 percentage points. However, what is not taken into account in either the 50% rule strategy, the MEY strategy, or the approximate optimal strategy is the exogenous biological target for ages 3+ cod which requires the 3+ biomass to grow to 1.3 million t by the year 2000. It is significant that only the $F_{0.1}$ strategy (1.32 million t) and (for all intents and purposes) the mathematical programming strategy (1.298 million t) — that accounts explicitly for this biological target — reach the desired goal. The other strategies are not flexible enough to take account of such a constraint. Moreover, among the two "feasible" strategies with respect to the biological target for 3+ biomass, the mathematical programming strategy shows a small 0.14% improvement in the economic performance over the $F_{0.1}$ schedule. As well, the characteristic form of the mathematical programming solution differs significantly from the monotonically declining $F_{0.1}$ (and 50% rule) reference point approach.

It is most important to note from this simple comparative analysis that alternative TAC adjustment strategies (with quite different schedule characteristics) may accomplish the same desirable, quantified targets over time. Moreover, the objective of scientific strategy development and evaluation is to examine alternative options toward making informed and effective policy decisions. It is perhaps unfortunate that history and entrenched approaches to fisheries management problems have tended, in our view, to narrow rather than expand the set of available management policy options.

Conclusions

This review of alternative bioeconomic approaches for determining TAC's was motivated by a perceived lack of integrated biological and economic considerations in the existing operational solutions to this problem. One goal of this study was initially to examine strong conceptual bioeconomic approaches to this problem of setting TAC's. Among these, a new methodology could emerge to meet the challenge of more effective and strategic management policy-making for the fishery.

It is informative to note that much of the existing theory, while credible from all formal viewpoints, is often intractable from an operational perspective. For example, the characteristic solution of the dynamic models as "bang-bang" adjustment may not be acceptable in practice due to the high social cost such as are being incurred under the Northern cod moratorium. What is missing from these modelling approaches is an explicit handling of the short-term adjustment problem. To accommodate social and economic adjustment, the costs of high variability in year-over-year TAC's must be directly incorporated.

The inevitable practice of continually adjusting to changing conditions in the fishery demands the clear articulation of long-term objectives. Only with these bioeconomic objectives in hand can an appropriate short and medium term development of policy occur. The traditional sole involvement of "science" in the TAC setting decision and the vacancy left by lack of clear operational bioeconomic policy underscores the difficulties of the problem. What is required is a fundamental paradigm shift away from owned independent analyses (and "objectives") by biology and economics and operations. One area of hope in this impasse may come from the problem framework offered by the scientific method of problem solving from the field of operations research/management science. In particular, the solutions offered by mathematical programming whereby biological and other operational conditions can be embodied as system constraints, and economic value considerations are explicitly optimized, provides both a framework for the development and bioeconomic evaluation of short term and long-run decision alternatives. Moreover, in this flexible modelling environment, other types of constraints (such as TAC stability over time) and multiple objectives can be incorporated. Finally, this approach emphasizes the role of systems in support of fisheries management problem solving in the development and evaluation of decision alternatives. The modelling exercise is a springboard for investigating system sensitivity to parameter value changes and random fluctuations inherent in the evaluation and management of risk in fisheries.

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Appendix A

Description of Mathematical Programming Method

This appendix describes the detailed methodology for the mathematical programming model (Lane and Kaufmann 1993) to develop alternative TAC schedules. The decision framework is defined to include annual TAC's or corresponding fishing mortality levels as the decision variables, strategic biological considerations incorporated as constraints, and economic value assigned as the objective of a dynamic mathematical programming problem.

Population Dynamics

The model uses initial stock data including numbers at age, average weight at age, partial recruitment, natural mortality and fishing mortality data (from the corresponding TAC schedule) to determine year-over-year stock at age distributions for (i) numbers of fish, (ii) stock biomass, and (iii) catch (weight and numbers). Model details are provided below.

The population dynamics model is based on the following system of equations:

$$\text{(A1)} \qquad N_{0,t} = R_t = c\alpha S_{t-1}\,(1 - \gamma\beta S_{t-1})^{1/\gamma}$$

$$\text{(A2)} \qquad N_{a,t} = N_{a-1,t-1}\exp(-M - \mathrm{PR}_a F_t)$$

$$\text{(A3)}\ C_{a,t} = w_a N_{a,t}\,(1 - \exp[-M - \mathrm{PR}_a F_t])\,\frac{\mathrm{PR}_a F_t}{M + \mathrm{PR}_a F_t}$$

for $t = 0, 1, 2, \ldots, T$, and $a = 0, 1, 2, \ldots, A$, where $t = 0$ denotes the initial (current period), and $t = T$ the final year of the planning horizon, and $a = 0$ denotes births and $a = A$ the oldest age of fish in the stock.

Equation (A1) represents the deterministic version of a generalized stock-recruitment function used to generate the yearly reproductive capacity of the stock. $N_{0,t}$ denotes the numbers of births at the beginning of each year t as a function of the spawning stock biomass (ages 7 and older for Northern cod, $S_t = \sum_{a=7}^{A} N_{a,t}$) and the scale, productivity, and shape parameters α, β, γ, respectively of the stock-recruitment function. The constant multiplicative factor c adjusts the empirical at age "recruitment to the fishery" numbers back to age 0 by adjusting for annual natural mortality, M, to the unrecruited stock, e.g., for recruitment to the fishery initially at age 3, then $c = \exp(+3M)$.

Equation (A2) defines the year-over-year changes in each cohort of the stock. The changes in the numbers of stock is a function of M, the continuous rate of natural mortality (assumed as elsewhere to be constant across all ages in the population), the partial recruitment at age, PR_a (constant over all years of the planning period), and the fishing mortality level, F_t for fully recruited ages of the stock. Using (A2) and the expected weight at age (at the beginning of each year, and on average over each year), corresponding biomass levels for different age groups can be determined.

Equation (A3) defines the catch weight at age, $C_{a,t}$ for a given level of fishing mortality by year. The average weight per fish at age a, w_a is determined over the entire yearly catch period. The adjusted weight at age for cod at the beginning of each year, w_a^b is approximated by the midpoint between successive age's average weights, i.e., $w_a^b = (w_{a-1} + w_a)/2$. This equation also provides the basis for determining (i) catch

numbers at age in a given year, (ii) the total catch (TAC) resulting from fishing mortality, F, and (iii) the average weight per cod at capture.

The TAC Schedule

A formal model to determine the upper limit on total harvest over the planning period while simultaneously satisfying all biological constraints was developed using mathematical programming. A simple form of the objective function of the optimization problem is to maximize the discounted sum of annual TAC's over the planning period. This objective function implicitly incorporates resource allocation considerations explicitly on biological stock targets. The implicit assumption is that the harvesting sector as a whole is interested in higher total catch allowances (market and price effects notwithstanding). The deterministic optimization results provide insight into the characterization of the optimal, feasible solution. Formally, the objective functional for this nonlinear dynamic optimization problem may be written as follows:

$$\text{(A4)}\quad Z = \max_{F_t} \sum_{t=0}^{T-1} \exp(-\delta T) \sum_{a=1}^{A} C_{A,t}\,(F_0, F_1, \ldots, F_{T-1})$$

where $C_{a,t}$ is defined as in equation (A3) and updated accordingly through (A1) and (A2) making (A4) nonlinear in the decision variables, F_t, and δ is the continuous rate of discount of annual TAC's.

The objective function (A4) is constrained by biological constraints that take the form of biomass targets specified over the planning period. In anticipation of the cod problem, these constraints may be defined as follows:

$$\text{(A5)}\quad \sum_{a=3}^{A} w_a^b N_{a,t} \geq B_t\left(3^+\right),\ t \in \{0, \ldots, T\}$$

$$\text{(A6)}\quad \sum_{a=7}^{A} w_a^b N_{a,t} \geq B_t\left(7^+\right),\ t \in \{0, \ldots, T\}$$

$$\text{(A7)}\quad 0 \leq F_t \leq r,\ t = 0, 1, \ldots, T-1.$$

The left hand sides of constraints (A5) and (A6) are expressions for the biomass at ages three and older, and ages seven and older respectively at time t. The right hand sides of these constraints are the specific biomass targets for each time t. (For northern cod targets see Dunne (1990).) Since each $N_{a,t}$ is a nonlinear function of the fishing mortality, F, then these constraints are also nonlinear in the decision variables, F_t. Equation (A7) declares annual upper (r) and lower (0) bounds on the annual fishing mortality decision variables. Bounds on the decision variables are used to restrict optimal solutions to a set of "operational" solutions. For example, under particular circumstances model solutions may result in high levels of exploitation alternating with zero fishing, or a "pulse fishing optimal policy. In most cases this type of policy is not deemed to be operationally acceptable.

The solution to the nonlinear optimization problem is found using the GAMS MINOS software system running through DOS v5.1 on a 386 personal computer. GAMS MINOS is a nonlinear mathematical programming package that provides solutions to this type of problem. Details on the precise model formulation in the GAMS programming language, as well as details of the run time results are available from the authors.

Appendix B

Deterministic Bioeconomic Model

This appendix contains a description of input requirements for the complete description of the dynamic behaviour of a single, age-structured stock and inputs for describing the summary economic performance of the commercial fishery.

The above set of tables presents (i) the input parameters and (ii) the mathematical relationships built into the model. Each table is subdivided into stock dynamics model components and the summary economic model components.

Table B1. Description of Model Input Parameters.

Input	Parameter Name	Units of Measure	Description
I. Stock Biology			
1	N_0	No. of fish ('000's)	Total number of fish in the system (all ages) at start of study
2	p_a, $a = 1, \ldots, A_{\max}$, $\sum p_a = 1$, $0 \leq p_a \leq 1$	Fraction	Proportional distribution of fish at age at start of study
3	α, β, γ, c	Diverse	Stock-Recruitment parameters from Schnute (1985)
4	PR_a, $a = 1, \ldots, A_{\max}$ $0 \leq \mathrm{PR}_a \leq 1$	Fraction	Partial recruitment to fishery by age from total gear selectivity
5	M	Continuous rate	Natural fishing mortality
6	w_a	Kilograms (kg)	Average weight per fish at age a
7	$B_a(t)$	Tonnes (t)	Stock targets as a function of fish at age or by age groupings, e.g., 3+, 7+ biomass
8	F_t, $t = 1, \ldots, T$	Continuous rate	Total annual fishing mortality on fully recruited ages
II. Economic Inputs			
1	p	\$/kg	Landed value prices per tonne landed
2	$c(X)$	\$/kg	Total Operating Costs per tonne landed
3	b	\$/(000's t)^2	Adjustment cost factor due to large decrease in year-over year TAC > 25,000 t

Table B2. Detail of Model Equations.

Item	Relationship	Description
I. Stock Biology		
1	$N_{a,0} = N_0 p_a$, $a = 1, \ldots, A_{\max}$, $\sum_a p_a = 1$	Initial age distribution of the stock at the start of the planning period, $t = 0$ (in 000's of fish)
2	$N_{0,t} = c\alpha S_{t-1} (1 - \beta\gamma S_{t-1})^{1/\gamma}$	Annual stock-recruitment relationship (weight of spawning stock yields numbers of recruited fish) at start of season t
3	$X_{a,t} = \left(\frac{w_{a-1} + w_a}{2}\right) N_{a,t}$	Stock biomass at age (in 000's t) at start of season t
4	$N_{a,t} = N_{a-1,t-1} \exp\{-M - \mathrm{PR}_a F_t\}$, $t = 1, \ldots, T$	Year-over-year net growth (numbers of fish) at start of season t
5	$C_{a,t} = w_a N_{a,t} (1 - \exp[-M - \mathrm{PR}_a F_t]) \frac{\mathrm{PR}_a F_t}{M + \mathrm{PR}_a F_t}$	Catch at age (in 000's t) in season t
6	$\mathrm{TAC}_t = \sum_a C_{a,t}$	Annual total allowable catch, TAC (in 000's t) for season t
II. Summary Economic Measures		
1	$\mathrm{LV}_t = p\mathrm{TAC}_t$	Landed value (000's \$) in season t
2	$\mathrm{OE}_t = \frac{c\mathrm{TAC}_t}{x\,(3^+, t)} + b\,(\mathrm{TAC}_{t-1} - \mathrm{TAC}_t)^2$	Total Operating Costs in season t (000's \$) as a function of biomass and adjustment costs for $\mathrm{TAC}_{t-1} > \mathrm{TAC}_t$ +25,000 t
3	$\mathrm{NOR}_t = \mathrm{LV}_t - \mathrm{OE}_t$	Annual Net Operating Revenue in season t (000's \$)

Appendix C

Input Data Values for the 2J3KL Northern Cod Case

This appendix contains the input data values for the case of 2J3KL Northern cod. The data values are organized into five major components as presented below. Data source is the Groundfish Subcommittee Report of CAFSAC 91/9 (Anon 1991).

1. Stock Initial Age Distribution: Estimates as of start of year 1991 (i.e., January 1, 1991).
2. Partial Recruitment by Age: Partial recruitment (PR) parameters are provided in Table C1.
3. Natural Mortality: $M = 0.20$ for all scenarios, all years, and all cod cohorts.
4. Recruitment Levels. Estimated values for the "average" scenarios are set at 300 million age 3 fish per year. These recruitment levels are assumed to be fixed each year over the projection period 1991 to 2000, for each strategy being analysed.
5. Weights at Age. Estimated weights at age (in kilograms per piece) for 2J3KL cod (Table C1).

Table C1. 1991 Stock Distribution, Average Weights and Partial Recruitment by Age for Northern cod.

Age	Numbers (millions cod)	Avg. Weight at Age (kg)	Partial Recruitment
3	300	0.48	0.007
4	526	0.73	0.158
5	312	1.06	0.472
6	122	1.43	0.965
7	34.4	1.80	1.00
8	23.7	2.16	1.00
9	25.0	2.53	1.00
10	12.6	3.05	0.976
11	4.4	3.98	0.760
12	2.3	5.22	0.654
13	1.3	6.88	0.50
14	0.4	8.64	0.50
15	0.1	10.06	0.50
16	0.0	10.63	0.50

Management Strategies: Fixed or Variable Catch Quotas

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Steinshamn, S. I. 1993. Management strategies: Fixed or variable catch quotas. p. 373–385. *In* S. J. Smith, J. J. Hunt and D. Rivard [ed.] Risk evaluation and biological reference points for fisheries management. Can. Spec. Publ. Fish. Aquat. Sci. 120.

The objective of this paper is to compare two different management strategies for a long-lived fish stock with fluctuating or stochastic recruitment; namely constant catch quotas (TAC's) vs. variable catch quotas based on a constant level of fishing effort. The analysis is based on a relatively simple aggregated biomass model. The first part of the paper considers some theoretical aspects of the problem and identifies expected outcomes under different assumptions about the economic relationships in the model. The second part discusses the implications of uncertainty and random fluctuations affecting recruitment. This part is supplemented by Monte-Carlo simulations to test the significance of the suggestions made in the theoretical part of the paper and to detect the importance of stock-dependent biological relationships that do not easily lend themselves to theoretical investigation.

La présente étude a pour objet de comparer deux stratégies différentes de gestion pour un stock de poissons à longue durée de vie et à recrutement fluctuant ou stochastique, et plus précisément les contingents de capture constants (TPA) par opposition aux contingents variables, sur la base d'un niveau constant de l'effort de pêche. L'analyse se fonde sur un modèle relativement simple de biomasse globale. La première partie de l'étude envisage certains aspects théoriques du problème et repère les résultats prévus selon diverses hypothèses sur les relations économiques dans le modèle. La deuxième partie analyse les incidences de l'incertitude et des fluctuations aléatoires qui affectent le recrutement. Cette partie est complétée par des simulations de Monte-Carlo qui testent la signification des suggestions faites dans la partie théorique de l'étude et qui repèrent l'importance des relations biologiques dépendantes du stock qui ne se prêtent pas facilement à l'étude théorique.

In a fishery where the biomass is fluctuating from year to year it often seems inevitable that annual catch, fishing effort and net income from the fishery will vary and cause undesired effects for fishermen, the processing industry and the marketing sector. It is, however, possible to keep one of these variables constant and let the other two vary. In this paper particular interest is put on two cases: 1) fixed catch quotas and 2) variable catch quotas based on a fixed level of fishing effort.

Fixed catch quotas will stabilize fishermen's income, and this may be a major priority since this is an occupation that is heavily affected by uncertainty. It may also help to secure stable supply to the processing sector and make marketing easier since a stable supply to foreign markets can be guaranteed.

It is also possible to list economic as well as biological arguments in favour of constant effort. This strategy may lead to a more stable utilization of labour and capital equipment in the harvesting sector, but at the cost of a more unstable income. There will also be a trade-off between the harvesting and the processing sector regarding stabilizing the use of labour and capital. Another argument is that constant effort may be the more profitable strategy in many cases, and an objective here is to see in which cases this is correct; see also Hannesson and Steinshamn (1991). An important biological argument in favour of constant effort is that it exerts a stabilizing effect upon the stock.

Constant effort or constant catch?

Some results from a two-period model

Results from a simple two-period model may illustrate some interesting aspects of the problem considered here, and may indicate the need to develop a more realistic and complex model in order to pursue the questions that arise further. These results particularly emphasize the effects of overlapping generations.

Assume first that the production function in the fishery is the widely used

$$u_t = q E_t B_t \tag{1}$$

where u is harvest, E is effort and B is biomass, all at time t, and q is the catchability coefficient. Since effort is here measured in arbitrary units, it is possible to define $q = 1$ and adjust the units of effort accordingly. Denote the price per unit harvest p, the cost per unit effort c. Then the net revenue function can be written either

$$\pi\,(u_t, B_t) = (p - c/B_t)\,u_t \tag{2a}$$

or

$$\pi\,(E_t, B_t) = (pB_t - c)\,E_t \tag{2b}$$

depending on whether u or E is considered to be the control variable. Assume also a survival parameter s, and a recruitment w, such that the dynamics in the model can be written either

$$B_{t+1} = s\,(B_t - u_t) + w_t \tag{3a}$$

or

$$B_{t+1} = s\,(1 - E_t)\,B_t + w_t. \tag{3b}$$

First the importance of the survival parameter will be considered, that is whether there are overlapping generations or not. If $s = 0$, then

$$B_{t+1} = w_t \tag{4}$$

and there is no dependence between B_{t+1} and B_t. An optimal policy will be to apply maximal effort (or maximal harvest) if $B_t > c/p$ and zero effort (or harvest) if $B_t < c/p$[1].

It is easily seen that any constant effort policy will yield a better result in terms of net revenue than the constant catch policy with the same average harvest. Denote, e.g., the biomass in a two-period model B_1 and B_2 with an average $\overline{B}$. The average harvest with a constant effort E, is

$$\bar{u} = E\overline{B}. \tag{5}$$

In order to take $\bar{u}$ as a constant harvest, an average effort level $\overline{E}$ is needed, which is given by

$$\overline{E} = \frac{E\overline{B}}{2}\left(\frac{1}{B_1} + \frac{1}{B_2}\right) \tag{6}$$

which, as the reader may easily verify, is always greater than E unless $B_1 = B_2$. This holds true also for most of the other production functions commonly applied in fisheries analysis, e.g.,

$$u = E^{\alpha} B^{\beta} \tag{7a}$$

or

$$u = B\left(1 - e^{-E}\right). \tag{7b}$$

A sufficient condition for this to hold true is that catch per unit effort (u/E) is an increasing function of B. The argument may be extended to a similar model with an arbitrary number of periods. Figure 1 illustrates this in a two-period model.

If, on the other hand, $s > 0$, then the biomass in the next period is no longer independent of the harvesting strategy applied in the present period. B_2 will not be the same with constant harvest as with constant effort. In this case the optimal constant policy may have an interior solution that can be found analytically or numerically. In the two-period case with B_1 given, a necessary condition for optimum is given by

$$E^* = \frac{p\,[B_1(1+s) + w] - 2c}{2psB_1}. \tag{8a}$$

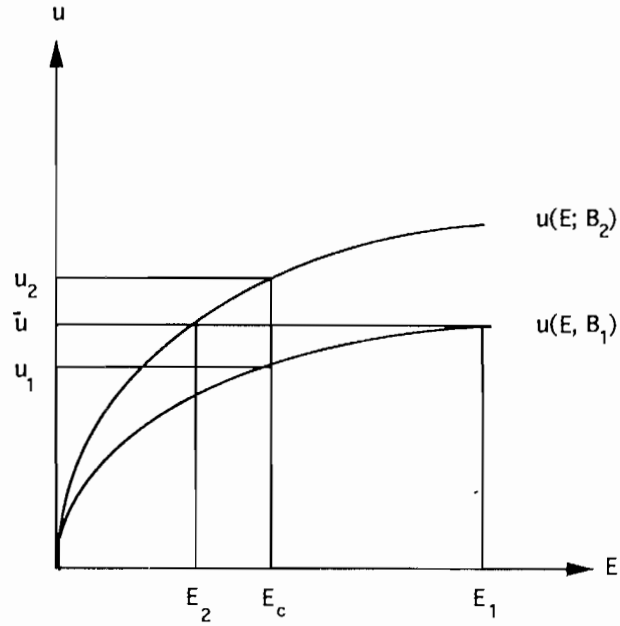


Fig. 1. A two-period model. The constant effort E_c gives $\bar{u}$ on average. $\bar{u}$ applied as constant catch on the other hand requires E_1 and E_2 in the two periods which have a higher average than E_c.

E^* is defined to be optimal only if $0 \leq E^* \leq 1$. Equation (8a) yields an interior solution ($0 < E^* < 1$) as long as

$$\begin{aligned} \frac{1}{2}\,[B_1\,(1+s) + w] - sB &< \frac{c}{p} \\ &< \frac{1}{2}\,[B_1\,(1+s) + w]. \end{aligned} \tag{8b}$$

If the left inequality is not fulfilled, $E^* = 1$, and if the right inequality is not fulfilled, $E^* = 1$. The optimal constant harvest, u^*, in a two-period model is given by

$$u^* = B_1 + \frac{w}{s} - \frac{1}{s}\left\{\frac{c\,(sB_1 + w)}{2p - c/B_1}\right\}^{1/2}$$

The constraints on u^* are:

i) $u^* \geq 0$,

ii) $u^* \leq B_1$,

iii) $u^* \leq B_2$.

i) is fulfilled if

$$\frac{c}{p} \leq \frac{2\,(sB_1 + w)}{1 + s + \frac{w}{B_1}}$$

ii) is fulfilled if

$$\frac{c}{p} \geq \frac{2B_1\,(sB_1 + w)}{B_1\,(1+s)^2 + (sB_1 + w)}$$

iii) is fulfilled if

[1] An optimal constant policy will be either zero or maximal effort depending upon whether $\sum(B_t - c/p)$ is greater than or less than zero.

$$\frac{c}{p} \geq \frac{2w^2 B_1}{w^2 + B_1 (sB_1 + w)},$$

where c/p is the cost-price ratio. It is easy to check that iii) is fulfilled whenever ii) is fulfilled as long as $B_1 \geq w$. With a time-horizon longer than two units, it will be necessary to resort to numerical solution methods to find E^* or u^*. The maximum net revenue from harvesting with the optimal constant effort is given by

$$J(E^*) = \frac{[pB_1 (1+s) + pw - 2c]^2}{4psB_1} \tag{9}$$

If we ignore the optimal constant catch for the moment and instead use the average harvest from the constant effort policy as a constant catch we will obtain

$$\bar{u} = \frac{[pB_1 (1+s) + pw]^2 - 4c^2}{8p^2 sB_1} \tag{10}$$

The necessary effort in each period to take this harvest is $\bar{u}/B_i$ $(i = 1, 2)$. The net value of harvesting with constant catch is given by

$$\begin{aligned} J(\bar{u}) &= \left(p - \frac{c}{B_1}\right)\bar{u} \\ &+ \left(p - \frac{c}{s(B_1 - \bar{u}) + w}\right)\bar{u} \end{aligned} \tag{11}$$

$J(\bar{u})$ will be higher than $J(E^*)$ if the average of $\bar{u}/B_i$ $(i = 1, 2)$ is smaller than E^*. Given that $B_1(1-s) < w < B_1$, an equilibrium effort, $0 < E^\dagger < 1$, can be found such that $B_1 = B_2$. This equilibrium effort is given by

$$E^\dagger = \frac{w - (1-s)B_1}{sB_1} \tag{12}$$

and applying this results of course in constant harvest as well as constant effort[2] Further it can be shown that with any E lower than $E^\dagger$, $J(\bar{u}) < J(E)$, and vice versa, where $\bar{u}$ is the average harvest generated from any effort level applied, not necessarily the optimal one (see appendix A). The interesting question now is to see how E^* in particular relates to $E^\dagger$. It can easily be shown that $E^* > E^\dagger$ if and only if

$$\frac{c}{p} < \frac{B_1 (3-s) - w}{2} \tag{13}$$

In this case there will always exist a constant catch that can do better than the optimal constant effort, since the average of the catches that could be produced by the optimal effort is such a catch. Furthermore, this average catch may not be the optimal constant catch; there might exist another constant catch that can perform even better. Note that the right side of (13) will always be less than the right side of (8b) as long as $w > B(1-s)$ which is necessary for $E^\dagger > 0$. The right side of (13) is obviously increasing in B and decreasing in w and s. Let us now sum up the five different cases that may appear in the two-period model:

(i) $\quad \frac{c}{p} \leq \frac{1}{2}[B_1 (1+s) + w] - sB \rightarrow E^* = 1$

(ii) $\quad \frac{1}{2}[B_1 (1+s) + w] - sB < \frac{c}{p} < \frac{B_1(3-s) - w}{2} \rightarrow E^\dagger < E^* < 1$

(iii) $\quad \frac{c}{p} = \frac{B_1(3-s) - w}{2} \rightarrow E^* = E^\dagger$

(iv) $\quad \frac{B_1(3-s) - w}{2} < \frac{c}{p} < \frac{1}{2}[B_1 (1+s) + w] \rightarrow 0 < E* < E^\dagger$

(v) $\quad \frac{c}{p} \leq \frac{1}{2}[B_1 (1+s) + w] \rightarrow E^* = 0$

Case (i): This case is not very interesting since the escapement, $B - u$, will be zero. Although recruitment is not directly linked to the escapement, there has to be at least some minimum escapement in order to secure future recruitment, a question that will be addressed further in a moment.

Case (ii): This expression is only meaningful if $B_1 > w$. Here constant catch is better than constant effort. The term on the left side is interpreted as the average biomass during the two periods in the absence of harvesting minus the biomass that disappears due to natural mortality. With the production function given by Equation (1), c/p is equivalent to the biomass resulting from a fishery with free access. The term on the right side of c/p is the same as $B_1 - \sigma$ where σ is the standard deviation of B_1 and $B_2(0)$. $B_2(0)$ is the stock level in period 2 without harvesting. This assumes that $B_1 < B_2(0)$, in other words that the stock is not above its natural carrying capacity.

Case (iii): Obviously in this case where $E* = E^\dagger$, the two strategies will perform equally well, and the biomass will remain at its equilibrium level. Case (iii), however, is a knife-edge case as there is only a single value of c/p that would lead to this case, and the actual value of c/p would only be equal to this by chance.

Case (iv): This expression is only meaningful if $B_1 < B_2(0)$, i.e., the initial biomass is below the natural carrying capacity of the stock. In this case constant effort is a better choice than constant catch. An interpretation of the term on the left side is given under case (ii). Thus the region in which constant effort is better than constant catch becomes larger as the standard deviation, σ becomes larger. The term on the right side of

[2] $E^\dagger$ is increasing in w, decreasing in B_1 and increasing in s provided that $B_1 > w$ which is necessary for $E^\dagger < 1$.

c/p gives the average biomass during the two periods without harvesting.

Case (v): In the case that $E* = 0$, the net revenue in each period is zero.

The two-period model describes a situation in which managers decide upon quotas for only one year at a time. This is the case in many fisheries due to, among other things, the fact that good estimates of recruitment are only available for one or a few periods hence. The results above indicate that whether constant catch is better than constant effort depends upon the relationship between the biomass resulting from free access (defined by economic parameters only) and the expected average biomass during the two periods without harvesting and its variation (standard deviation) in the way described above.

In a two-period model an important difference between the two strategies considered here lies in the escapement after period two is over. One reason why constant catch performs better in case ii) (and constant effort in case iv)), is that it leaves behind a lower escapement, $(B_2 - u)$, at the end of the time horizon. In some cases the optimal constant policy may be the one that results in $B_2 - u = 0$, and some constraint therefore has to be placed on the minimum allowable escapement, e.g., $B_2 - u \geq \xi_{\min}$.

$E^\dagger$ plays a crucial role also with respect to escapement. Defining the escapement at the end of period two with constant effort as

$$\xi(E) = [sB_1(1 - E) + w](1 - E) \tag{14}$$

and with constant catch as

$$\xi(u) = s(B_1 - u) + w - u \tag{15}$$

it is easily shown that $\xi(\bar{u}(E)) < \xi(E)$ whenever $E > E^\dagger$ where

$$\bar{u}(E) = \frac{1}{2}[B_1 + sB_1(1 - E) + w]E$$

is the average catch for any E. In other words, the price that is paid for a better economic performance is a smaller escapement at the end of the two periods since whenever $E > E^\dagger$, $J(E) < J(\bar{u}(E))$ and $\xi(E) > \xi(\bar{u}(E))$ and vice versa. It may be considered worthwhile to pay this price if it is difficult to prove any relationship between escapement and future recruitment, and a minimum escapement is the only necessary requirement. If the cost-price ratio is low, the optimal strategy may be the one that drives the escapement down to its minimum level at the end of the time horizon and only in this case will it be possible to increase the economic performance of any of the strategies by decreasing $\xi_{\min}$.

Even when the constant catch is calculated as the average harvest produced by some constant effort level, there may be a significant difference in the net revenues yielded by the two strategies. If an optimal constant catch is applied, this difference becomes even more pronounced. Another point worth noting is that the average biomass level is higher with constant catch than with constant effort whenever $E > E^\dagger$ and vice versa.

Now it may be interesting to compare these constant policies with the optimal unconstrained policies. Regarding the harvest, u, as the control variable, the optimal policy is easily found using Bellman's principle for dynamic optimization (Bertsekas 1987). Denoting the harvest in the two periods u_1 and u_2 respectively, the optimal solution requires

$$u_2 = \begin{cases} u_{2\max} & \text{if } p - c/B_2 > 0; \\ 0 & \text{if } p - c/B_2 < 0. \end{cases}$$

$u_{\max}$ is given by $B_2 - \xi_{\min}$. Applying the optimal policy in period two gives

$$J_2^*(B_2) = \max_{u_2} \pi(B_2, u_2). \tag{16}$$

π is given by equation (2a). In period 1

$$J_1^*(B_1) = \max_{u_1} \{\pi(B_1, u_1) + J^*(B_2)\} \tag{17}$$

Whenever $u_1 < B_1 + w/s - c/ps$, $u_2 = u_{\max}$. Further, when this is satisfied

$$\frac{\partial J_1^*(B_1)}{\partial u_1} = \frac{2c\xi s^2}{[s(B - u_1) + w]^3} > 0 \tag{18}$$

such that the only two values of u_1 in the interval $[0, B_1 + w/s - c/ps]$ that need to be taken into account are the end-values. If $u_1 > B_1 + w/s - c/ps$, then $u_2 = 0$ and

$$J_1^*(B_1) = \pi_1(B_1, u_1) \tag{19}$$

implying that

$$u_1 = \begin{cases} u_{1\max} & \text{if } p - c/B_1 > 0; \\ 0 & \text{if } p - c/B_1 < 0. \end{cases}$$

In order to find the optimal policy, the value of $J_1(B_1)$ has to be calculated for these three values of u_1:

$$\begin{aligned} u_1 &= 0, \\ u_1 &= B + \frac{w}{s} - \frac{c}{ps} \text{ and} \\ u_1 &= u_{1\max} \equiv B_1 - \xi_{\min}. \end{aligned}$$

In Table 1, I present some numerical examples to illustrate the consequences of the optimal constant policies and the unconstrained optimal policy. This table gives biomass, harvest and net revenue in each period as well as average values. Values of the parameters were arbitrarily chosen as $B_1 = 10$, $s = 0.8$, $w = 4$, $p = 1$ and $c = 7$.

The constant catch policy with the same average as the optimal constant effort gives five per cent higher average net revenue than constant effort, and the optimal constant catch gives six per cent higher net revenue. The unconstrained (not constant) optimal policy on the other hand gives 69% higher net revenue than the optimal constant effort. The constant catch policies clearly have the lowest variance in net revenues, and this may weigh heavily when a best policy has to be chosen.

Is there anything to be learned from this two-period model that can be of practical use and is there any reason to apply this

Table 1. A numerical example of a two-period model. B is biomass, E is effort, u is catch, π is profits and $\xi = B_2 - u_2$ is escapement.

Time Period (i)	B_i	E_i	u_i	π_i	ξ
Optimal constant effort policy					
1	10	0.5	5	1.5	
2	8	0.5	4	0.5	
average	9	0.5	4.5	1.0	4
Constant harvest policy					
1	10	0.45	4.5	1.35	
2	8.4	0.54	4.5	0.75	
average	9.2	0.49	4.5	1.05	3.9
Optimal constant harvest policy					
1	10	0.495	4.952	1.49	
2	8.04	0.616	4.952	0.64	
average	9.02	0.556	4.952	1.06	3.09
Optimal unconstrained policy ($\xi = 3.1$)					
1	10		0	0	
2	12		8.1	3.375	
average	11		4.05	1.69	3.1

further? To answer this it is necessary to compare the results from the two-period model with a longer time horizon. On the other hand, the time horizon may be infinite. The result from a model with infinite time horizon, is that optimal constant effort and optimal constant catch coincide with the equilibrium solution, assuming that recruitment varies stochastically. Defining optimal as maximizing average net return per period, the optimal long-run equilibrium biomass level is given by:

$$B^{\infty} = \sqrt{\frac{cw}{p(1-s)}}$$

and the corresponding optimal long-run catch and effort are

$$u^{\infty} = \frac{w - \sqrt{(1-s)cw/p}}{s}$$

and

$$E^{\infty} = \frac{1}{s}\left\{\sqrt{\frac{(1-s)pw}{c}} - (1-s)\right\}.$$

respectively. In the equations above w can be both a constant recruitment or the expectation of a stochastic recruitment sequence. It is relatively easy to check that $E^{\infty} < E^*(B^{\infty})$ as long as the effort level is subject to $0 \leq E \leq 1$. The optimal effort level for a finite time horizon will only approach E^{∞} as the time horizon approaches infinity. A short time horizon model may be considered as a substitute for a high discount rate, i.e., it emphasizes the effects of a myopic decision pattern. A two-period model is chosen since this may give some interesting analytical results as opposed to models with a longer but finite horizon, given the dynamics assumed here. An objective of this paper is to check by numerical simulations whether these results also hold true in a stochastic model with a long time horizon.

Discounting of the net revenue in the second period by introducing a discount rate in the two-period model, may affect some of the conclusions above. A very high discount rate may, for example reverse some of the results from Table 1 whereas a moderate or low discount rate (less than 10%) probably will have no effect on the qualitative conclusions. Introducing a discount rate in the model with an infinite horizon, will increase the harvest rate and reduce the equilibrium biomass level, and thus make this result more similar to the short horizon model.

So far the economic parameters p and c have not played any important role. The main results were based on physical units of effort and catch, not their economic values. The economic parameters were mainly used to find the optimal effort, E^*. If, however, p and c cease to be constant parameters, but become variables dependent on u and E respectively, their role will become more important.

Consider the role of a downward sloping demand function first, i.e., $p = p(u)$[3]. In order to investigate the partial effect of a downward sloping demand curve, it is only necessary to look at the gross revenue. It is easy to find examples of downward sloping demand functions that will make the constant catch strategy a relatively better choice compared to the situation with a fixed price. A linear demand function is one such example:

$$p(u) = a - bu. \tag{20}$$

With a linear demand function the following will always be satisfied, except when $u_1 = u_2$:

$$p(u_1)u_1 + p(u_2)u_2 < p\left(\frac{u_1+u_2}{2}\right)\cdot\frac{u_1+u_2}{2}. \tag{21}$$

If $-p''(u)u/p'(u) < 2$, (21) will always be fulfilled. However, if $p''(u) > 0$, (21) may not be satisfied. For example, if $p(u) = au^{-1}$, the increased price when u is low exactly offsets the decreased price when u is high, and (21) becomes an equality and thus equivalent to a fixed price. As may be expected from this, if $p(u) = au^{\beta}$, (21) will only be satisfied if $-1 < \beta < 0$. Thus if $\beta < -1$, a downward sloping demand function will favour the constant effort strategy. It is tempting therefore to suggest the conclusion that an inelastic demand will favour constant effort whereas an elastic demand will favour constant catch. This conclusion is a bit premature since, as it was seen above, a linear demand function which can be both elastic and inelastic, will unequivocally favour constant catch. In the case of a demand function

[3]Remember that we are only interested here in maximizing the producers' surplus and therefore are ignoring the consumers' surplus.

with constant demand elasticity, $1/\beta$, of the type above, the conclusion holds true. The conclusion therefore is that the following types of demand functions at least will favour constant catch: 1) linear demand functions, 2) concave demand functions and 3) constant elasticity demand functions which are elastic.

The whole story however is not told by this partial analysis of the gross revenue. The downward sloping demand function may force the optimal constant values of harvest and effort down in order to take advantage of the higher price resulting from a lower output. It is known that when the optimal effort is forced below the equilibrium value $E^{\dagger}$, such that $B_2 > B_1$, constant effort will be a better alternative than constant catch, and this argument is based on the physical characteristics of the system. This means that a demand function that in a partial perspective favours constant catch, may force the optimal constant harvest or effort below the equilibrium level, and the effect of this may outweigh the partial effect of the demand function on the gross revenue.

Continuing the numerical example from the previous section may illustrate this. Note that the demand function $p(u) = 4 - 0.5u$, results in an optimal constant effort level $E* = 0.33$ implying $J(E^*) = 10.72$, whereas the optimal constant catch $u^* = 3.2$ implies $J(u^*) = 10.75$. Clearly the catch rate is reduced and the difference between the strategies is smaller compared to a fixed price $p = 1$. If the demand curve is shifted down and given by $p(u) = 3 - 0.5u$, then $E^* = 0.22$ and $J(E^*) = 5.320$ whereas $u^* = 2.25$ and $J(u^*) = 5.318$. The difference between the two strategies is still small but now it is reversed, making constant effort the better choice, as a result of the linear demand function which by itself was supposed to favour constant catch. This is due to the fact that the catch rate is forced below the equilibrium level given by $E^{\dagger} = 0.25$ and $u^{\dagger} = 2.5$.

Next, take a look at the situation where cost per unit effort is a function of the effort level applied. For this purpose it is only necessary to take a partial view at the cost side, and the reasoning will in many respects be similar to the downward sloping demand function case. The function $c(E)$ is of course increasing, and if it is linear, i.e., $c(E) = a + bE$, the following will always hold true:

$$(22)\quad c(E_1)E_1 + c(E_2)E_2 > c\left(\frac{E_1 + E_2}{2}\right)\cdot\frac{E_1 + E_2}{2}$$

except when $E_1 = E_2$. In other words a linear cost function will favour the constant effort strategy[4]. The inequality in (21) will also be fulfilled with a convex cost function, i.e., $c''(E) > 0$, and in fact any cost function satisfying

$$(23)\qquad \frac{-c''(E)E}{C'(E)} < 2.$$

In the case of a function with constant cost elasticity, $c(E) = aE^{\beta}$, (23) reduces to $-(\beta - 1) < 2$ implying $\beta > -1$, which of course is satisfied since $c(E)$ is supposed to be increasing. The only case in which (23) is not fulfilled, is if $c(E)$ is highly concave which a priori does not seem very likely.

Also the increasing cost function will, like the downward sloping demand function, reduce the catch rate under an optimal (constant) policy. In this case the reduced catch will reinforce the most likely effect of an increasing cost function, namely that it favours constant effort. For these reasons it seems that a priori it is quite likely that an increasing cost per unit effort will favour constant effort, whereas the effect of a downward sloping demand function is a bit more ambiguous. The analysis makes it clear, however, that in neither case will it be possible to get an unambiguous result, even within the framework of this simple two-period model. It is therefore necessary to undertake an empirical study to make the analysis complete. This will be the subject of the next section.

So far it has been assumed that there is no risk-aversion, in other words the utility function has been assumed to be linear in net revenue. If this is not the case, i.e., if the utility function is concave, this will favour the strategy with less variation in net revenue. The numerical example above indicates that this will be the constant catch policy, but this is not necessarily the case, and the effect of a concave utility function will therefore not be totally unambiguous either. With a constant catch strategy, the gross revenue will of course be constant, but costs will vary, with constant effort it is the gross revenue that varies when costs are constant thus making the total effect ambiguous. However, under the relatively mild assumption that the fishery yields a non-negative expected net revenue, the following proposition will hold true in the two-period context:

Proposition 1 *Assuming a time-horizon of two periods, a non-negative expected revenue, a production function given by (1) and dynamics given by (3a) or (3b), a constant effort policy will never have less variance in net revenues than the constant harvest policy yielding the same average catch. For a proof, see appendix B.*

It is also interesting to note the effect that risk-aversion will have in combination with a downward sloping demand function or increasing cost function under the different policies. In the case of constant catch the demand function will have no effect whereas an increasing cost function will have an amplifying effect on the variance in net revenue. With constant effort, on the other hand, the cost function will have no effect but a downward sloping demand curve will reduce the variance in net revenue. Thus both a downward sloping demand function and an increasing cost function will counteract the difference in variance between the two strategies pointed out above, making unambiguous a priori results even harder to obtain.

[4]The use of the term cost function is strictly speaking not correct here since a cost function is supposed to be a function of output, not the input in production. However, the meaning of the term, as it is used here, has been established as the common usage in fisheries economics.

Stochastic analysis

This section is devoted to explore the effects of stochasticity. The difference between the constant effort and the constant catch strategy will be explored and a reference to the deterministic counterpart of the model will be made in order to find the shadow price of having uncertainty. The basic model will be extended in several directions; i.e., biological and economic complexities will be included in the model, one by one, to the extent this is possible without making the model intractable. The basic model is in other words a special case of the more general model, in which most of the biological and economic parameters are treated as constants. Experience show that analytical results from such models are difficult to obtain, except for some special cases, and therefore numerical results will be provided in most of this chapter.

The model

The model is an aggregate biomass model where the dynamics are given by Equations (3a) and (3b). The production function is assumed to be the same as in Equation (1). The objective function is given by

$$V^* = \max \sum_{t=0}^{T} \alpha^t \pi(\cdot, B_t) \tag{24}$$

where π is the net revenue function in each period, see Equations (2a) and (2b), and α is the discount factor $(1+r)^{-1}$, r is the discount rate. $\partial\pi/\partial B > 0$ whereas $\partial\pi/\partial u$ and $\partial\pi/\partial E$ are only greater than zero if $(p - c/B) > 0$.

Comparing constant effort with constant catch will only be interesting if the fish stock biomass varies for some reason, and in this section the assumption will be that this variation is due to stochasticity in recruitment. The dynamics are still given by Equations (3a) and (3b), but now the recruitment w_t is a stochastic variable with expectation μ. However, if the recruitment is a stochastic variable varying freely and independently of the parent population, except for the minimum escapement requirement, the optimal policies will not be much different from the constant recruitment case. This is rather obvious since in this case recruitment will be an additive stochastic variable in the dynamic equation. Although the optimal solution is the same in the stochastic model as in the deterministic, the performances of the two strategies need not be similar in the stochastic case, and in fact they are not. If expected recruitment is μ, the expected biomass in each period will be $\overline{B}$ assuming that the initial biomass is at an equilibrium level. Suppose that the actual B_t lies in the range $[B^L, B^U]$ which is symmetric around $\overline{B}$ as the distribution of B is assumed to be uniform. Then the expected net revenue with constant catch (CC) will be

$$\varepsilon\pi(CC) = \int_{B^L}^{B^U} (p - c/x) u f(x)\, dx$$

and with constant effort (CE)

$$\varepsilon\pi(CE) = \int_{B^L}^{B^U} (p - c/x) E f(x)\, dx$$

It can easily be shown that $\varepsilon\pi(CE) > \varepsilon\pi(CC)$ given that $B^U > B^L$ when the probability density function $f(\,)$ is defined to be constant (see appendix C).

There exist several ways to make the analysis more interesting. One is to include the cost of irreversible investments in fixed capital. Another is to introduce a spawning stock-recruitment relationship and this is the approach that will be concentrated on here. With a deterministic discrete time model the conditions for optimal harvest policies in such models are well known from e.g., Clark (1990). Assuming w_t in Equations (3a) and (3b) is a function

$$w_t = f(B_t - u_t) \tag{25}$$

the dynamic recurrence equation can be written as a function

$$B_{t+1} = F(\xi_t) = s\xi_t + f(\xi_t) \tag{26}$$

where

$$\xi_t \equiv B_t - u_t.$$

ξ_t is the escapement in period t. Given an objective function

$$J = \sum_{t=0}^{\infty} (1+r)^{-t} \pi(B_t, u_t) \tag{27}$$

where

$$\pi(B_t, u_t) = \int_{S_t}^{B_t} [p - C(x)]\, dx, \tag{28}$$

the optimal escapement, ξ^*, is given by

$$F'(\xi^*) \frac{p - C(F(\xi^*))}{p - C(\xi^*)} = 1 + r, \tag{29}$$

where r is the annual rate of discount. In this case the optimal solution implies both constant effort, constant catch and constant escapement. With a specific recruitment function, however, this condition soon becomes quite messy, and therefore the problem naturally lends itself to numerical solution methods. As an example, note that the optimum condition in the case of a Beverton-Holt recruitment function, $F(\xi) = \alpha\xi/(1+\beta\xi)$, is that ξ^* given by

$$\begin{aligned} ps(s-1)\left(1+\beta\xi^*\right)^3 + p\alpha(s-1)\left(1+\beta\xi^*\right)^2 \\ + (p\alpha s + c\alpha\beta)\left(1+\beta\xi^*\right) + p\alpha^2 \quad = \quad 0 \end{aligned}$$

which has three roots to be investigated.

Unfortunately the real world is not as simple as the deterministic model suggests. If annual recruitment is influenced by stochasticity, the optimum conditions will change. Assume that the recurrence equation is given by

$$B_{t+1} = F(B_t, u_t, \varepsilon_t) = s\xi_t + f(\xi_t)\,\varepsilon_t \tag{30}$$

where ε is a uniformly distributed random variable with expectation one. A multiplicative random variable seems appropriate in the presence of a spawning stock-recruitment relationship as it guarantees that the recruitment will never be negative. Assume further that revenue is linear in catch and that there is a constant cost function; i.e., the objective function is given by

$$J = \mathrm{E}\left[\sum_{t=1}^{T}(1+r)^{1-t}u_t\right]. \tag{31}$$

Here E denotes expectation. According to Parma (1990) and Reed (1974 and 1979), if the recruitment function is continuously differentiable and strictly concave, the optimal policy is to harvest in any year t:

$$u_t^* = \begin{cases} B_t - \xi_t^* & \text{if } B_t \geq \xi_t^*; \\ 0 & \text{if } B_t < \xi_t^*. \end{cases} \tag{32}$$

where $\xi_t^* = g(\xi_{t-1})$. If there is a lag in the recruitment function of k periods, then the escapement levels $\xi_{t-2}, \ldots, \xi_{t-k}$ have to be included in this function as well. This result assumes a positive survival of older fish. The stochastic model with a lag has been analyzed by Parma (1990), the stochastic model without a lag by Reed (1974 and 1979) and the deterministic model with a lag by Clark (1990). They all conclude that the optimal policy has to be based on a target escapement level, ξ_t^*. In the stochastic models $\xi_t^* \geq \xi^d$ where ξ^d is the optimal solution in the deterministic counterpart of the model. Of course, except in deterministic models, an optimal policy implies both variable catch and variable effort.

In the following the difference between constant effort and constant catch in such a model will be discussed under varying assumptions using Monte Carlo simulations.

These are the basic parameter values in the model: $p = 1$, $c = 6$, $s = .8$, $\alpha = 1$, $\beta = .1$ where α and β are the parameters in the Beverton-Holt recruitment function (recall the earlier discussion on this function on page 379). These parameter values may be thought of as representing some stylized characteristics of the North-East Arctic cod stock when biomass is measured in 100,000 tonnes and effort is measured in arbitrary units. First the optimal catch, effort and equilibrium stock are calculated for the deterministic equivalent in each case. This equilibrium stock is then used as initial stock in the stochastic model, and the optimal catch and effort from the deterministic model is used as fixed catch and effort in the stochastic model. By comparing the economic result, i.e., present value of net revenue, from the stochastic model with the deterministic model, the cost of uncertainty is found.

The optimal deterministic solution is found by maximizing

$$\pi = pEB - cE \tag{33}$$

subject to

$$B = \frac{\alpha(1-E) - [1 - s(1-E)]}{\beta(1-E)[1-s(1-E)]} \tag{34}$$

Table 2. Characteristics of the simulations. ε is a random variable, r is the discount rate, p is the price and c is the marginal cost.

	Simulation						
Characteristic	1a	1b	2	3	4	5	6
$0.75 \leq \varepsilon \leq 1.25$	+		+		+	+	+
$0.50 \leq \varepsilon \leq 1.50$		+		+			
$r = 0.0$	+	+					
$r = 0.1$			+	+	+	+	+
$p = a - bu$				+			
$c = a + bE$					+		
$s = a - bB$						+	
$u = EB$	+	+	+	+	+	+	
$u = B\left(1 - e^{-E}\right)$							+

This gives $E^* = 0.148$, $B^* = 19.7$, $u^* = 2.91$ and $\pi^* = 2.023$. In the stochastic case the simulation is performed over 50 periods with recruitment in each period calculated according to the Beverton-Holt recruitment function and multiplied by a random, uniformly distributed variable ε, see Equation (30). By the Central Limit Theorem it is expected that the distribution of $J(\cdot)$ is normal, and that the true value of J must lie within two standard deviations from its mean with 95 per cent confidence. A five per cent significance level is used in all simulations presented here, and 95 per cent confidence intervals are constructed for the differences between the means of the two strategies. The standard deviation applied here is $s/\sqrt{n}$, where s is the estimated standard error and n is the number of observations. The estimated variance of the difference is the pooled estimate of the variance from each of the two strategies when the individual variances were equal. In the Monte Carlo simulations 100 different patterns of the random variable are drawn. Two different levels of stochasticity may be used: $\varepsilon \in [0.75, 1.25]$ is defined as the low stochasticity level and $\varepsilon \in [0.5, 1.5]$ is defined as the high stochasticity level. A variation in recruitment of the North-East Arctic cod stock of $\pm 50\%$ is not unrealistic.

The characteristics describing each of the simulations are given in Table 2. In Table 3 E^*, B^*, u^* and J^* from the deterministic counterparts of the models are listed. The results from the simulations based on the stochastic models are given in Table 4. $\overline{J}$ is the average J for all random patterns. The number of extinctions reported in Table 4 refers to the number of simulations where it is not possible to adhere to the constant catch strategy throughout the entire time horizon of 50 years without endangering the stock. This can be thought of as an index on the feasibility of the strategy. This is a rather strong criterion since in reality it is more likely that a certain constant catch will be recommended for five years, not 50. Some comments on each of the simulations are made below.

Simulation 1.

The first simulation is based on the basic model with a zero discount rate. With the low level of stochasticity there

Table 3. Results from the deterministic model. E^* is the optimal effort, B^* is the optimal biomass level, u^* is the optimal catch and J^* is the net revenue in optimum.

Simulation Number	E^*	B^*	u^*	J^*
1	0.148	19.7	2.91	101.15
2	0.19	16.06	3.05	20.83
3	0.082	26.72	2.20	28.20
4	0.171	17.59	3.02	21.27
5	0.235	15.32	3.60	23.89
6	0.19	17.46	3.02	20.51

is a significant difference (at the 5% level) between the profitabilities of the two strategies, but no instances of extinction. With increased stochasticity; i.e., $\varepsilon \in [0.5, 1.5]$, there is not only a difference between the two strategies, but in the case of CC, 23 out of the 100 recruitment patterns that are drawn result in extinction of the stock. These 23 patterns are removed from the simulations of both strategies in order to make them comparable. $\overline{J}$ for CC has decreased and the $\overline{J}$ for CE has increased, in other words increased stochasticity decreases the profitability of CC but increases the profitability of CE. In comparing with the net revenue in the deterministic case, which was $J = 101.16$ over 50 periods, only the result obtained with the low level of stochasticity can be taken into account since the 23 simulations that were removed will be favourable for the stochastic case compared to the deterministic (only the "worst" random patterns leading to extinction were removed). In any case, the shadow price of uncertainty seems to be relatively small with the low level of stochasticity.

The reason why the constant catch leads to extinction and not the constant effort stems from the different stability properties of the two strategies. With CE only part of the total biomass is harvested each period and since recruitment is always non-negative, the stock can never go completely extinct. It can be shown that in the deterministic case with this strategy there will always exist a global, stable equilibrium population (Fig. 2). With constant harvest, on the other hand, the stock may at a point in time not be able to support the long-run optimal harvest, and the stock will consequently go extinct. The dynamics in the deterministic model in this case show that there may be zero, one or two equilibrium stocks, of,which only one can be stable. The occurrence of several "bad years" with recruitment may force the stock down below the unstable equilibrium, and the likelihood that the stock ultimately will go extinct increases considerably. This is illustrated in Fig. 2 where B^* is the optimal long-run equilibrium with both strategies and B' is the unstable equilibrium with the CC strategy.

Simulation 2

So far the discount rate has been zero. The next question therefore comes naturally: what is the effect of a positive discount rate? Even with the moderate level of stochasticity, 30% of the recruitment patterns lead to extinction and therefore are discarded. This excludes CC as a practical alternative when both the size of the discount rate and the level of stochasticity are considerable, unless there exist other factors that may modify this conclusion. The result based on the remaining 70 observations in the case of moderate stochasticity is reported in Table 4. However, our null-hypothesis saying that the difference between the strategies is zero can not be rejected. In other words, if the CC strategy has to be ruled out, it is not due to lack of profitability since although the average profitability is lower it is not significantly lower at a five% significance level. The reason why a higher discount rate increases this risk, is the higher optimal catch rate implied by a high discount rate.

Simulation 3

A factor which may work in the opposite direction, is a downward sloping demand function. According to the theory from a previous section, a linear demand function should work unequivocally in favour of CC. This is tested in the stochastic case by using the following demand function:

$$p(u) = 2.5 - 0.5u.$$

Table 4. Results of simulation study. Characteristics for simulation number 1–6 are described in Table 2. Note confidence intervals (CI) which do not include zero indicate a significant difference between $\overline{J}_{cc}$ and $\overline{J}_{ce}$ at the 5% significance level.

	Constant Catch		Constant Effort	
Simulation Number	Number of Extinctions	Average Net Value $\overline{J}_{cc}$	Average Net $\overline{J}_{ce}$	95% CI for $(\overline{J}_{cc} - \overline{J}_{ce})$
1a	0	99.72	100.99	(−2.37, −0.17)
1b	23	98.59	102.91	(−6.35, −2.86)
2	30	20.92	21.21	(−0.59, 0.01)
3	0	28.07	27.85	(0.10, 0.34)
4	11	21.01	21.36	(−0.67, −0.03)*
5	31	23.81	24.13	(−0.60, −0.04)
6	14	20.39	20.67	(−0.56, 0.004)

* corrected for unequal variances.

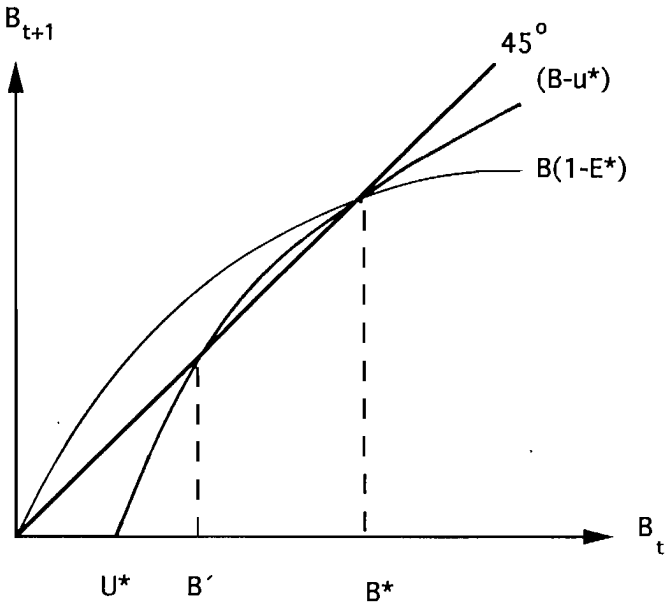


Fig. 2. B^* is assumed to be the optimal equilibrium stock with both strategies. B' is an unstable equilibrium with constant catch.

The optimum from the deterministic model is reported in Table 3. It is noticed that the optimal catch rate is drastically reduced due to the downward sloping demand function even though a positive discount rate, $r = 0.1$, is retained. With these values for E, B and u above as input, the results from the Monte Carlo simulations are given in Table 4. Even though the high level of stochasticity is applied and a positive discount rate is used, the downward sloping demand function makes the optimal harvest so low that no instances of extinction is observed in the case of CC. Further, the result from the Monte Carlo simulations is that zero is not included in the interval, thus making CC the preferred strategy. In other words, a downward sloping demand function may outweigh both the effect of high stochasticity and a high discount rate, not only to the extent that the two strategies can compete but even to the extent that CC is better than CE. This conclusion is fairly robust both to changes in the slope and in the intercept of the demand curve. For example, with the demand function $p(u) = 7 - 2u$, we have $E^* = 0.055$, $u^* = 1.67$ and $B^* = 30.4$. In this case the average $\overline{J}(\mathrm{CC}) = 63.0$ whereas $\overline{J}(\mathrm{CE}) = 62.4$.

Simulation 4

A linear cost function, in the sense that cost per unit effort is a linear function of the effort level applied, is supposed to work in favour of constant effort. However, it will reduce the optimal effort level and catch rate, thus reducing the risk of extinction in the case of CC. The following cost function is suggested:

$$C(E) = 4.5 + 10E,$$

Remember that $C(E)$ is here the cost per unit effort such that total costs, $C(E)E$, will be convex in effort. The optimum from the deterministic model is given in Table 3. Only the slight level of stochasticity is used in the Monte Carlo simulations. The result is that 11 out of the 100 simulations have to be discarded because they caused extinction in the case of CC. However, the 95% confidence interval based on the 89 remaining observations does not contain zero (Table 4). The conclusion is that an increasing cost function significantly increases the difference between the strategies with respect to net present value of income. On the other hand, an increasing cost function does imply a lower effort level and catch rate which reduces the risk of extinction in the case of CC.

Simulation 5

The next question is: What effect does it have if the survival parameter (or natural mortality) is made density dependent? In this case it is hard to say anything a priori about the likely difference between the two strategies. It is, however, obvious that if the natural mortality is an increasing function of the stock size, the optimal catch rate will be higher in order to increase the survival of the remaining stock. As it has been seen earlier, a high catch rate combined with a high level of stochasticity may be detrimental if a CC strategy is applied. Only the moderate level of stochasticity is used here to compare the two strategies. The following linear density dependent survival function is applied:

$$s = 1 - 0.01B.$$

The optimal equilibrium in the deterministic case is reported in Table 3. Even with the low level of stochasticity, a CC strategy leads to complete extinction in 31 out of 100 simulations. Thus the CC strategy has to be ruled out in this case, both because it performs worse in the presence of density dependent natural mortality, and because the higher optimal catch rate implied endangers the stock.

Simulation 6

The next step is to take a closer look at the production function. So far it has been assumed that catch is proportional to effort for a given stock and vice versa. This is at best a simplification.

For one thing, the profit function in Equation (28) gives the profit as an integral from B_t to S_t. In the simulations presented here, the profit in each period is calculated according to the expression in Equation (33). This simplification does not seem to make any big difference. An obvious way in which the form of the production function may affect the choice of strategy is by the degree of interaction between the fish stock and the fishing effort. The larger the "role" of the fish stock in the production function, the greater the difference between the two strategies may be. If the catch, on the other hand, is proportional to the effort level and independent of the stock, there will be no difference between the two strategies.

As a more realistic alternative to the production function used so far, $u = EB$, the following function is suggested:

$$u = B\left(1 - e^{-E}\right).$$

In this case u is still linear in B but concave in E. Further, whatever units are chosen for E, it will not be possible to harvest more than the entire stock, thus there need not be any limitations on the range of E. Another way to put it is to say that in the alternative formulation above E is the instantaneous fishing mortality whereas in the original formulation E is the average fishing mortality. The natural mortality (M) can be made instantaneous and could have been included in the formulation as well, in which case the production function would be:

$$u = \frac{E}{E+M} B\left(1 - e^{E+M}\right)$$

With the same parameter values as earlier for p, c, α and β, the optimum in the deterministic case is given in Table 3. Except for the production function this simulation is equivalent to simulation 2 above. Using the low level of stochasticity, 14 observations have to be discarded due to extinction. In this case there is no significant difference between the strategies since the interval does contain zero (Table 4). In other words, using this production function improves the case for constant catch.

Conclusions

In the theoretical part of this paper it was made clear that it is not possible to prove analytically that one of the strategies considered here is better than the other, except perhaps in some special cases. However, it seems clear that if the main priority is to stabilize fishermen's income, then constant catch is most effective in this respect. This strategy is however less profitable than constant effort and it may lead to extinction in certain cases depending on the economic and biological circumstances and on the degree of uncertainty. In some cases this may call for a ban on the constant catch strategy whereas in other cases it may be sufficient to apply an adaptive attitude to management and perhaps let the catch vary within certain limits. Simulations based on a more realistic multi-cohort Beverton-Holt model show that the difference between the strategies may be smaller than suggested here and that the risk of extinction is less pronounced in the multi-cohort model than it is in the aggregated biomass model presented here, see Steinshamn (1991). In any case, the risk of extinction with this strategy is only pronounced when there is a strong spawning stock recruitment relationship, and that is not the case for many fish stocks in the world. However, in the absence of a spawning stock recruitment relationship, i.e., when recruitment is an additive stochastic variable in the recurrence function, it has been shown that with the production function assumed here constant catch is unequivocally a less profitable alternative.

A summary of the simulations (with a spawning stock recruitment relationship) may be appropriate. In three out of six simulations constant effort was favoured, two were indeterminate and in one constant catch was favoured. This indicates that there is a difference between the economic performances of the two strategies in favour of constant effort. If the degree of stochasticity is increased, this will affect the choice of strategy in two ways: First, in most cases the difference between the strategies measured by the net present value of income will increase. Secondly, this will severely increase the probability of depleting the total population when constant catch quotas are applied.

Further, in most of the simulations presented here, comparisons of the stochastic model with its deterministic equivalent indicate that the shadow price of having uncertainty is relatively small[5]. These comparisons have to be based only on simulations where 100% of the observations are retained; i.e., where there are no cases of extinction. In the simulations reported here the introduction of uncertainty does not reduce the net present value by more than at most 1.5%. The points made above indicate that the higher the level of stochasticity, the better is the case for constant effort. In many cases a high level of stochasticity combined with constant catch may be detrimental.

An overall conclusion may be that neither of the strategies analyzed above can be ruled out a priori. They may both be viable alternatives depending on the particular situation.

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[5]This may of course be a result of the risk neutrality implicitly assumed. Simulations based on a concave (logarithmic) utility function in net revenues indicate that risk aversion does not have any major impact (Steinshamn 1991).

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Appendix A

The purpose of this appendix is to show that

$$J(\bar{u}) < J(E) \text{ when } E < E^{\dagger}$$

and

$$J(\bar{u}) > J(E) \text{ when } E > E^{\dagger}.$$

E is here any constant effort level and $E^{\dagger}$ is defined such that $B_2(E^{\dagger}) = B_1$. $\bar{u}$ is the average catch caused by E, that is,

$$\bar{u}(E) = \frac{E}{2}\left[B_1 + B_2(E)\right].$$

$J(\bar{u})$ and $J(E)$ are both increasing and concave functions in E. The only two values of $E \in [0, 1]$ for which $J(\bar{u}) = J(E)$ are $E = 0$ and $E = E^{\dagger}$. The proposition made above will therefore be proved if it can be confirmed that

$$\left.\frac{\partial J(\bar{u})}{\partial E}\right|_{E=0} < \left.\frac{\partial J(E)}{\partial E}\right|_{E=0}$$

and

$$\left.\frac{\partial J(\bar{u})}{\partial E}\right|_{E=E^{\dagger}} > \left.\frac{\partial J(E)}{\partial E}\right|_{E=E^{\dagger}}$$

Since the revenue-parts of $J(\bar{u})$ and $J(E)$ are equal by definition, it is sufficient here to look at the cost-parts. The costs with constant catch are given by

$$C(\bar{u}) = \frac{c\bar{u}}{B_1} + \frac{c\bar{u}}{B_2(\bar{u})}$$

and the costs with constant effort are simply

$$C(E) = 2cE.$$

Since c is a constant it can be removed from these expressions without influencing the line of argument. It is now sufficient to show that

$$\left.\frac{\partial C(\bar{u})}{\partial E}\right|_{E=0} > \left.\frac{\partial C(E)}{\partial E}\right|_{E=0}$$

and

$$\left.\frac{\partial C(\bar{u})}{\partial E}\right|_{E=E^{\dagger}} < \left.\frac{\partial C(E)}{\partial E}\right|_{E=E^{\dagger}}$$

We have

$$\frac{\partial C(\bar{u})}{\partial E} = \frac{1}{B_1}\frac{\partial \bar{u}}{\partial E} + \frac{B_2\frac{\partial \bar{u}}{\partial E} - \bar{u}\frac{\partial B_2}{\partial E}}{B_2^2}$$

and

$$\frac{\partial C(E)}{\partial E} = 2$$

Further,

$$\frac{\partial \bar{u}}{\partial E} = \frac{B_1}{2} + \frac{sB_1}{2} - sB_1E + \frac{w}{2}$$

and

$$\frac{\partial B_2(\bar{u})}{\partial E} = -\frac{sB_1}{2} - \frac{s^2B_1}{2} + s^2B_1E - \frac{sw}{2}$$

Substituting this into the expressions above and using the facts that $\bar{u} = 0$ and $B_2 = sB_1 + w$ when $E = 0$ and $B_1 = B_2$ when $E = E^{\dagger}$, the expressions

$$\left.\frac{\partial C(\bar{u})}{\partial E}\right|_{E=0} > \left.\frac{\partial C(E)}{\partial E}\right|_{E=0}$$

and

$$\left.\frac{\partial C(\bar{u})}{\partial E}\right|_{E=E^{\dagger}} < \left.\frac{\partial C(E)}{\partial E}\right|_{E=E^{\dagger}}$$

both reduce to the same; namely

$$[sB_1 + w - B_1]^2 > 0,$$

which is obviously fulfilled.

Appendix B

This appendix gives a proof of proposition 1.

Proof

Denote the constant levels of u and E by u_f and E_f. The proposition above is equivalent to saying

$$c^2\text{var}E\left(B; u_f\right) < p^2\text{var}u\left(B; E_f\right)$$

or

$$\text{(B1)} \qquad c\sigma_E < p\sigma_u$$

where σ is the standard deviation. Since it is assumed that

$$c\overline{E} < p\bar{u}$$

(where the bar indicates expected or in this case average values), it is sufficient to show that

$$\text{(B2)} \qquad \frac{\sigma_E}{\overline{E}} < \frac{\sigma_u}{\bar{u}}.$$

The last two inequalities can then be multiplied in order to give (B1). We have

$$\frac{\sigma_E}{\overline{E}} = \left|\frac{B_1 - \left[s\left(B_1 - u_f\right) + w\right]}{B_1 + \left[s\left(B_1 - u_f\right) + w\right]}\right| = \left|\frac{B_1 - B_2\left(u_f\right)}{B_1 + B_2\left(u_f\right)}\right|$$

and

$$\frac{\sigma_u}{\bar{u}} = \left|\frac{B_1 - [sB_1(1-E_f)+w]}{B_1 + [sB_1(1-E_f)+w]}\right| = \left|\frac{B_1 - B_2(E_f)}{B_1 + B_2(E_f)}\right|$$

Condition (B2) can be rewritten

$$\text{(B3)} \qquad \left|\frac{B_1 - B_2(u_f)}{B_1 + B_2(u_f)}\right| < \left|\frac{B_1 - B_2(E_f)}{B_1 + B_2(E_f)}\right|$$

where $u_f = \frac{E_f}{2}[B_1 + sB_1(1-E_f) + w]$ in order to yield the same average catch. The function $g(B_2) = |(B_1 - B_2)/(B_1 + B_2)|$ is monotonically decreasing for $B_2 < B_1$ and monotonically increasing for $B_2 > B_1$;

$$\text{(B4)} \qquad |B_1 - B_2(u_f)| < |B_1 - B_2(E_f)|$$

is thus necessary and sufficient for (B2) (and (B1)) to be fulfilled. In (B4) both sides will have the same sign when the absolute values are not taken into account. Assuming that both sides are positive, (B4) (without absolute values) reduces to

$$\text{(B5)} \qquad E_f > \frac{w - (1-s)B_1}{sB_1} \equiv E^{\dagger}.$$

However, letting $E < E^{\dagger}$ contradicts the assumption that both sides are positive since $E^{\dagger}$ is the equilibrium level of effort, (and vice versa assuming both sides are negative). Equations (B4), (B2) and (B1) are therefore satisfied for all values of E.

Appendix C

If

$$\text{(C1)} \qquad \varepsilon\pi(\text{CC}) = \int_{B^L}^{B^U} (p - c/x)u\,dx$$

and

$$\text{(C2)} \qquad \varepsilon\pi(\text{CE}) = \int_{B^L}^{B^U} (px - c)E\,dx$$

where $u = \text{E}(B^U + B^L)/2$, then

$$\text{(C3)} \qquad \varepsilon\pi(\text{CE}) > \varepsilon\pi(\text{CC})$$

Taking the integrals, it is seen that this is equivalent to saying

$$\text{(C4)} \qquad \frac{2(B^U - B^L)}{B^U + B^L} < \ln(B^U/B^L).$$

Assume that B^L is fixed and let the left side of this inequality be a function $g(B^U)$ and the right side a function $h(B^U)$. Then it is seen that $g(B^L) = h(B^L) = 0$, and the derivatives are

$$\text{(C5)} \qquad \frac{\partial g(B^U)}{\partial B^U} = \frac{4B^L}{(B^U + B^l)^2}$$

and

$$\text{(C6)} \qquad \frac{\partial h(B^U)}{\partial B^U} = \frac{1}{B^U}.$$

The condition in Equation (C1) is fulfilled if

$$\frac{\partial h(B^U)}{\partial B^U} > \frac{\partial g(B^U)}{\partial B^U}$$

and by inserting the expressions for the derivatives above and simplifying it can be seen that this condition reduces to

$$(B^U)^2 - 2B^U B^L + (B^L)^2 > 0$$

which is always satisfied.

Bioeconomic Impacts of TAC Adjustment Strategies: A Model Applied to Northern Cod

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This paper presents a framework for the development and evaluation of alternative total allowable catch (TAC) adjustment strategies for managing commercial fisheries to meet stock population targets. An age-structured model of the fish population is developed to examine the implications of annual TAC harvesting. Catch allocations are made for separate inshore and offshore fishing fleets and annual economic benefits and costs are modelled for both the harvesting and processing sectors. Mathematical programming is used to derive TAC schedules having value-based objective functions subject to stock targets expressed as constraints. These TAC schedules are compared to fishing mortality reference point adjustment strategies commonly used in the provision of scientific advice for Atlantic Canada groundfish stocks. Uncertainty was modelled in a dynamic Monte Carlo simulation of the biological and economic subsystems. Simulation results are used to evaluate the probability associated with policy performance in achieving key economic and stock size targets over time. The model is applied to Canadian data from the NAFO Div. 2J3KL northern cod fishery. The results show that the proposed decision framework provides improved bioeconomic TAC alternatives compared with the single valued *ad hoc* reference point strategies currently in use.

Notre étude présente un cadre d'élaboration et d'évaluation de nouvelles stratégies d'ajustement des totaux des prises admissibles (TPA) pour une gestion des pêches commerciales visant à atteindre des cibles de population pour les stocks. Nous élaborons un modèle de structure par âge de la population de poissons pour examiner les incidences du prélèvement annuel du TPA. Les allocations sont fixées séparément pour les flottilles côtières et hauturières, et les profits et les coûts économiques annuels sont modélisés pour les secteurs de la capture et de la transformation. La programmation mathématique sert à établir les TPA, avec les fonctions objectives basées sur la valeur et sujettes aux cibles fixées pour les stocks exprimées comme contraintes. Nous comparons ces séries de TPA aux stratégies d'ajustement des points de référence de la mortalité par pêche couramment utilisées pour fournir des avis scientifiques sur les stocks de poisson de fond de l'Atlantique canadien. L'incertitude est modélisée dans une simulation dynamique de Monte-Carlo portant sur les sous-systèmes biologiques et économiques. Les résultats de la simulation servent à évaluer la probabilité associée au succès des politiques par rapport aux cibles principales fixées pour le plan économique et la taille des stocks sur une période donnée. Le modèle est appliqué aux données canadiennes sur la pêche de la morue du nord dans 2J3KL. Les résultats montrent que le cadre décisionnel proposé offre de meilleures alternatives bioéconomiques au TPA par rapport aux stratégies des points de référence *ad hoc* fondées sur une seule valeur qui sont actuellement en usage.

This paper presents a framework for the development and evaluation of alternative stock adjustment strategies for a single commercially harvested fish stock. The objective is to establish a formal decision mechanism to quantify the integrated biological (stock) and economic (harvesting and processing) implications of annual total allowable catch (TAC) schedules. The TAC decision methodology is applied to the Canadian management of the northern cod stock complex in NAFO Divisions 2J, 3K and 3L. Currently TAC harvesting strategies are based on the $F_{0.1}$ biological reference point that is used in the provision of scientific advice for Atlantic Canada groundfish stocks.

Problem Definition

The problem of setting annual harvesting policies (TAC's) is primary to the management of commercial groundfish stocks in Atlantic Canada. Issues associated with this problem have been the topics of task forces and royal commissions on the northern cod fishery in particular (e.g., see Kirby 1982; Alverson et al. 1987; Harris 1989; Dunne 1990). Research into the TAC adjustment problem has focused primarily on the biological state and the projected growth potential of fish stocks. Many researchers (e.g., Brown 1989;

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Goodyear 1989; Rosenberg and Brault 1991) have presented general approaches to quantifying long-run stock targets that concentrate wholly on stock population dynamics. Others, notably Gabriel et al. (1989), Hoenig et al. (1990), and Restrepo et al. (1990) have analysed the biological uncertainty associated with setting TAC's and attempted to quantify the "risk" implications on the stock.

From a decision perspective, the TAC setting problem can be stated generally as

1. determining a set of alternative annual TAC's over an adjustment planning period, such that
2. the alternatives each satisfy prespecified conditions (e.g., on expected stock growth) and
3. that they are comparable against measurable value criteria.

In Canada, fisheries management in general and TAC strategy building in particular are guided by the mandate of the Fisheries Act of Canada to: (i) conserve fish stocks for the benefit of future generations; (ii) develop commercially viable Canadian fisheries; and (iii) distribute fish resources among Canadians who benefit from these resources (Laubstein 1987). Since 1977 and in support of the conservation mandate, extensive biological stock assessment and fisheries research have been carried out through the Canadian Atlantic Fisheries Scientific Advisory Commission (CAFSAC). Notably, however, there is no similarly empowered institutional arrangement or integrated structured mechanism that treats questions of economic viability and operations management issues related to items (ii) and (iii) above. In order to treat the varied mandates of fisheries management, an integrated bioeconomic approach is required. This paper presents one such approach related to developing and evaluating strategic TAC's.

Toward developing an integrated approach, it is necessary to articulate attainable short-term and long-term goals. In the TAC adjustment problem, stock conservation and growth is of overriding significance. Consequently, stock population targets should be viewed as constraining factors in the development of strategic harvest policy. The inclusion of biological goals or objectives expressly as constraints represents an important paradigm shift (versus biological reference point methods) in dealing with the TAC adjustment decision problem. Cunningham et al. (1985) state that "... the correct way to interpret biological considerations is as a constraint on our ability to achieve the best socioeconomic situation"(p. 107); Clark(1985) states that "biological objectives can often be expressed instead as additional constraints — for example, the constraint $X \geq X^0$ would imply that ... a stock level below X^0 must not be permitted" (p.146); and, finally, Larkin(1988) states emphatically that

> The approach [to fisheries management] must be anthropocentric. It is a contradiction to speak of biological objectives of fisheries management. Much more logical is to speak of biological constraints to management... The real questions are: what should be the biological constraints and what should be the social objectives. The answers are: whatever is necessary to preserve future biological options until we know more biology and, whatever seems appropriate to the society at the time. (p. 289)

Exactly what these "biological constraints" should be depends on (i) the individual stock's characteristics for growth, recruitment, etc., and (ii) an "appropriate" level of exploitation. For the TAC adjustment problem, such constraints should take the form of desirable stock size targets. Such targets would be measurable in the same units as stock abundance estimates from assessment procedures that could be tracked and monitored over time thereby making the process directly accountable. Necessarily, stock targets levels must at the same time incorporate assumptions about an "appropriate" level of stock exploitation. As such, stock constraints are not mere statements of physical or biological extinction but operational constraints that support the fisheries sector and socioeconomic considerations at some prescribed level.

It is noteworthy that this "anthropocentric" approach to management is not contained within the widely used conventional $F_{0.1}$ reference point approach to management. Instead, the reference point approach expresses "scientific advice" directly in the form of harvesting levels based solely on a fixed fraction of the assessed stock abundance at age. The reference point approach presents only a single option to management. Alternatively, our proposed procedure determines a set of feasible TAC harvesting schedules that explicitly embody the integrated biological and fisheries sector constraints. As a guideline for evaluating feasible TAC strategies, value-based objectives, e.g., value of catch by age, value of catch by harvesting sector, value of catch by community, etc., can be applied to rank various TAC alternatives.

The problem solving approach outlined above with its formal elements of (1) decision alternatives, (2) constraining factors, and (3) objective measures, is consistent with the scientific method of problem solving from the operational research literature (e.g., Wagner 1969). The following sections describe this procedure for dealing with the TAC adjustment problem in detail.

Methods

This section presents the biological (population dynamics) model of the stock, and the economic model of the harvesting and processing sectors.

Population Dynamics

The age-structured population dynamics model presented here is constructed using the data structure from assessment and projections procedures of CAFSAC. Model dynamics are based on well-defined relationships that have been used elsewhere (Gulland 1983; Schnute 1985; Walters 1986; Restrepo

et al. 1990; Rosenberg and Brault 1991) in modelling single stock population dynamics subject to exploitation. The model uses initial year stock data including numbers at age, average weight at age, partial recruitment, natural mortality and fishing mortality data (from the corresponding TAC schedule) to determine year-over-year stock at age distributions for (i) numbers of fish, (ii) stock biomass, and (iii) catch (weight and numbers).

The population dynamics model (equations (1) to (3) below) is determined over the annual planning horizon $t = 1, 2, \ldots, T$, where $t = 1$ denotes the initial year, and $t = T$ the final year of the planning horizon, and $a = 0, 1, 2, \ldots, A$, where $a = 0$ denotes age at birth and $a = A$ the oldest age of fish in the stock.

Equation (1) represents the stochastic version of a generalized stock-recruitment function (Schnute 1985) as the mechanism used to generate the yearly reproductive capacity of the stock.

$$(1) \qquad N_{0,t} = R_t = c\alpha S_{t-1}\,(1 - \gamma\beta S_{t-1})^{1/\gamma} \exp(z\sigma),$$

where $N_{0,t}$ denotes the numbers of births at the beginning of each year t as a function of the spawning stock biomass, and the scale, productivity, and shape parameters α, β, γ respectively of the stock-recruitment function; z is the standard normal variate and σ is the standard deviation in the log-normally distributed error term. The constant multiplicative factor c adjusts the empirical at age "recruitment to the fishery" numbers back to age 0, e.g., $c = \exp(+3M)$, for a 3 year lag in actual recruitment to fishable biomass.

Equation (2) defines the year-over-year changes in each cohort of the stock.

$$(2) \qquad N_{a+1,t+1} = N_{a,t} \exp(-M - \mathrm{PR}_a F_t)$$

The changes in the numbers of stock are a function of M, the continuous rate of natural mortality (assumed as elsewhere to be constant across all ages in the population), the partial recruitment at age, PR_a (constant over all years of the planning period), and the fishing mortality level, F_t for fully recruited ($\mathrm{PR}_a = 1$) ages of the stock. Using (2) and the expected weight at age (at the beginning of each year, and on average over each year), corresponding biomass levels for different age groups, e.g., harvestable biomass or adult spawning stock biomass, can be determined.

Equation (3) defines the catch weight at age, $C_{a,t}$ for a given level of fishing mortality by year.

$$(3)\; C_{a,t} = w_a N_{a,t} (1 - \exp[-M - \mathrm{PR}_a F_t]) \frac{\mathrm{PR}_a F_t}{M + \mathrm{PR}_a F_t}.$$

The average weight per fish at age a, w_a is determined over the entire yearly catch period. This equation also provides the basis for determining (i) catch numbers at age in a given year, and (ii) the total catch over all ages resulting from fishing mortality, F_t.

In order to generate the results of the iterative set of related equations (1)–(3) for stock population dynamics, an initial distribution of the stock numbers, $N_{a,1}$ at the beginning of the initial year is required along with either an annual schedule of (i) fishing mortalities, F_t to apply to the harvestable fraction of successive cohorts, or (ii) the desired TAC's. If the TAC schedule is provided explicitly, then corresponding fishing mortalities that yield the specified TAC's are directly determinable (Allen and Hearn 1989).

TAC Benchmark Schedule

Using equations (1)–(3), a formal model to determine the annual TAC's over the planning period was developed using mathematical programming. For illustration purposes, a "benchmark" problem was formulated to choose decision variables for the fishing mortality, F_t in order to maximize the discounted sum of annual TAC's over the planning period subject to the stock target constraints. Alternative value-based objective functions could also be considered. (Each alternative objective would generate alternative TAC strategies for further evaluation and review.)

Discounting of the annual TAC's reflects the economic value of the harvest of cod and the opportunity cost of future harvests over time to the fishery sector. Discounting quantifies the notion that fishermen and society would be willing to pay more to harvest a fish today, than to have the promise of harvesting a fish of equal value next year. In simple terms, the fish harvested today provides cash that can be used to earn interest and be consumed for the immediate welfare of society.

Formally, the objective functional for the optimization problem may be written as:

$$(4) \qquad Z = \max_{F_t} \sum_{t=1}^{T} \exp(-\delta t) \sum_{a=1}^{A} C_{a,t}\,(F_1, F_2, \ldots, F_t)$$

where $C_{a,t}$ is defined as in equation (3) and updated accordingly through (1) and (2) making (4) nonlinear in the decision variables, F_t; δ is the continuous rate of social discount of annual TAC's.

The objective function (4) is constrained by the stock target constraints over the planning period. These constraints may be defined as follows:

$$(5) \qquad \sum_{a=a_i} w_a^b N_{a,t} \geq B_t\,(a_i+),\; t \in \{1, \ldots, T\}$$

$$(6) \qquad l \leq F_t \leq u,\; t = 1, 2, \ldots, T$$

where average weight at age, w_a^b are adjusted weights for the beginning of the calendar year t. The left hand sides of constraints (5) are expressions for the biomass at key ages, a_i and older (e.g., where a_i represents lower age limits for harvestable or adult stocks). The right hand sides of these constraints, B_t are the specific biomass targets at each time t over the planning period. Since each $N_{a,t}$ is a nonlinear function of the fishing mortality, then these constraints are also

nonlinear in the decision variables, F_t. Equation (6) declares annual lower ($0 < l$) and upper ($l < u$) limits on the annual fishing mortality decision variables.

The solution to the nonlinear optimization benchmark problem provides insight into the form of the TAC schedule that yields the largest present value catch while satisfying the stock targets explicitly. It remains to consider the economic implications of biologically "feasible" alternative TAC schedules.

The Commercial Fishery Sector

The economic status of the primary (harvesting) and secondary (processing) sectors in the fishery depend on the annual TAC. Annual economic measures of comparison are calculated for each defined sector of the fishery and include (i) Net Operating Income (the difference between operating revenues and costs including Labour), (ii) Cash (the Net Operating Income less fixed costs, debt expense, and income tax payable), and (iii) Value Added (the Net Operating Income plus labour less fixed costs).

The commercial fishery is subdivided into three sectors that represent the major activities in groundfish production in Atlantic Canada. These sectors include (1) an inshore harvesting sector of fixed and mobile gear including traps, longliners, and small trawlers; (2) an offshore harvesting sector comprised primarily of large mobile otter trawlers; and (3) the processing sector comprised of plants that produce processed fish products.

The economic performance of the inshore sector depends on the realized inshore catch by age. The inshore fleet is assumed to catch up to an annual inshore allowance of the TAC. The modelled inshore catch by age in each year is distributed according to the historical proportion of catch at age distribution to the population at age.

The economic status of the offshore sector is modelled to depend on the harvest allocation remaining after the removal of the inshore catch allowance. The offshore share of the total annual TAC is determined from the residual of the total TAC less the prespecified allowance for the inshore fishery. Offshore harvesting economics then varies as a function of the residual TAC, the age distribution of the stock, and the residual pattern of offshore catch by age.

The economics of the processing sector depend on (i) the age distribution of the catch, (ii) the share of inshore and offshore total catches, and (iii) the filleting procedure used in the processing operation, i.e., manual or machine processing. The production possibilities and earnings for final products and the costs of filleting operations vary with the age of fish at harvest. Revenues and material costs tend to increase with larger, older fish, while labour costs tend to decrease with the increasing size of fish. In general, there is more economic value attributable to older fish. Annual total "Raw Material" operating costs of the processing sector equals the total landed value of all inshore and offshore harvesting.

Deterministic and Stochastic Analyses

The performance of alternative TAC schedules are evaluated by comparing the model output measures against the specified targets and economic impacts of the fishing sectors. A deterministic analysis fixes data values at "best estimate" levels for all population and economic data requirements. The results of this analysis provides "best estimates" for the output measures. These measures provide insight into the relative performance of alternative strategies. Prospective TAC schedules may then be selected for further analysis and others eliminated as viable options.

Biological output measures from the deterministic analysis reported here include annual stock specific information over the planning period: (1) the schedule of annual TAC's, (2) fishing mortalities, F_t, and (3) biomass level values corresponding to stock target measures. Economic output measures include annual schedules for each of the three sectors (inshore and offshore harvesting and processing) and all sectors totals for net operating incomes. The total annual net operating income for all sectors discounted and summed for the period 1991–2001 is also included as an aggregate point estimate of economic performance.

Selected strategies that have been reviewed in the deterministic performance analysis undergo further analysis to capture the inherent uncertainty of the system. In this stochastic analysis, probability distributions for key input variables are assigned. The system is then simulated under each selected TAC strategy subject to random fluctuations assigned from probability distributions for all variable inputs. Multiple trials are simulated to replicate possible states of the projected system. Output from the stochastic analyses include measures of central tendency (mean, median, mode), measures of variation (standard deviation, coefficient of variation, value range) and probability distributions for each of the biological and economic output measures mentioned above.

Application

2J3KL Northern Cod

Management strategies for northern cod have been developed in conjunction with the recent history of fisheries management in the North Atlantic. Halliday and Pinhorn (1985) and Pinhorn and Halliday (1990) trace the evolution of Atlantic management practice from an historical perspective. Rivard and Maguire (1993) examine the impact of the $F_{0.1}$ reference point strategy on Atlantic groundfish stocks.

In the early 1980's, the traditional Newfoundland inshore fishery was unable to harvest its "share" of the global allotted 2J3KL cod TAC. This fact, coupled with the growing predominance of the offshore northern cod fishery over the same period prompted scientists to reconsider optimistic forecasts for sustainable cod harvests that proceeded from the promise of extended jurisdiction in 1977. In 1987 the Task Group on the Newfoundland Inshore Fishery (TGNIF) (Alverson et al. 1987) reported on the high degree of uncertainty associated

with our understanding of the interrelated dynamics of the cod stock complex subject to harvesting. The task group recommended a downward adjustment in the total TAC from the mid-1980's levels in order to allow the stock to recover from a series of low recruitment years. The Harris Report (Harris 1990) extended the analysis and recommendations of TGNIF by concentrating on the impacts of continued lower than anticipated recruitment levels. The Report recommended lower harvesting levels in the short term in order to allow the mature stock to build back up to provide a higher likelihood of stronger recruitment in future years and a reduced risk of low recruitment.

The Implementation Task Force (Dunne 1990) detailed precise biomass target levels corresponding to the directives laid out in more general terms by Harris. In terms of biological goals, Dunne identified stock rebuilding as the primary concern with emphasis on specific short and long term milestones of stock performance measures related to temporal biomass size, fishing mortality levels, and age distribution of the stock. These biological goals for northern cod are given in Table 1. The planning period for setting TAC schedules in the current study encompasses the 11 years from 1991 to 2001 corresponding to the period of the temporal biological goals.

Dunne's target biomass levels are the primary constraints (condition (5)) of the mathematical programming problem. Fishing mortality levels culminating in sustained harvesting at $F_{0.1}$ levels are secondary conditions (6) for feasible TAC schedules. For northern cod, the $F_{0.1}$ reference level of fishing mortality is equal to 0.25 (Anon. 1991, p. 12). Finally, desirable TAC schedules are required to increase the average weight of fish harvested through increases in the proportion of older fish in the population.

Table 1. Biological goals for 2J3KL northern cod. Source: Dunne (1990) Implementation Task Force, p. 2.

1. Biomass Levels

Year	Age 3+ Biomass	Age 7+ Biomass
1994	1,000,000 t	450,000 t
2000	1,300,000 t	650,000 t

2. Long-term Fishing Mortality Level (F)

TAC's set so as to decrease F from current levels down to the level of $F_{0.1}$ over time and be maintained there.

3. Age Distribution

More fish in older age groups.

Population Dynamics

Initial cod numbers by age estimates were obtained from CAFSAC assessment documents (Baird et al. 1990, Anon. 1991). These provide updated estimates on age distribution numbers for ages 3 to 16 years, average weight at age, 3+ and 7+ stock biomass positions, partial recruitment at age functions, and stock recruitment assumptions. Using the stock data estimates reported for January 1989 (adjusted as per Anon. 1991, p. 12) and January 1991, an automatic spreadsheet model for stock dynamics (Lane 1993) was tested and validated against the provided stock positions over the three year period, 1989 to 1991. Other parameter values included the natural mortality, M, constant for all ages at $M = 0.20$; and the stock-recruitment function equal to the 1978–1990 geometric mean of 300 million cod at age 3 per year (Baird and Bishop 1990). (Equivalently, 300 million cod at age three equate to 240 million cod at age 4 when $M = 0.20$. This level of recruitment is approximated by a Beverton-Holt stock-recruitment function using equation (1) with parameters $\alpha = 25.0$, $\beta = 0.099$, and $\gamma = -1$.) Table 2 presents a summary of the model inputs and outputs for the test period. The results compare very favourably with the documented CAFSAC positions as summarized from Baird and Bishop (1990) and Anon. (1991) for 1989 to 1991. It is significant to note the higher stock levels for ages 3 and 4 cod in 1990. These years of high recruitment for the 1986 and 1987 cohorts have an important impact on the stock over the course of the planning period 1991 to 2001.

Economic Data

Input data for the three sectors of the cod fishery are presented below. All values estimate real 1991 prices and costs. The data are applied to each year of the planning period, 1991–2001.

1. Inshore Harvesting — The annual inshore catch of cod at age is set to the inshore allowance of 115,000 t each year. Catch at age is determined according to the historical distribution of inshore catch at age as a proportion of the population biomass at age (Baird et al. 1990, table 6, p. 20). Since the inshore catch is scaled to the constant inshore allowance each year, the expected inshore harvest landed value (and corresponding net operating income) varies only in relation to the relative catch at age distribution in each period. Table 3 provides the breakdown of inshore operating costs in dollars per dollar of landed value (operating revenue). Operating revenues are determined as a function of the catch by age.

2. Offshore Harvesting — The annual net operating income for offshore harvesting varies as a function of total TAC catch by age less the annual inshore allowance catch by age. Table 3 provides the breakdown of offshore operating costs in dollars per kilogram of cod landed. As for the inshore harvesting sector, operating revenues are determined as a function of the catch by age.

Table 2. Initial year stock age distribution parameters, 1989-1990. Values are from the QuattroPro spreadsheet model described in Lane (1993). Parameters values are based on CAFSAC stock assessment for Northern cod. Source: Baird and Bishop (1990). F values are fishing mortality rates and h values are corresponding exploitation factors for fully recruited ages.

I. Global Parameters

Parameter	Value	Description
N	2400	Initial Stock Size (millions of cod), all ages at start of period 1989
M	0.20	Continuous rate of natural mortality (all years and for all ages of cod)

II. Age Input Parameters

	Ages															
Parameters	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16
Initial Proportion (%)	18.6	32.7	21.1	10.5	4.6	4.18	4.55	2.3	0.78	0.34	0.17	0.07	0.05	0.04	0.03	0.00
Partial Recruitment (PR)	0.00	0.00	0.007	0.158	0.472	0.965	1.00	1.00	1.00	0.976	0.76	0.654	0.50	0.50	0.50	0.50
Average Weight (kg)	0.10	0.28	0.48	0.73	1.06	1.43	1.80	2.16	2.53	3.05	3.98	5.22	6.88	8.64	10.06	10.63
Begining of Year Weight (kg)	0.05	0.19	0.38	0.605	0.895	1.245	1.615	1.98	2.345	2.79	3.515	4.5	6.05	7.75	9.35	10.345

III. Start of Year Numbers

	Ages															
Year	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16
1989	446.9	784.8	505.7	252.0	110.4	100.3	109.2	55.2	18.7	8.2	4.1	1.7	1.2	0.96	0.72	0.0
1990	447.5	365.9	642.5	412.4	189.4	70.0	48.7	52.0	26.3	8.9	3.9	2.2	1.0	0.75	0.60	0.45

IV. Initial Year Values

Year	F	h	TAC (000's t)
1989	0.542	0.3826	252.832
1990	0.518	0.3696	219.067

Table 3. Operating cost components for the inshore and offshore harvesting sectors. Inshore costs are estimated as a proportion of landed value in dollars; offshore costs are estimated as a function of dollars per landed weight in kg. Source: Economic Planning and Policy, Department of Fisheries and Oceans, Ottawa.

Components	INSHORE SECTOR Dollars per Landed Value	OFFSHORE SECTOR Dollars per Landed Weight (kg)
OPERATING REVENUE	1.000	N/A
Fishing Effort Operating Costs (Ice, Provisions, Fuel, Bait)	0.119	0.110
Repair and Maintenance	0.028	0.058
Gear/Electronics	0.140	0.030
Labour (Captain and Crew Shares)	0.224	0.304
TOTAL OPERATING COST	\$0.511/\$ LV	\$0.502/kg
NET OPERATING INCOME	\$0.489/\$ LV	N/A

3. Processing Sector — Revenues and costs from processing cod depend on the size/age of the landings. Based on average size at age, it is assumed that only cod ages 3 to 10 years may be filleted. Among these landings, 40% of all inshore catch aged 3 to 10 years is filleted by machine. For the offshore sector this figure is 50%. All other cod aged 3 to 10 years are filleted manually. Table 4 provides the breakdown by age categories of operating revenues and costs for machine filleted processing operations and manually filleted operations as provided in Anon. (1990) and Anon. (1989b). Processing sector "Raw Material" costs equate to total inshore and offshore harvesting operating revenues (landed values).

Scenario Development

For purposes of comparison, two TAC schedules are developed for northern cod over the period 1991 to 2001. These include a "Base Case" schedule based on the $F_{0.1}$ reference point, and a "Benchmark" schedule developed from the solution to the mathematical programming problem described in equations (4)–(6).

Base Case

The 1991 Canadian Management Plan for northern cod in 2J3KL (Anon. 1991, p. 13) recommended TAC's for 1991 to 1993. This partial schedule (over the planning period of 1991–2001) forms the basis of the Base Case TAC scenario. Annual TAC's for the remaining years of the planning period are determined using the policy implied by the biological goals (Table 1). Specifically, the remaining TAC's in the schedule to 2001 are based on the $F_{0.1}$ fishing mortality for cod. TAC's for the years 1994 to 2001 are calculated at the same $F = 0.25$ level until the end of the planning horizon.

The Base Case strategy actually exploits the stock below the $F_{0.1} = 0.25$ reference level in 1993 ($F = 0.225$). Thereafter, exploitation at $F = 0.25$ actually raises the annual TAC to 195,000 t (or greater) each year. Moreover, the gap between the Base Case 1993 TAC and the $F = 0.25$ TAC in 1994 is over 25,000 t — a large single year-over-year adjustment. Analysis of the Base Case TAC schedule and biological targets suggests that the high level of adults entering the fishery in 1993 and 1994 tend to support the fishery over the remainder of the planning period. Moreover, there may be more of an opportunity for increased levels of harvesting of older stock in the short run prior to the full entry of the large 1986 and 1987 year classes into the fishery. The resulting Base Case TAC schedule for the entire planning period, 1991 to 2001 is presented in Table 5 along with output measures for the deterministic analysis.

Benchmark

The benchmark TAC schedule problem of equations (4)–(6) for northern cod contains 16 year classes and 11 years in the planning period. The solutions were obtained using GAMS/MINOS software on a PC/AT microcomputer (Brooke et al. 1988). The original "optimal" TAC schedule from (4)–(6) assigns a very large TAC to 1992 (over 300,000 t). Accordingly, the 3+ biomass falls slightly from 1.15 million t at the beginning of 1992 to just over 1 million t in 1993. Feasibility is maintained, however, since the 1994 biomass targets are exceeded (3+ biomass exceeds 1 million MT, and 7+ biomass is over 500 thousand t). The strategy proceeds from 1994 through a series of decreasing TAC's (some below $F_{0.1}$) reaching a minimum TAC of 165,000 t in 1999 to meet the year 2000 biomass targets for 3+ and 7+ cod, as required. TAC's for 2000 and 2001 increase toward the long run average harvest that approaches 200,000 t (as for the Base Case).

The benchmark TAC schedule projects the current (deterministic) status of the stock out to 2000, prior to determining the appropriate TAC adjustment strategy. The method establishes that for the problem at hand, the 3+ year 2000 biomass target of 1.3 million t is the most restrictive or "binding" constraint. Consequently, the ongoing status of the large 1986 and 1987 year classes are fundamental to meeting the target. Conversely, the harvestable stock (ages 5+) early in the horizon (1991-1993) contribute little to realizing this long run

Table 4. Processing sector operating cost components for machine filleted and manual filleted operations. Processing costs are estimated by size or age groupings of landed cod. All values are in dollars per kg processed. "Raw Material" costs are equivalent to landed value prices paid to harvesters. Source: Anon. (1990).

ITEM	AGES						
	3	4	5	6–7	8–10	11–12	13–14
	MACHINE FILLETED OPERATIONS						
Operating Revenue	1.514	1.526	1.543	1.587	1.587	—	—
Expenses Items: Labour	0.488	0.348	0.273	0.215	0.177	—	—
Raw Material	0.274	0.522	0.678	0.678	0.717	—	—
Packaging	0.026	0.030	0.037	0.038	0.038	—	—
Variable Overhead	0.154	0.110	0.086	0.068	0.056	—	—
Total Operating Cost	0.942	1.010	1.074	1.000	1.988	—	—
Net Operating Income	0.572	0.516	0.470	0.587	0.599	—	—
	MANUAL FILLETED OPERATIONS						
Operating Revenue	1.453	1.465	1.481	1.523	1.523	1.523	1.523
Expenses Items: Labour	0.528	0.377	0.302	0.244	0.208	0.186	0.165
Raw Material	0.274	0.522	0.678	0.678	0.717	0.717	0.717
Packaging	0.025	0.029	0.035	0.037	0.037	0.037	0.037
Variable Overhead	0.166	0.119	0.095	0.077	0.066	0.059	0.052
Total Operating Cost	0.994	1.047	1.111	1.036	1.028	0.999	0.971
Net Operating Income	0.459	0.417	0.370	0.487	0.495	0.524	0.552

constraint. As a consequence, benchmark TAC's are higher early in the planning period when the 1994 biomass targets are not binding due to the large 1986 and 1987 cohorts. As the system begins to stabilize under projected constant average recruitment (and the relative strength of the 1986 and 1987 cohorts decreases), the harvest schedule of the benchmark approaches the same approximate equilibrium result as the Base Case by the end of the planning period.

The benchmark strategy takes advantage of the "surplus" and aging stock that exists prior to the "arrival" of the 1986 and 1987 year classes to become part of the spawning stock biomass. This strategy is counter to the proposed Base Case management plan for 1991–1993 that is characterized by a monotonically decreasing exploitation schedule. However, the formal optimization model (4)–(6) does not explicitly impose restrictions on the year over year changes in annual TAC's.

In consideration of the above, a "modified benchmark" TAC strategy is proposed that explicitly accounts for year-over-year TAC adjustment costs (through additional constraints on (4)–(6)). This case attempts to capture the characteristic pattern of "optimal" adjustment while minimizing the large shifts in the TAC's from year to year. The result is a feasible (i.e., meets all biological targets) TAC schedule that smooths out as much as possible the large year over year changes in the optimal model solution. For the modified benchmark schedule, the 1992 TAC increases (relative to the 1991 TAC) by 6,000 t from 219,000 t (1990 and 1991) to 225,000 t (1992). Thereafter, the TAC's from 1993 to 2001 vary in the narrow range between 180,000 t and 200,000 t.

Sensitivity analysis of the characteristic solution of the benchmark problem due the value of the discount rate show the primacy of the stock target constraints in this depressed stock state. The benchmark solution for cod depends on the initial state of the stock and the binding 3+ biomass target at year 2000 but not on the specific value of the discount rate in this case.

The modified benchmark TAC's schedule is presented in Table 6 (details of the year by year age structure of the stock and economic performance measures) under this scenario. Figure 1 graphs and contrasts the time series of annual TAC's for the Base Case and the Benchmark case.

Comparative Results — Deterministic Analysis

Population Dynamics

Both TAC schedules exhibit a decrease in the fishing mortality in the initial years. The average weight per cod at capture is projected to move from 1.35kg to approximately 1.90kg over the planning period. The cod population biomass is expected to rise from current levels to meet the biological milestones for 3+ and 7+ biomass at 1994 and 2000. The Base Case provides a more rapid approach to higher sustained 3+ biomass (as a consequence of lower initial year TAC's) while the benchmark schedule has a smoother biomass growth rate over the planning period (Figs. 2 and 3). For the Base Case, projection to the year 2000 target for 3+ biomass of 1,300,000 t is nearly achieved by 1994-95 with little growth thereafter out to 2001.

Deterministic analysis of the long run equilibrium position of this stock under the constant recruitment scenario is sensi-

Table 6. Benchmark Case: Deterministic Analysis Biological and Economic Outputs. Actual yearly catches by the inshore sector vary between 112,000 t and 115,000 t according to (i) the availability of stock by age, and (ii) the historical proportional catch by age of this sector. The constant inshore catch results in the nearly constant results for inshore net operating income of approximately $38 million annually.

Year	Offshore Harvest (000's t)	TAC's* (000's t)	Fishing Mortality F	Biomass		Net Operating Income		
				Age 3+ (000's t)	Age 7+ (000's t)	Inshore*	Offshore	Process
1989	137.8	253	0.542	964	396	37.8	25.8	129.2
1990	104.1	219	0.518	1064	314	37.3	19.8	97.9
1991	103.9	219	0.446	1105	245	37.1	19.0	100.0
1992	110.2	225	0.364	1141	264	37.8	20.2	103.5
1993	85.5	200	0.276	1155	380	37.8	15.9	90.8
1994	77.1	190	0.256	1192	573	37.5	14.7	86.8
1995	69.0	180	0.238	1214	586	37.3	13.7	81.9
1996	73.4	185	0.240	1233	607	37.4	14.6	82.5
1997	73.4	185	0.238	1256	626	37.5	14.6	74.7
1998	77.6	190	0.245	1275	647	37.7	15.4	66.9
1999	81.0	194	0.250	1298	663	37.9	16.1	69.6
2000	81.9	195	0.250	1316	682	37.8	16.3	68.9
2001	83.9	197	0.250	1327	699	37.8	16.7	68.4

* Annual inshore harvest totals are set at the inshore allowance level of 115,000 t.

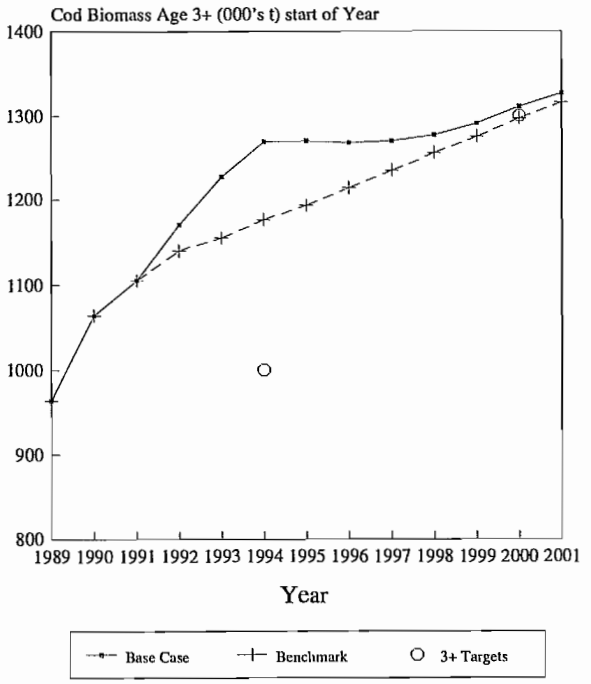


Fig. 2. Annual ages 3+ Northern cod biomass, 1989-2001 for the two alternative TAC strategies: (i) Base case, and (ii) Benchmark from the deterministic analysis. Dunne's stock target levels for 1994 and 2000 are noted.

and stock assessments are not necessarily proof of a healthy stock nor grounds for increased levels of exploitation for the medium to long run. If there are successive years of low recruitment following then a strategic view would actually lead to reduced exploitation in the short run in anticipation of lower stock levels to follow. This was precisely the situation in the Northern cod stock from 1986 to 1988 when TAC's remained high (over 250,000 t) on the strength of catch and survey results. However, when the low recruitment years of 1982–1984 began to appear in the fishery after 1989, it precipitated drastic adjustment in TAC's. The current situation that has led to the closure of the northern cod fishery in 1992 is in reaction to the observed low state of abundance of the now adult stock from these low recruitment years.

Curiously, the 1986 and 1987 cohorts that will be adult and prime market size at the projected end of the moratorium in 1994 will give a large boost to cod biomass levels. This is also evident from the large jump in the Base Case projections of TAC's at $F_{0.1}$ in 1994 (Table 5). If these large cohorts are followed by substantial follow-up recruitment then there would be some justification for returning to pre-moratorium catch levels almost immediately. However, as it appears from preliminary data, recruitment after 1987 does not appear to be good. Accordingly, and consistent with the results of the characteristic strategic planning approach, we must be more conservative in establishing TAC's in the initial years after the cod moratorium.

As noted previously, preliminary analysis of the projected assessed cod stock in 1994 and the emergence of the two large year classes would lead $F_{0.1} = 0.25$ reference point managers to proclaim TAC's exceeding 200,000 t in 1994. This myopic view ignores problems in later years that would doubtless reoccur given the apparent poor follow-up recruitment that has been estimated thus far. The hard lessons from the late 1980's and the crisis situation that precipitated in the 1990's is that a strategic, forward-looking and stock dependent plan

Table 5. Base Case: Deterministic Analysis Biological and Economic Outputs. Actual yearly catches by the inshore sector vary between 112,000 t and 115,000 t according to (i) the availability of stock by age, and (ii) the historical proportional catch by age of this sector. The constant inshore catch results in the nearly constant results for inshore net operating income of approximately $38 million annually.

Year	Offshore Harvest (000's t)	TAC's* (000's t)	Fishing Mortality F	Biomass		Net Operating Income		
				Age 3+ (000's t)	Age 7+ (000's t)	Inshore*	Offshore	Process
1989	137.8	253	0.542	964	396	37.8	25.8	129.2
1990	104.1	219	0.518	1064	314	37.3	19.8	97.9
1991	75.3	190	0.379	1105	245	37.1	14.2	83.2
1992	70.3	185	0.280	1170	281	37.8	13.1	82.0
1993	68.5	180	0.225	1227	432	36.9	12.7	81.7
1994	90.9	206	0.250	1269	658	38.1	17.4	96.8
1995	89.7	204	0.250	1270	661	38.4	17.9	94.2
1996	88.1	202	0.250	1268	661	38.2	17.6	89.2
1997	85.7	199	0.250	1270	663	38.1	17.1	78.6
1998	82.7	196	0.250	1277	669	37.9	16.5	65.8
1999	81.9	195	0.250	1290	678	37.9	16.3	67.7
2000	82.8	196	0.250	1311	694	37.8	16.5	68.0
2001	84.7	198	0.250	1327	710	37.8	16.9	68.0

* Annual inshore harvest totals are set at the inshore allowance level of 115,000 t.

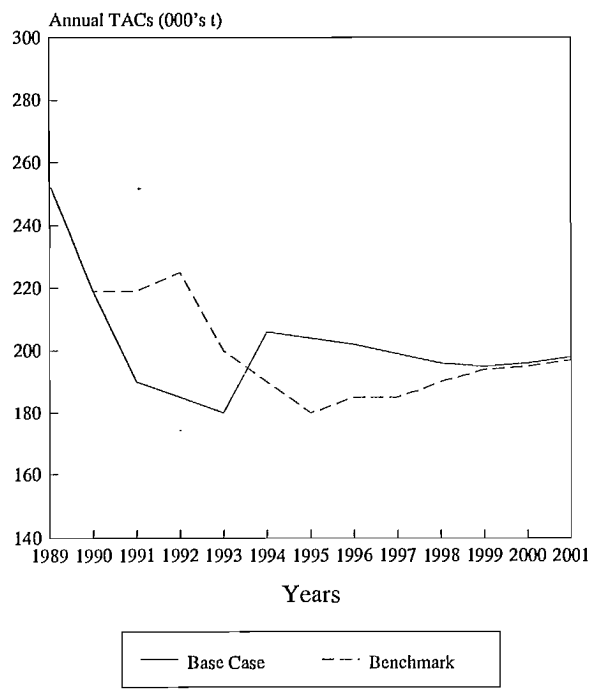


Fig. 1. Annual TAC schedules, 1989–2001 for two alternative strategies: (i) Base case, and (ii) Benchmark.

tive to the number of year classes specified for the population (e.g., truncation at 16, 17, or 20 years of age for the oldest cod). For oldest age at 16 years, the long run equilibrium for fishing at $F_{0.1}$ is actually under the 1.3 million t target for 3+ stock (1.27 million t), and just over the 650 thousand t target for 7+ spawning biomass (655 thousand t). These differences may suggest an inherent problem with the initial setting of these stock targets (i.e., the incompatibility of the 3+ and 7+ targets vis-á-vis the stock's long run equilibrium position), or may arise simply from differences in data assumptions. These observations do serve, however, to point out the dependencies in the stock dynamics as affect the setting of stock targets. Cod spawning stock biomass (age 7+) grows dramatically from 1993 to 1994 in both cases as a result of the large 1987 cohort maturing in 1994. For the Base Case, the 7+ biomass in 1994 approaches the year 2000 target of 650,000 t as a result of this large 1987 year class. It is also noteworthy that the growth in the 7+ biomass after the 1994 jump is slower for this case.

In summary, the deterministic population dynamics of 2J3KL suggest that the conservative Base Case strategy that considers only decreasing TAC's in the short run is not the only means of reaching the specified biomass targets. Other characterizations can be presented whereby dynamic targets are similarly attainable over time by other than "most rapid" approach paths. Moreover, economic implications may not support leaving potential stock surplus in the water when that surplus is made possible by strong year classes that will support the future of the fishery in the medium to long term. Finally, under the somewhat artificial situation of constant annual recruitment, the long run dynamics of the stock are independent of feasible TAC schedules since the biomass targets are close to the equilibrium stock positions. In the long run, feasible TAC schedules under the $F_{0.1}$ level of harvesting, and constant recruitment of 300 million age 3 cod per year, approaches 200,000 t per year. This is the case for both TAC schedules presented here.

Conversely, the presence of large year classes fully recruited to the fishery and evident in commercial catches

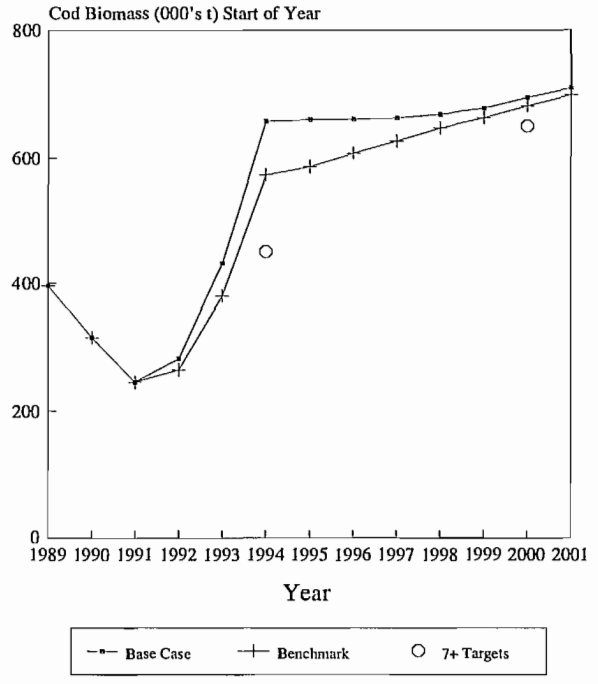


Fig. 3. Annual ages 7+ Northern cod biomass, 1989–2001 for the two alternative TAC strategies: (i) Base case, and (ii) Benchmark from the deterministic analysis. Dunne's spawning stock target levels for 1994 and 2000 are noted.

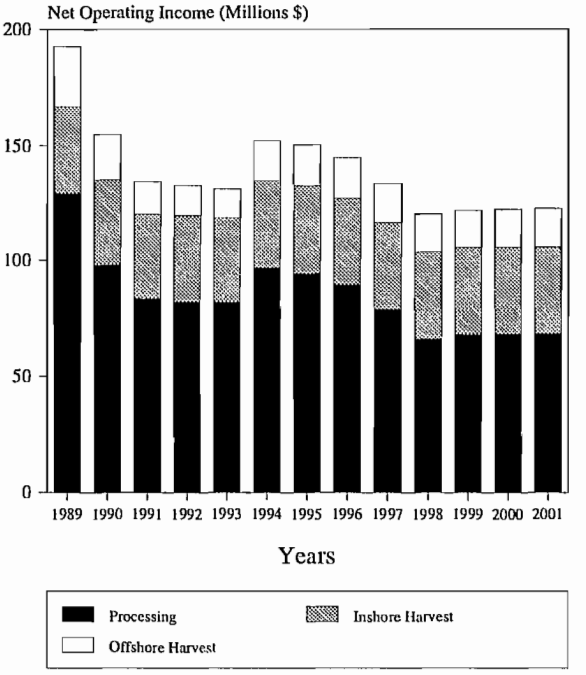


Fig. 4. Annual net operating income for the Base Case TAC strategy for each economic sector: processing, inshore and offshore harvesting.

is required for TAC adjustment. This plan must also incorporate scientific and management accountability associated with explicitly moving a stock to a target position over time.

Economic Impacts

The deterministic results of the economic analysis reflect the differences in the alternative TAC schedules. The results of the "deterministic" economic analysis for the Base Case and the Benchmark schedules are presented in detail in Tables 5 and 6. Figure 4 gives the graphic of the yearly net operating income for the Base Case. The time trend for the year end cash position and the yearly value added are very similar, i.e., a slight decline in economic performance over the years 1991 to 1993, a jump of approximately 15% in 1994 (following the rise in the TAC this year), declining from there until 1998 (at levels slightly below 1991–1993 performance), and remaining constant from 1998 to the end of the planning period.

In Fig. 5 the economic performance of the Benchmark is compared to the Base Case results for the discounted differences in yearly net operating income. (Similar differences exist between year end cash and value added measures.) As expected, the inshore harvesting sector shows little difference between the two cases as a consequence of the economic benefits from the inshore allowance of 115,000 t that is assumed to be caught each year independent of the TAC strategy. Differences between cases by sector are accounted for by the differences in the TAC schedules by year and the age distribution of the stock. As before, significant differences are noted especially in the potential processing net income due to the alternative TAC schedules.

The discounted results over the short term accentuate the economic benefits of the TAC alternative with higher relative initial exploitation. The long run discounted results tend to balance out the Base Case with slightly higher TAC's in later years over the planning period. However, the total discounted results favour the Benchmark economic benefits over the Base Case by approximately +11% in the short run (1991–1993), and +2.0% ($20 million) over the planning period.

Comparative Results — Stochastic Analysis

The nature of fisheries management, including the unobservability of the exploited stock, exogenous sources of error due to environmental disturbances and data measurement difficulties, require an assessment of the inherent uncertainties associated with any management plan. Analysis of uncertainty typically involves describing the key sources of random fluctuations in model inputs and quantification of the impact these fluctuations make on model outputs. This is carried out in a stochastic simulation analysis of the population dynamics and economic impacts models for cod. Inherent "risk" of each

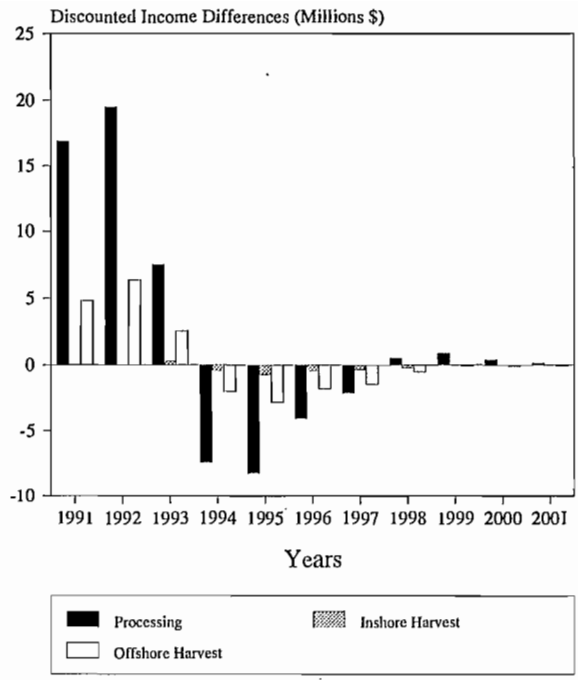


Fig. 5. Annual differences in net operating income between the Base Case TAC strategy and the Benchmark strategy results for each economic sector: processing, inshore and offshore harvesting. Values above zero denote cases where the discounted net operating income from the Benchmark strategy exceeds that of the Base Case.

strategy can be measured by examining whether observations generated by the simulation model have satisfied the specified target levels. Details of the stochastic analysis for the Base Case and the Benchmark are presented below.

Random Inputs

For the purposes of this analysis, and as done elsewhere (Rice and Evans 1988; Hoenig et al. 1990; Restrepo et al. 1990; Rosenberg and Brault 1991), randomness in cod population dynamics are assumed to occur for the following model inputs:

1. Natural mortality — the continuous rate of natural mortality, M is assumed to vary uniformly between 0.15 and 0.25 in each year. The expected value of M equals the deterministic value for $M = 0.20$.

2. Stock-recruitment — the Beverton-Holt function for the historical mean (300 million age 3 cod per year) is assumed to have many, statistically independent sources of random fluctuations. This suggests that a lognormal error term (equation (1)) would be appropriate. In this analysis, the lognormally distributed error has a normal parameter having mean 0 and standard deviation of $\sigma = 0.25$. The mean error term is $\exp(\sigma^2/2)$ or 1.032 with 95% interval factors are on the error term, $\exp(z\sigma)$ of (.61, 1.63). Thus for a given spawning stock biomass, the expected recruitment is approximately 310 million age 3 cod with likely range between 180 and 490 million age 3 cod per year for the historical mean recruitment case.

3. Initial population size — the simulation model is initialized by setting the total numbers and distribution of the 1989 cod stock by age (Table 2). In this analysis, the total number of cod across all age classes (1 to 16 years) in 1989 is assumed to be normally distributed with mean equal to 2400 million fish and standard deviation of 400 million fish. Thus, in the simulation model, the estimated numbers of the 1989 cod stock (ages 1 to 16) may vary between 1600 million fish and 3200 million fish 95% of the time. Given the total numbers, the distribution of cod by age is assigned according to the estimated percentage of cod at age in 1989. These values are recorded in Table 2. The values for the 1989 stock and the resulting age distribution affect the age distributions in future years over the planning period.

Simulation Model

Each trial of the simulation model is initialized at the start of year 1989. The dynamics of the first two initial years, 1989 and 1990, are generated for annual random selections of M, and age 4 recruitment. The 1989 and 1990 TAC's are fixed at actual values 253,000 t and 219,000 t. These results provide the initialization of the stock by age distribution for 1991, the first year of the planning period. Thereafter, the generation of random natural mortality and recruitment combined with the selected TAC schedule determine the yearly distributions of stock at age out to 2001.

Each simulation scenario is comprised of one hundred 11 year "trials". Each simulation adopts a TAC schedule from either Table 5 (Base Case) or 6 (Benchmark) and subjects the population to annual fluctuations according to random realizations from the probability distributions for the model inputs described above. The performance measures for each trial are recorded and statistics calculated over all 100 complete trials of the simulations for each of the two TAC schedules. (It is noted that simulated results also yielded trials for which stock abundance were reduced below the point where the specified TAC's could be taken in each period. Such cases were ignored and reinitialized. Consequently, summary statistics are slightly positively biased. This accounts for observed differences in expected values and simulated means for some output measures.) The results of the simulation analyses are described below.

Population Dynamics

Summary biological results of each simulation include annual descriptive statistics for 3+ and 7+ biomass values at year 2000. Summary results for the cases are presented in Fig. 6

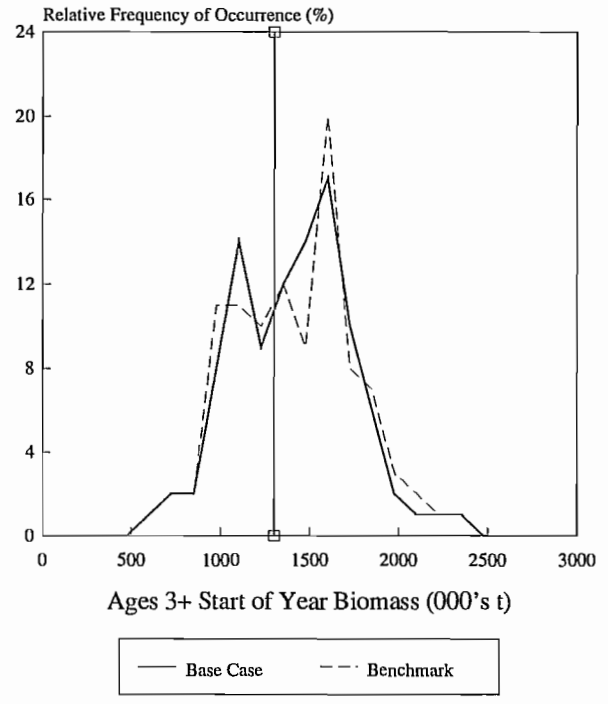


Fig. 6. Probability distributions of ages 3+ cod biomass at the year 2000 from the results of the simulation analysis from 100 independent trials for the Base Case and Benchmark strategies. Vertical line indicates stock target (Table 1).

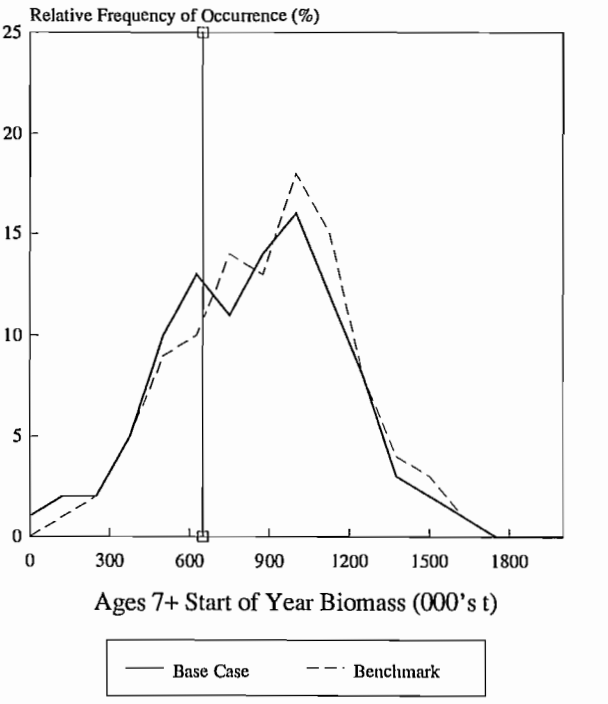


Fig. 7. Probability distributions of ages 7+ spawning stock cod biomass at the year 2000 from the results of the simulation analysis from 100 independent trials for the Base Case and Benchmark strategies. Vertical line indicates stock target (Table 1).

and 7 for cod biomass. Figure 6 shows the relative frequency distribution for 100 trials of ages 3+ cod biomass at year 2000 for the Base Case and Benchmark, respectively. Similarly, Fig. 7 shows the relative frequency distribution for 100 trials of ages 7+ cod (spawning stock) biomass at year 2000.

As for the deterministic results, expected biomass levels exceed the specified targets levels at 1994 and 2000. However, from Fig. 6 and Table 7, it can be noted that of the 100 trials in the simulation of both TAC strategies, 39 out of the 100 trials were observed to be below the 3+ biomass target in year 2000 (1,300,000 t). Similarly comparable measures of riskiness were found for the year 2000 ages 7+ spawning stock biomass (37% of observations below the 650,000 t target for both cases). The extent of riskiness associated with both strategies reflect the binding nature of the ages 3+ and 7+ stock targets as being set close to (or even slightly above for ages 3+, as suggested by equilibrium analyses) the long run capacity of the stock under the assumptions of $F_{0.1}$ harvesting levels, and constant recruitment. A summary of statistical measures for ages 3+ and 7+ biomass measures of the population dynamics are found in Table 7.

Economic Impacts

The economic results of the stochastic analysis mirrored closely those for the deterministic economic analysis. Not surprisingly, the dependence on the unchanging TAC in the face of random fluctuations in the stock, leave the economic outputs relatively unaffected. The stochastic inputs are solely in the population dynamics, i.e, M, initial stock numbers, and annual stock-recruitment. The economic impacts simulation model maintained all economic parameters, e.g., for unit prices and costs, at their deterministic values. This enables a direct attribution of economic value to alternative TAC strategies on the basis of stock effects alone.

Differences in the stochastic analysis are the results of changes in the age distribution of the stock over time. Table 7 presents the statistical summary of the stochastic analysis economic output for total 1991 to 2001 discounted (to 1991 at 10% per annum) net operating incomes of the three sectors: inshore harvesting, offshore harvesting and processing, and the all sectors total. Figure 8 presents the relative frequency distributions for 100 trials of the simulation model for total discounted net operating income by sector. The narrow range of the distributions and the low standard deviation values are indicative of the dependence of economic benefits on the TAC schedule. Constant prices and costs throughout permits the variability of the results to be isolated on the stock effects due primarily to changing age distributions over time within and across each TAC strategy. As these results indicate, there is a slight (approximately 2%) additional economic benefit to the Benchmark strategy versus the Base Case over all sectors.

Table 7. Simulation analysis summary of results for the Base Case and the Benchmark TAC strategy alternatives. The t-test values are for null hypotheses, H_0 that ages 3+ and 7+ mean biomass values in year 2000 equal the Dunne stock targets, versus alternative hypotheses H_A that the mean exceed the targets.

Base Case							
Item	Mean	Standard Deviation	95% CI on mean	Range Minimum	Range Maximum	t-test on mean	% Below Target
Ages 3+ Biomass, Start of Year 2000 (000's t)	1377	373	1303–1451	514	2572	2.07 $H_0 : \mu = 1300$	39
Ages 7+ Biomass, Start of Year 2000 (000's t)	747	336	680–814	28	1952	2.90 $H_0 : \mu = 650$	37
Total Discounted NOI - Inshore (million $)	264.0	1.2	263.7–264.2	259	266		
Total Discounted NOI - Offshore (million $)	109.3	3.8	108.6–110.1	96	115		
Total Discounted NOI - Processing (million $)	705.8	8.4	704.1–707.4	676	717		
Total Discounted NOI - All Sectors (million $)	1079	13.4	1076–1082	1031	1098		

Benchmark Case							
Item	Mean	Standard Deviation	95% CI on mean	Range Minimum	Range Maximum	t-test on mean	% Below Target
Ages 3+ Biomass, Start of Year 2000 (000's t)	1399	365	1326–1471	550	2560	2.70 $H_0 : \mu = 1300$	39
Ages 7+ Biomass, Start of Year 2000 (000's t)	759	323	695–823	87	1941	3.38 $H_0 : \mu = 650$	36
Total Discounted NOI - Inshore (million $)	263.0	1.0	262.8–263.2	259	265		
Total Discounted NOI - Offshore (million $)	114.2	4.1	113.4–115.0	100	119		
Total Discounted NOI Processing (million $)	720.0	8.7	718.3–721.8	689	732		
Total Discounted NOI - All Sectors (million $)	1097	13.6	1094–1100	1048	1117		

This gain is attributable to the offshore (+5% or $5 million) and processing sectors (+2% or $15 million) and amounts to a total expected present value gain of $20 million 1991 dollars.

Discussion

This paper presents a decision framework for the development and evaluation of alternative TAC strategies for a single commercial exploited groundfish stock in Atlantic Canada. A model is described for the current status of the stock using assessment data, and population dynamics under commercial exploitation compatible with the existing biological modelling approach of CAFSAC. A benchmark procedure for determining TAC's is established using a mathematical programming model that incorporates the notion of the value of catch over time into an objective function while explicitly accounting for stock targets as constraints in the problem formulation. From the benchmark solution, alternative TAC scenarios can be developed and compared that take into account other important considerations such as year over year TAC adjustments. Comparison of the performance of alternative TAC strategies results in improved strategies that have desirable biological and economic characteristics.

The results of deterministic analyses further refines the selection of alternative decision options for the TAC schedule. Deterministic analyses are followed by an analysis of the inherent uncertainties of the system. A simulation model is developed to incorporate simultaneous stochastic changes in system inputs and outputs. The model generates probability distributions for key biological and economic output variables.

The decision framework is applied to the pre-moratorium period for the 2J3KL northern cod fishery. This application is used to demonstrate the evaluation methodology and to provide rationale for developing and comparing alternative strategies. Although fundamentally different in method and resulting schedules, both cases analysed performed equally well with respect to achieving the medium and long term stock targets from a deterministic analysis point of view, and with similar probability distributions for the stock size. On the basis of the economic results, however, the Benchmark represents an improvement over the Base Case. The inclusion

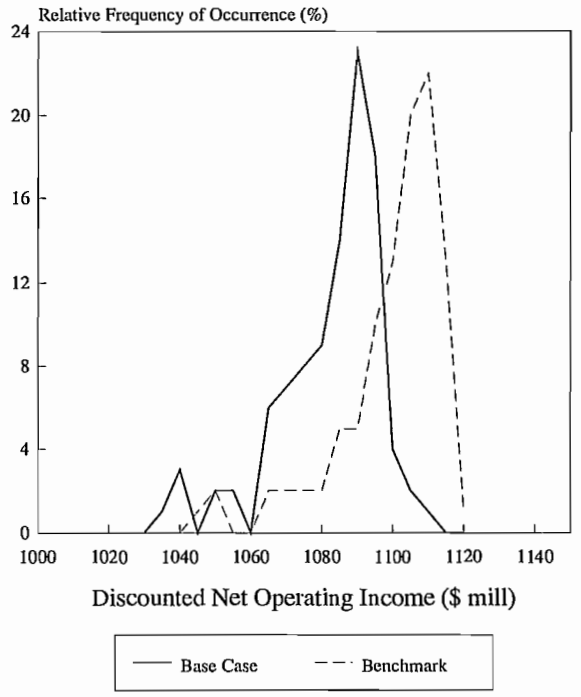


Fig. 8. Probability distributions of total discounted net operating income for all sectors over the planning period. Results are shown for the Base Case and the Benchmark alternative TAC strategies.

of year-over-year TAC adjustment costs would accentuate the value benefit of the Benchmark case due to its smoother adjustment pattern versus the Base Case.

The social, economic, and political importance of the northern cod fishery make it an ideal case to illustrate shortcomings in the current methodology for presenting "scientific advice" for fisheries management. The $F_{0.1}$-based TAC strategy does not provide decision alternatives. As an artifact of historical analyses, the domination of the $F_{0.1}$ reference point strategy preempts all other management options, in particular, those options that explicitly take into account the current status of the resource (including follow-up recruitment levels) and the socioeconomics of the fishing industry. In place of standardized reference point methods what is proposed here is a methodology that allows for the wider development and evaluation of alternative TAC options. Such a methodology recognizes the need to define the current situation, and measurable short and long term objectives. The reference point approach is viewed optionally as a short term and a long run strategy (Rivard and Maguire 1993). This conveniently avoids the need to quantify specific stock targets that could otherwise be monitored and to which scientific advice could be held directly accountable over time. This argument supports the establishment of explicit biomass targets for commercial stocks as have been postulated and used for northern cod (Dunne 1990).

The proposed methodology also acknowledges that there may be different means of achieving specified objectives over time. Indeed, the two strategies developed and analysed in this paper for northern cod may be seen as two extreme means of achieving the same stock targets in the long run. It follows that many other acceptable TAC schedules may exist "between" these two strategies. A scientific approach to problem solving would require that all such alternatives be identified and evaluated accordingly. Only in this manner can a complete picture of the TAC adjustment problem be developed toward providing a valuable support system for decision making in fisheries management.

Recent experience from the 1991 2J3KL northern cod fishery emphasizes the uncertainties affecting management measures. Unexpected difficulties with environmental conditions (including low water temperature and ice coverage) seem to have induced behavioral changes in cod that ultimately affected harvesting activities over the course of the season. The disequilibrium caused by these random events would seem to favour more stock dependent management policy making in contrast to the standardized, reference point strategies.

Despite the obvious socioeconomic hardships, the current period of closed fishing for cod in 2J3KL can also be seen as a period of great opportunity — not so much to investigate why the cod population crisis has occurred, but rather how from a management perspective, this tragedy can be avoided in the future. The message in this analysis is that fisheries management and the TAC setting process in particular must become an integrated, forward-looking planning exercise involved in moving the resource explicitly in a desired direction for the conservation of the stock as well as for the benefit of the fisheries sector.

Acknowledgements

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Experimental Management Programs for Two Rockfish Stocks off British Columbia, Canada.

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Leaman, B. M. and R. D. Stanley. 1993. Experimental management programs for two rockfish stocks off British Columbia, Canada. p. 403–418. *In* S. J. Smith, J. J. Hunt and D. Rivard [ed.] Risk evaluation and biological reference points for fisheries management. Can. Spec. Publ. Fish. Aquat. Sci. 120.

Experimental management programs for two Pacific ocean perch (*Sebastes alutus*) stocks off British Columbia were initiated in the early 1980's. One program involved a 5 yr period of specified overfishing on a stock off southwest Vancouver Island, and the other a 5 yr period of unspecified overfishing, followed by an equivalent period of closure, in the Langara Spit region of Dixon Entrance. Both of these experiments were designed and conducted with commercial industry participation. Their purposes were to test hypotheses about stock dynamics, biomass and productivity estimates, ageing methods, and the values of population parameters. The Vancouver Island experiment was conducted from 1980–1984 and was intended to harvest at ≈ 160% of the estimated sustainable yield. Field surveys, commercial fishery statistics, and biological characteristics of the stock all indicate that a large decline in biomass resulted from the experimental overfishing. The Langara Spit experiment began in the fall of 1983 and has continued to date. Commercial fishery statistics, declining participation and fishing effort, and biological characteristics also indicate reduced biomass in association with the unrestricted fishery. The fishery is now based almost entirely on annual recruitment, and the relatively strong 1976 cohort has been reduced to a subordinate status. Results from these experiments support current perceptions of stock dynamics, but have also indicated a general unreliability of the absolute magnitude of biomass estimates generated from swept-area surveys for rockfishes. In addition, they have indicated that cooperative experiments using the fishery as a research tool are possible, but correct interpretation of results is highly vulnerable to deliberate mis-reporting of catch and effort data. This is particularly important where the indices of population status undergo only subtle changes. Rigorous and costly monitoring of fishing activities may be required for experiments to be successful. Results also indicate that, once an open-fishing regime has been instituted, it is difficult for managers to alter.

Au début des années 1980, on a entrepris des programmes de gestion expérimentaux de deux stocks de sébaste à longue mâchoire (*Sebastes alutus*) se trouvant près de la côte de la Colombie-Britannique. L'un des programmes portait sur la surpêche dirigée d'un stock près de la partie sud-ouest de l'Île de Vancouver pendant une période de 5 ans, et l'autre, sur une surpêche non dirigée pendant une période de 5 ans, suivie d'une période de fermeture d'égale durée dans la région de la flèche Langara dans l'entrée Dixon. Ces deux expériences ont été conçues et menées en collaboration avec le secteur de la pêche commerciale. Elles visaient à vérifier des hypothèses sur la dynamique des stocks, les estimations de la biomasse et de la productivité, les méthodes de détermination de l'âge et les valeurs de paramètres démographiques. L'expérience de l'Île de Vancouver a porté sur la période de 1980 à 1984 et le prélèvement prévu devait atteindre environ ≈ 160 % du rendement soutenu estimé. Les relevés sur le terrain, les statistiques sur la pêche commerciale et les caractéristiques biologiques du stock indiquent tous une importante baisse de la biomasse due à la surpêche expérimentale. L'expérience de la flèche Langara a débuté à l'automne de 1983 et s'est poursuivie jusqu'à aujourd'hui. Les statistiques de la pêche commerciale, une baisse de la participation et de l'effort de pêche, et des caractéristiques biologiques montrent également une diminution de la biomasse associée à la pêche sans restrictions. La pêche repose maintenant presque entièrement sur le recrutement annuel, et la cohorte relativement forte de 1976 a été réduite à un rôle mineur. Les résultats de ces expériences corroborent les perceptions actuelles de la dynamique des stocks, mais ils ont également montré un manque de fiabilité générale de l'ampleur absolue des estimations de la biomasse produites à partir de relevés des sébastes par balayage. De plus, ils ont également révélé qu'il est possible de réaliser des expériences en collaboration faisant appel à la pêche comme outil de recherche, mais une interprétation correcte des résultats est très sensible à une fausse déclaration volontaire des données sur les prises et l'effort. Ce point est particulièrement important lorsque les indices de l'état de la population ne subissent que des modification subtiles. Il peut être nécessaire d'exercer une surveillance rigoureuse et coûteuse des activités de pêche pour la réussite des expériences. Les résultats montrent également que, lorsque le régime de pêche libre est établi, il est difficile pour les gestionnaires de le modifier.

Fisheries for rockfishes (genus *Sebastes*) throughout the world have been characterized by initial years of very high production followed by rapid declines, and prolonged periods where catches are less than 10–15% of peak values (Leaman 1987). Several factors contribute to these performance histories. Rockfishes aggregate and appear to respond to specific bathymetric and hydrographic features, so that given locations will have predictable presence of fish. These species are also acoustically dense and their visibility to modern echo sounders makes populations extremely vulnerable to fishing. Managing to a sustainable harvest of these species is made difficult by their slow growth, low natural mortality, and late recruitment (Archibald et al. 1983). Application of new ageing methodology to rockfishes in the late 1970's implied that instantaneous natural mortality rates (M) were less than 50% of previous estimates (Archibald et al. 1981). For example, the estimate of M for Pacific ocean perch (*S. alutus*) decreased from 0.12–0.15 to 0.05. The optimal total mortality rate (Z) based on a policy of setting $F = M = 0.05$ would therefore be 0.10, compared with previous estimates of 0.24–0.30. This low rate of productivity (≈5–10% of stock biomass per year) requires that large adult stocks be maintained to support even modest fisheries.

Long intervals between recruitment of the strong cohorts which dominate *Sebastes* populations also renders rockfish populations sensitive to exploitation (Fig. 1). The lengths of these intervals (6–18 yr) and the range of spawning biomass they encompass suggest that environmental influences may be more important than spawning biomass in determining cohort success. However, it is also important to know whether these dominant cohorts are strong in absolute abundance, or only in relation to surrounding cohorts. Uncertainty about whether stock size or environment is the major factor in cohort abundance influences the choice of management strategy. Managers should therefore know the form and variance about the stock-recruitment relationship, and whether there is a critical spawning biomass below which recruitment is imperiled.

The answers to these questions are difficult to determine from simple observations or existing fishery statistics. Most rockfish stocks off British Columbia are now at low abundance, and we have few observations of undisturbed stocks (Leaman 1987). The late age at recruitment for many commercial rockfishes (10–20 yr) has generated relatively static management strategies, because the time required to recover from errors is equally long. This lack of contrast in fishery statistics and management strategies means that even long periods of observation may be insufficient to resolve actual causative factors from potential alternatives (Walters 1986). Therefore, in 1979, the Canadian Department of Fisheries and Oceans (DFO) initiated programs to examine this uncertainty and attempt to resolve the disagreements between DFO and the fishing industry about rockfish management strategies. An important strategy of these programs was to create a partnership between DFO and the fishing industry, with the objective of testing the validity of biomass and mortality estimates derived from field surveys and analytic models, and examining the hypotheses about stock productivity derived from simulation models (e.g., Archibald et al. 1983). An additional objective was to deliberately manage fisheries for information, by creating contrast in observations.

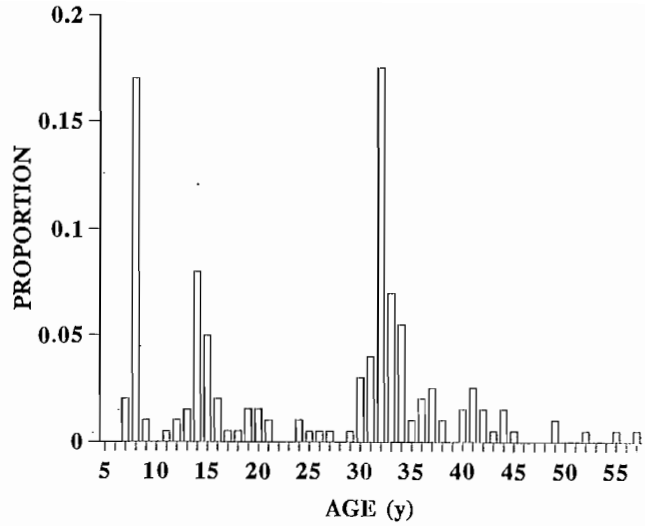


Fig. 1. Age composition of Pacific ocean perch off the west coast of the Queen Charlotte Islands in 1984, showing the typical pattern of infrequent strong cohorts.

Two experimental programs were initiated on stocks of Pacific ocean perch (*Sebastes alutus*, POP) off the west coast of Canada (Fig. 2). One program (Vancouver Island) generated a specified overharvest over a limited period, while the other (Langara Spit) was to be an unlimited harvest over a specified period, followed by an equivalent period of no harvest. The experiments attempted to implement concepts from the field of adaptive management, first articulated by Holling (1978) and cast in a formal framework by Walters (1986). Recent programs, particularly an excellent one developed off western Australia (Sainsbury 1988), have employed this formal planning and evaluation framework. Although that framework suggests approaches for simultaneous testing of alternative hypotheses, biological, historical, geographic and economic considerations limit the number of alternatives that can be examined simultaneously in real-world fisheries. Thus, for the experimental programs we began in 1979, a sequence of experiments was chosen to address a limited number of questions about stock dynamics and productivity.

This paper presents the background to these experiments and results to 1991. While we examine all of the available indices of stock condition, we do not present exhaustive assessments for the experimental stocks. These assessments will be presented in another forum. Rather, we concentrate here on the process of establishing and conducting these experiments. We also examine some of the causes for success and failure, and suggest some elements that should be included in the design of future experiments.

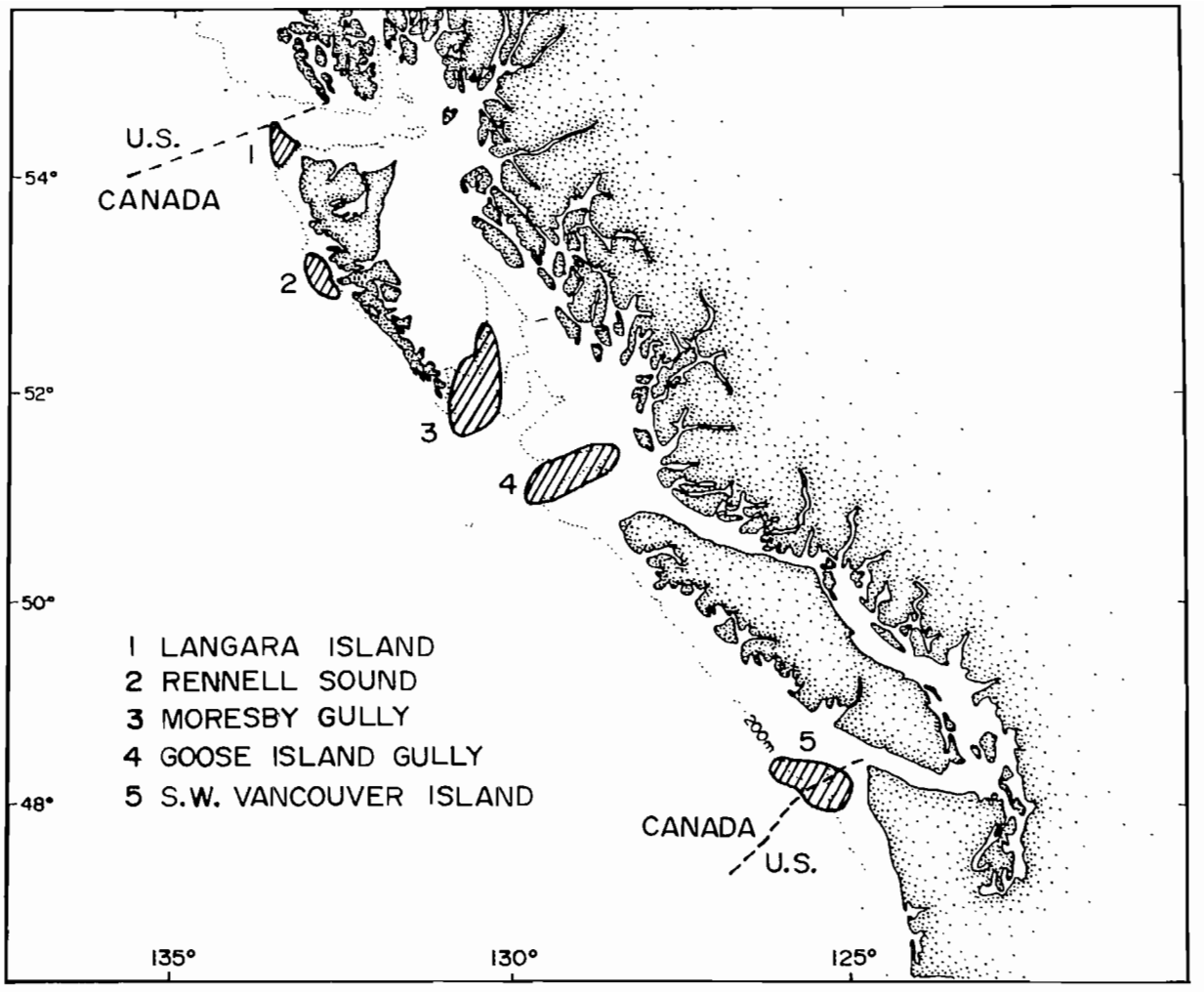


Fig. 2. Locations of principal Pacific ocean perch stocks off the coast of British Columbia.

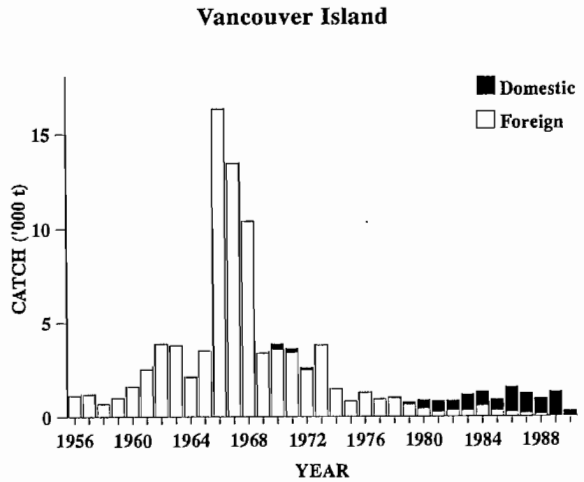
Vancouver Island

Background

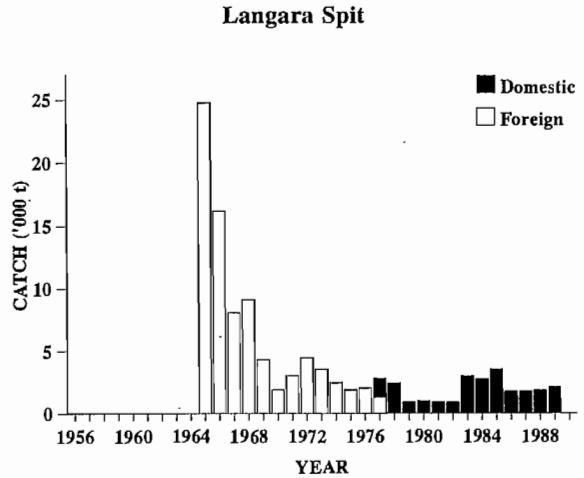
Foreign fleets conducted intensive fisheries on the Vancouver Island stock of POP in the mid-1960's (Fig. 3). Analyses by several investigators, using a variety of techniques, showed stock decreases ranging from 69–82% over the 1965–1977 period (Gunderson 1981; Kimura 1981; Kimura and Tagart 1982). These analyses also concluded that the POP stock extended across the southern boundary area between Canada and the United States. Through a bilateral technical committee, Canada and the U.S. prohibited directed fishing on this stock in 1977, in an attempt to initiate rehabilitation. This regulation was in effect in both countries, until 1980.

In 1979, the British Columbia groundfish industry began to press for a re-opening of this area, based on their acoustic observations of "large" aggregations of rockfishes. DFO biologists had no evidence from biological sampling programs to suggest a major change in stock abundance since 1977 and suggested that, prior to 1977, the size of aggregations had been affected by fishing activity. Rockfishes naturally aggregate (Leaman et al. 1990), particularly when undisturbed by fishing. Fisherman have reported that fishing disrupts these aggregations and re-distributes the fish, resulting in lower catch rates than those on the initial aggregations. The lack of fishing activity on this stock during 1977–1979 may have contributed to the observations reported by the fleet. However, the industry view that existing estimates of biomass and yield were too conservative persisted. To break this impasse, DFO proposed, after extensive internal discussions, an experimental overharvesting program in cooperation with industry. It would examine both the validity of the existing biomass and productivity estimates, and assist in the validation of the ageing method. If yield estimates were correct, the overharvest would decrease biomass and inject a strong negative anomaly into the age composition. It was hypothesized the latter would occur because cohorts recruiting during the experimental period would experience relatively higher fishing mortality, than preceding or succeeding cohorts. A regular progression of this anomaly over time would help to validate the ageing methodology.

The derivation of a suitable index of abundance for rockfishes has occupied a central place in research programs within

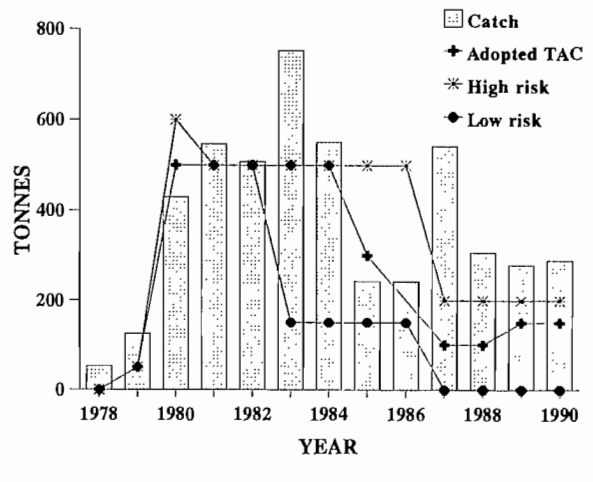


a.

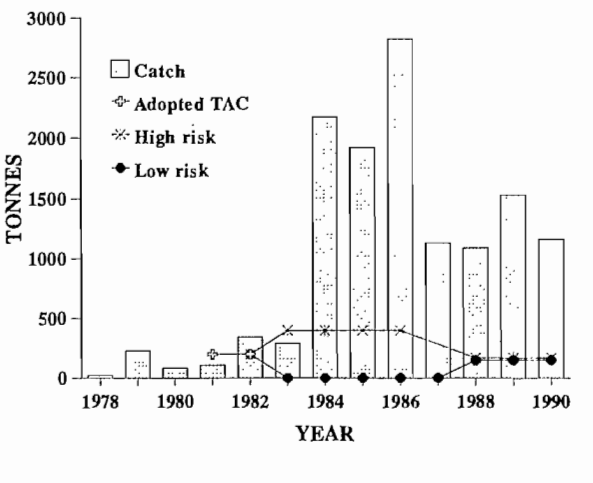


b.

Fig. 3. Foreign and domestic landings of Pacific ocean perch from the Vancouver Island (a.) and Langara Spit (b.) stocks, 1956–1990.



a.



b.

Fig. 4. Management history of the southwest Vancouver Island (a.) and Langara Spit (b.) stocks of Pacific ocean perch, 1978–1990. High and low risk values bound the range of estimated total allowable catch (TAC).

DFO and other agencies. Swept-area surveys for aggregated species have several sources of potential bias (Byrne et al. 1981; Smith 1981, 1990). The relationship of absolute stock abundance to that estimated from such surveys is uncertain and evidence from analytical models (e.g., Archibald et al. 1983) has suggested that, although surveys may capture broad trends well, they may not produce reliable estimates of absolute abundance. Catch rates (CPUE, t/h) of commercial fisheries are also regularly examined for evidence of stock trends, but changes in fishing power and catchability, or potential interactions of effort and catch rate may also affect the reliability of these indices. For the fisheries of concern here, no uniquely suitable index was identified, and evaluation of experimental results therefore incorporated all available indices and biological sampling data.

The experimental design called for comprehensive trawl surveys before and after a specified period of overharvesting. Although the shortcomings of trawl surveys were acknowledged, the planning team for the experiment believed that some standardized measurement of relative abundance would be a valuable adjunct to fishery statistics and biological samples.

The period of overharvesting was proposed as three years, after which the quota would be returned to the sustainable level. The estimated sustainable yield for the Vancouver Island stock in 1980 was 300 t (Fig. 4; Stocker 1981). The estimates of low-risk and high-risk yields in Fig. 4 are based on the probability distributions of yield and stock biomass as a function of fishing mortality, developed by Archibald et al. (1983). The proposed harvest of 500 t was equal to the high-risk yield, 67% greater than the estimated sustainable level. Industry accepted this proposal and participated in the design and conduct of the survey program. At the same time, U.S. agencies were conducting a similar survey on the portion of the Vancouver Island stock present in the U.S. zone. The U.S. management approach being taken at that time was a

Table 1. Biomass estimates (t) for rockfishes (*Sebastes* spp.) from the 1979 and 1985 swept-area trawl surveys of the Vancouver Island experimental fishing area.

Year Depth (m) Number of tows	1979 183–365 45	1985 160–439 56	Relative change (%)
Sebastes aleutianus	312	81 ± 70%	−74.2
S. alutus	4217	1862 ± 17%	−55.8
S. babcocki	158	94 ± 46%	−40.6
S. borealis	138	20 ± 167%	−85.6
S. brevispinis	34	197 ± 202%	483.1
S. crameri	148	48 ± 99%	−67.6
S. diploproa	1207	258 ± 20%	−78.6
S. elongatus	42	41 ± 176%	−2.4
S. entomelas	9	2 ± 238%	−82.7
S. flavidus	–	19 ± 283%	–
S. helvomaculatus	19	8 ± 249%	−60.2
S. paucispinis	351	199 ± 326%	−43.2
S. pinniger	13	16 ± 138%	25.3
S. proriger	173	121 ± 162%	−30.2
S. reedi	153	127 ± 188%	−17.1
S. zacentrus	106	387 ± 109%	236.9
TOTAL	7080	3481 ± 28%	−50.8

20 yr rehabilitative strategy (Ito et al. 1987). This strategy specified a yield of 600 t from the U.S. zone, however it was based on an assumed instantaneous natural mortality rate (M) of 0.12. The estimate of M now accepted by scientists of both nations is approximately 0.05 (Archibald et al. 1981). If this estimate is correct, the U.S. strategy was also an overharvest. The combined quota for the stock during the course of the experiment was therefore 1100 t, compared with an estimated sustainable yield of approximately 300 t each in U.S. and Canadian waters.

Pre-experiment Surveys and Experimental Harvesting

The Canadian portion of the Vancouver Island stock was surveyed by a chartered, commercial trawler in 1979 (Lapi and Richards 1981). Survey coverage was intense, by normal stratified-random survey standards, and both the vessel master and DFO staff were satisfied with the areal and depth coverage. Trawling was conducted in approximately 26% of the total bottom area in the depth ranges to which POP is common (Leaman et al. 1988). Total rockfish biomass was estimated at ≈7100 t, of which POP accounted for ≈4200 t (59.5%; Table 1). The total catch rate (CPUE) of the survey vessel was ≈1.9 t/h for all rockfishes and ≈1.2 t/h for POP (Table 2). These CPUEs were approximately 70% greater than those recorded by the commercial fleet prior to the 1977 closure and were therefore seen by industry to be consistent with their contention that yield in the area was substantially greater than estimates made prior to the survey. Industry believed that previous assessments had underestimated biomass, rather than that the stock had recovered after the short period of no harvest. The 1979 U.S. survey estimated the biomass of POP in the U.S. portion of the stock area as approximately 5500 t, with a 95% confidence interval of ±75% (Ito et al. 1987).

The quota for the Vancouver Island stock was raised to 500 t in 1980 and remained at that level until 1984 (Fig. 4). Although initially scheduled to last three years, the overharvesting experiment was extended an additional two years because the 1980 catch was below the target quota. During the experimental period after 1980, the catch of POP by the Canadian fleet always reached the 500 t level and exceeded it by 250 t in 1983. The U.S. fleet also achieved or exceeded their quota of 600 t. Unstandardized catch rates for the Canadian fleet over the experimental period declined rapidly (Table 3). The data from catch records with ≥ 25% POP in the hauls were standardized using a general linear model to account for varying fishing power (Gavaris 1980; Kimura 1981). Because of small sample sizes, the 90% confidence limits for the estimates were derived with the bias-corrected, percentile method of bootstrapping (Stanley 1992). These estimates of standardized CPUE also showed a general decline during the 1979–1985 period (Fig. 5a).

Post-Harvest Surveys and Changes in Biological Characteristics

The quota for the Vancouver Island stock was returned to the sustainable level of 300 t in 1985, and reduced subsequently to 150 t, in an attempt to begin rehabilitation. Both Canada and the U.S. conducted trawl surveys of the stock in 1985. Total rockfish biomass estimated by the Canadian survey was ≈3500 t, with POP accounting for ≈1900 t (Table 1). The point estimates from the survey represented a relative decline of 51% from 1979 values. Localities receiving the greatest fishing pressure over the 1980–1985 period recorded the greatest declines in estimated biomass (Leaman

Table 2. Catches and catch rates for rockfish (*Sebastes* spp.) from the 1979 and 1985 swept-area trawl surveys of the Vancouver Island experimental fishing area.

Year	1979		1985		Relative change in CPUE (%)
Depth (m)	183–365		160–439		
Hours Fished	94		100		
	Catch (kg)	CPUE (kg/h)	Catch (kg)	CPUE (kg/h)	
Sebastes aleutianus	8803	93.65	876	8.76	−90.6%
S. alutus	108,070	1149.68	24,146	241.46	−78.9%
S. babcocki	2760	29.36	1144	11.44	−61.0%
S. borealis	1902	20.23	586	5.86	−71.0%
S. brevispinis	1121	11.93	1104	11.04	−7.5%
S. crameri	2921	31.07	751	7.51	−75.8%
S. diploproa	32,287	343.48	3574	35.74	−89.6%
S. elongatus	692	7.36	208	2.08	−71.7%
S. entomelas	120	1.28	40	0.40	−68.8%
S. flavidus	–	–	188	1.88	–
S. helvomaculatus	206	2.19	47	0.47	−78.5%
S. paucispinis	13,228	140.72	1354	13.54	−90.4%
S. pinniger	191	2.03	120	1.20	−40.9%
S. proriger	3232	34.38	843	8.43	−75.5%
S. reedi	2077	22.10	668	6.68	−69.7%
S. zacentrus	2602	27.68	2161	21.61	−21.9%
TOTAL	180,212	1917.15	37,810	378.10	−80.3%

et al. 1988). The CPUE for all rockfishes combined fell 68%, to ≈0.4 t/h, while that for POP declined from ≈1.2 to ≈0.2 t/h (−79%), from 1979 values (Table 2). The 1985 U.S. survey estimated the POP biomass to be ≈ 2000 t, a decline of 63% from the 1979 U.S. survey estimate (Ito et al. 1987).

A comparison of biological samples from the 1979 and 1985 Canadian and U.S. surveys (Ito et al. 1987; Leaman et al. 1988) showed lower frequencies of those sizes of fish newly recruited to the fishery (32–36 cm) in 1985. Mean sizes of fish in samples from both Canadian and U.S. 1985 surveys showed slight increases from those in 1979, but no evidence of strength for cohorts in the 32–36 cm range, upon which the fishery will depend in future years (Fig. 6). The left limb of the catch curves provide little insight, aside from the observation that smaller sizes are less abundant in 1985 than in 1979. This result could reflect either high fishing mortality or low recruitment. However, the right limbs of the 1979 and 1985 length frequencies for fully recruited fish may be compared with those generated from known mortality and growth rates, using the method of Stanley (1987). This method, which assumes constant recruitment, generates length frequencies from known mortality rates and uses stochastic functions for both individual growth and overall growth curves. Known fishing mortalities are applied to simulated cohorts of these fish and the resulting length frequencies were compared with survey samples. These comparisons suggest that the larger fish present in the stock in 1985 experienced higher mortality rates, than those which had been experienced by the larger fish present in 1979. A total instantaneous mortality rate (Z) of ≈ 0.50 is consistent with the slope and location of the right limb of the 1985 length frequency. This level is approximately 400% above the estimated equilibrium rate of 0.10.

Evaluation of the Vancouver Island Experiment

Changes in the stock abundance indices during the course of this experiment were consistent with a significant decline in biomass. Relative biomass estimates from both Canadian and U.S. surveys were lower by at least 50%. Catch rates in the Canadian commercial fishery were also substantially lower following experimental harvesting. In addition, the size frequency of POP in 1985 suggested a total mortality rate at least four times the optimum level. The question of the relative influence of stock biomass or environment in determining cohort strength was not resolved by these results. Since the 50% recruitment age for a cohort is ≈ 11 yr, it is also too early to determine whether the overharvest has generated an anomaly in the age frequency spectrum.

The cooperation of the industry in the execution of this experiment was very good. There were no reports of data falsification and there was general acknowledgement that the biomass in 1985 was lower than in 1979. However, there was still disagreement over the level of harvest from the area that could be sustained. Therefore, DFO and industry did not achieve as much progress toward a shared view of the resource as was desired by either party. In spite of this, the healthy partnership forged for the experiment encouraged the participants to continue trying to resolve disagreements in an experimental framework.

Langara Spit

Background

The Langara Spit stock of *S. alutus* was also the object of major foreign fisheries during the mid 1960's (Fig. 3, Leaman

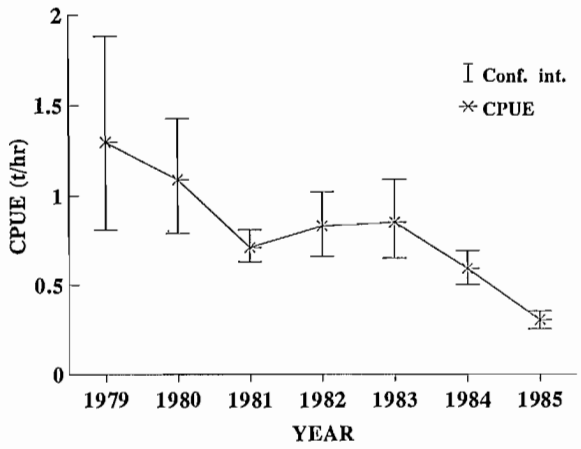
Table 3. Canadian landing statistics for Pacific ocean perch in the Vancouver Island experimental fishing area, 1967–1990. Percent qualification is the percentage of Pacific ocean perch in catch necessary for inclusion of data.

YEAR	Total landings	25% Qualification		
		Landings (t)	Effort (h)	CPUE (t/h)
1967	7.02	6.83	14.50	0.471
1968	0.12	–	–	–
1969	2.49	1.26	12.50	0.101
1970	303.86	273.11	293.20	0.931
1971	218.38	200.82	400.70	0.501
1972	117.25	12.67	14.30	0.886
1973	–	–	–	–
1974	–	–	–	–
1975	5.46	1.46	7.00	0.209
1976	1.29	0.87	1.00	0.870
1977	16.17	8.81	46.30	0.190
1978	53.06	50.95	38.90	1.310
1979	124.86	121.03	85.70	1.412
1980	429.85	395.38	380.90	1.038
1981	547.32	504.96	709.80	0.711
1982	507.97	452.48	555.00	0.815
1983	751.52	325.23	411.10	0.791
1984	551.17	404.10	720.60	0.561
1985	243.11	195.86	692.70	0.283
1986	242.10	140.49	185.10	0.759
1987	542.27	394.80	460.60	0.857
1988	307.46	77.64	228.80	0.339
1989	279.16	146.47	426.00	0.345
1990	289.60	148.28	490.98	0.302

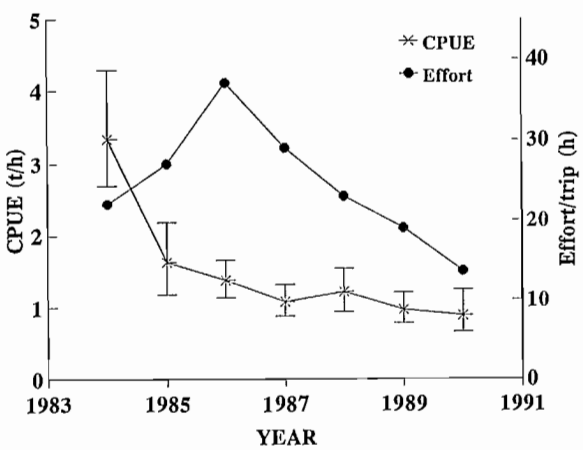
1987). However, these fisheries were documented poorly. The data on species composition and catch locality for Soviet fisheries are at a gross level of resolution, while species composition data of Japanese catches are for broad categories of rockfish only. However, Ketchen et al. (1978) and Ketchen (1980) developed methods for higher resolution of these data. That work suggests approximately 85% of the foreign catches of rockfishes from the west coast of the Queen Charlotte Islands originated in the Langara Spit area (Fig. 2). Evidence from DFO surveys in the early and mid-1960's (Westrheim 1970, 1972) indicated that the fishery during the initial 1965–1970 period harvested primarily POP. However, a shift in species composition occurred about 1973, as indicated by an increase in records of "Other Rockfish" in Japanese catches (Ketchen et al. 1978). By 1977, the catches recorded as "ocean perch" were composed of up to 85% red rockfishes, other than *S. alutus* (Leaman et al. 1978). Unlike other POP stocks on the B.C. coast, this stock was not exploited by the domestic fleet during the major foreign fishery. Directed fishing by the domestic fleet began in 1979 (Table 4) and was restricted by quota until the fall of 1983. Catches of POP during this period averaged < 250 t/yr.

The purpose of the Langara experiment, like that for Vancouver Island, was to improve the estimation of biomass and available yield for rockfish stocks. These estimates are derived from a variety of sources including trawl surveys, catch statistics, analytic models, and interpretation of biological samples. Scientists and industry had concerns about the validity of each information source. Research surveys in this area during the late 1970's (Leaman and Nagtegaal 1982, 1986) had created strong reservations about the reliability of trawl survey estimates of absolute abundance for rockfishes. The reservations were based on the observed behavioural patterns of these species in response to tidal cycles, illumination level, and the trawl gear itself. These patterns produce orders of magnitude change in the availability of the species to the survey gear, as expressed in survey CPUE. Recent work has used hydroacoustic technology to describe some of these rockfish behavioural patterns (Leaman et al. 1990).

The Vancouver Island experience showed that cooperative experiments could be used to increase the rate of learning about rockfish stock productivity and dynamics. Although that experiment did not achieve all the objectives of the design team, both the DFO and industry believed that continued cooperation would help resolve remaining disagreements. Managers and scientists attached importance to the exploitation history and biological characteristics of stocks, in the development of management strategies. Undoubtedly, they were also influenced by mandated responsibility for resource conservation, and the lack of any successful management of rockfish stocks subject to major commercial fisheries elsewhere on the Pacific coast (Pacific Fishery Management Council 1991). On the other hand, industry representatives attached greater importance to the quantity and duration of their observations while fishing and, naturally, were also concerned with maintaining profitable fisheries.



a.



b.

Fig. 5. Standardized catch rates and 90% confidence intervals for Pacific ocean perch fisheries off southwest Vancouver Island (a.) and Langara Spit (b.), 1979–1985. The mean effort per fishing trip to the Langara Spit area is also presented in 5b.

Meetings were therefore held between DFO and industry in the fall of 1982 to plan a second experiment, with an even greater intervention. This experiment required a relatively isolated target stock, for which there was good historical data. The Rennell Sound POP stock (Fig. 2), met both these criteria and was relatively undisturbed by previous exploitation. However, it was also isolated and industry argued that travel costs to this area could be prohibitive. As an alternative, the Langara Spit POP stock was chosen for the experiment. From an experimental perspective, its shortcomings were the extensive history of foreign fishing and the potential movement or exchange of fish along the isobaths to which POP is common, north into U.S. waters. Its strong points were that it had been surveyed extensively in 1979, and it offered the expectation of continuous industry participation.

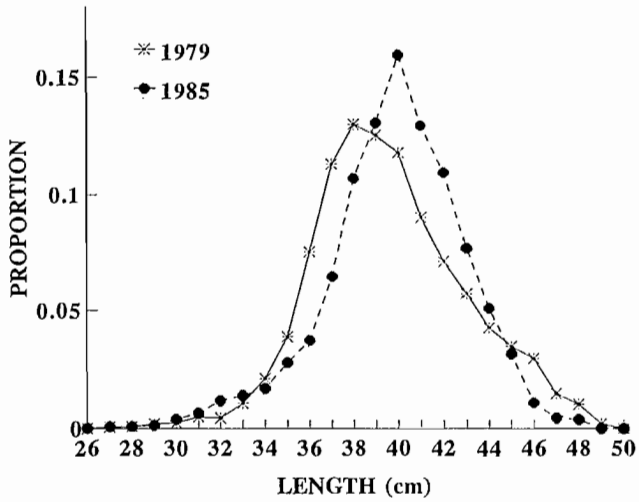


Fig. 6. Length frequency distribution of Pacific ocean perch catch during 1979 and 1985 biomass surveys off southwest Vancouver Island.

The experiment was to provide unconstrained fishing mortality bordered by periods of sustainable or conservative harvest, to obtain maximum contrast with the period of high fishing mortality. Therefore, the planning team agreed that the experiment would consist of two treatments: 3–5 yr of unrestricted harvest, followed by an equivalent period of conservative harvest or closure. The detailed objectives of the experiment were:

- to test the validity of trawl survey biomass estimates;
- to develop estimates of fishing mortality based on removals;
- to examine the stock-recruitment relationship relative to other stocks and to fishing mortality;
- to develop a detailed biological and fishery statistics database for use in analyses, e.g., fishing power estimation;
- to validate ageing techniques through the injection of a large negative anomaly in the age spectrum of the stock;
- to examine movements as they might be interpreted from micro-scale distribution of fishing effort; and
- to involve industry in both research and management programs.

A number of conditions, under which the experiment was to be conducted, were agreed upon and explicitly stated at the outset. These were:

- a trawl survey would precede and follow the experiment;
- industry would supply detailed logbook data (tow locations, durations and depths, species compositions, etc.) for their fishing in the area; and,
- the fishery would have to be closed for a period equivalent to the open period, after the unrestricted fishing portion of the experiment.

Table 4. Canadian landing statistics for Pacific ocean perch in the Langara Spit experimental fishing area, 1976–1990. Percent qualification is the percentage of Pacific ocean perch in catch necessary for inclusion of data.

YEAR	Total landings	25% Qualification		
		Landings (t)	Effort (h)	CPUE (t/h)
1976	–	–	–	–
1977	1.42	0.70	2.30	0.304
1978	22.22	6.65	16.80	0.396
1979	227.49	223.83	108.50	2.063
1980	84.56	64.80	39.50	1.641
1981	109.22	53.15	24.10	2.205
1982	342.23	194.18	109.30	1.777
1983	291.98	208.28	193.50	1.076
1984	2173.86	1779.38	980.00	1.816
1985	1921.21	1712.09	1514.50	1.130
1986	2725.37	2558.46	2319.30	1.103
1987	1129.70	1015.88	1119.60	0.907
1988	1088.79	1027.99	1373.60	0.748
1989	1532.50	1401.89	1574.40	0.890
1990	1162.00	1123.65	1336.09	0.841

Pre-Experiment Surveys and Experimental Harvests

The experiment began with a joint industry-DFO, two-boat trawl survey in July, 1983 (Leaman and Nagtegaal 1986). This survey repeated work conducted during a similar two-boat survey in 1979 (Lapi and Richards 1981). Leaman and Nagtegaal (1986) compared the design of the two surveys and presented comparisons of biomass estimates, by area, after correcting for differences in gear and effective path width (Table 5). The prime difference between the two surveys was that the 1979 survey used a stratified systematic design, while the 1983 survey used an encounter-response design (Leaman 1981). The first method estimated biomass by area, while the second employed detailed echo-sounding and estimated biomass only for those aggregations of fish encountered and mapped. Both surveys identified the same areas of rockfish abundance. Although comparisons of the estimates for these areas can be made, it should be noted that the methods used for extrapolating to total biomass for the two surveys were different. The 1979 survey extrapolated over strata whereas the 1983 survey extrapolated over schools.

Both surveys produced similar estimates for the western Langara Spit area (POP biomass of 1208 t vs. 1074 t for 1979 and 1983, respectively; Fig. 7, aggregations C and D). Neither the 1979 nor 1983 survey achieved consistent coverage of the eastern Langara Spit area, although all vessels identified it as a secondary centre of rockfish abundance (Fig. 7, aggregation E). That area has extremely rough bottom and not all vessels could successfully fish on it. The 1979 survey estimated approximately equal proportions of *S. alutus, S. reedi,* and *S. zacentrus* in this area, with *S. alutus* abundance being equal to the estimate for the western portion of Langara Spit. The biomass of *S. zacentrus* in this area is of limited importance because most fish are less than the minimum commercial size (30 cm). A much smaller aggregation of rockfishes was also located slightly north of the Canada-U.S. boundary area (Fig. 7, aggregation B). This aggregation was estimated to contain only 10% of the biomass of the major aggregation on the western portion of Langara Spit. No significant schools of rockfishes were encountered going north into U.S. waters until approximately 55°05′ N, near Cape Bartolomé.

Annual landings of total rockfishes increased steadily for the first three years of the experiment, to a peak of almost 5000 t (Fig. 8), before declining dramatically in 1987. Since 1986, the catch has averaged approximately 2300 t/yr. Whereas S. alutus comprised over 70% of the catch in 1984, it accounted for $< 55\%$ of the total landings in 1990. Increasing quantities of alternative species such as *S. aleutianus, S. proriger, S. reedi,* and *S. brevispinis* appeared in the landings and may have been used to supplement fishing on the major target species. The occurrence of these other species was not consistent over the period of the fishery. The proportions of both *S. reedi* and *S. proriger* first increased and then decreased to lower but more stable levels. A strong market for frozen *S. aleutianus* also developed over the course of the fishery.

The most striking aspect of the experimental fishery was that the landings far exceeded even the most optimistic estimate of exploitable biomass derived from the trawl surveys. The recruitment of incoming cohorts, while clearly able to increase biomass over that estimated from the survey, could not account for the differences. Recruitment to POP fisheries by a cohort is extremely gradual, with partial recruitment exceeding 0.5 only at ages $\geq$ 11 yr, and full recruitment not occurring until $\approx$ 15 yr (Fig. 9). Even the aggregation of POP, which could be sampled by only one vessel in the eastern Langara Spit area during 1979 and 1985 (Fig. 7, aggregation E), could not alone account for the difference between the exploitable biomass estimates and the subsequent catches.

It was apparent even early in the experiment that the requirement for accurate reporting by vessels, agreed upon by the planning team, was not being met. There was strong evidence that some proportion of landings, reported to have

Table 5. Comparison of biomass estimates (t), by subarea, from the 1979 and 1983 surveys of the Langara Spit experimental fishing area (after Leaman and Nagtegaal 1986). Separate estimates for the 1979 survey are for the two survey vessels used.

	SUBAREA AND SURVEY YEAR			
	Western Spit		Northeastern Spit	
SPECIES	1979	1983	1979	1983
Sebastes alutus	1143/206	1005	1064/206	138
S. babcocki	1/-	1	81/6	1
S. brevispinis	13/15	8	59/10	2
S. entomelas	-/-	27	-/-	0.5
S. proriger	26/10	16	142/10	–
S. reedi	-/5	1	923/245	3
S. zacentrus	25/23	16	1086/137	6
TOTAL	1208/259	1074	3355/614	151

come from the experimental area, originated elsewhere. Indeed, industry has subsequently acknowledged that some data submitted prior to 1989–1990 were "unreliable". While difficult to quantify, as much as 40% of the fishing observations may have been incorrect. Although these data have been analyzed and included in discussions between DFO and industry, aspects of them are presented here for illustration only.

Catch Rate Trends

Standardized CPUE declined from 3.34 t/h in 1984 to 0.89 t/h in 1990 (Fig. 5). It accompanied a rise and fall in average total fishing effort per trip (Fig. 5). By 1989, vessels were only spending 20 hours, about 2 fishing days, before leaving the area. This is also reflected in the fact that, for these vessels, there was no indication of diminishing catch rates with increasing hours towed. Not enough towing time was spent on the grounds to produce a fishing effect during a single visit, as has been seen for other B.C. rockfish fisheries (Stanley 1992).

Changes in Biological Characteristics

The size composition of POP shifted dramatically during the course of this experiment (e.g., Fig. 10 for females). In 1983, prior to the initiation of the experiment, the modal sizes of males and females were 39 and 41–42 cm, respectively. The modal sizes have since declined to 35 and 37 cm, respectively. In 1983, 65% of the females and 28% of the males in the stock were > 40 cm, whereas in 1989 the respective percentages were 20% and 14% (Fig. 11). The mortality rate associated with the 1989 length frequency spectrum was also analyzed with the simulation model of Stanley (1987). Results of this analysis (Leaman 1989, 1990) indicate that POP in the Langara Spit region has experienced total instantaneous mortality rates (Z) in the 0.50–0.60 range, since the initiation of the experiment. These rates are 500–600% higher than the estimated equilibrium level of total mortality ($Z = 0.10$).

Samples from the 1990 fishery, which include those from dockside (3 samples) and observers at sea (32 samples), show a consistent pattern. They indicate a weakening of the contribution from the size range associated with the 1976 cohort, and a small increase in the relative contribution of fish > 40 cm, compared with 1989 (Fig. 11). However, this contribution is still only 13% and 25% of the total number of males and females, respectively, compared with 28% and 65% in 1983. The gradual trend to increasing proportion of larger fish in the most recent years, together with declining catch rates, reflects the recruitment of relatively weak cohorts.

At the beginning of the major fishing pressure in the experiment (1984), the fully recruited component of Langara Spit POP stock was dominated by age groups in the 13–20 yr range (Fig. 12). While the 1969 cohort was the most abundant of the recruited cohorts, there were still remnants of the 1952 cohort, one of the strongest of this century (Archibald et al. 1981). The 1976 cohort, although only partially recruited, was beginning to appear in strength, at age 8. This 1976 year class appears to have been relatively stronger than surrounding cohorts in this stock, although not as numerically strong as in other stocks (Leaman 1990). The age spectrum for the Langara Spit stock reflected the historical foreign fishery but was qualitatively similar to that for the Rennell Sound stock, immediately to the south. However, the effects of the high fishing mortality produced during the experimental fishery can be seen in the 1989 comparison of the age spectra of these two stocks (Fig. 12). The Langara Spit stock age composition has been severely truncated and the 1976 cohort reduced to subordinate status.

For the Langara Spit stock, the progressive truncation of the age composition towards younger fish and the declining catch rates indicate that the biomass has declined. The 1976 cohort in the Langara stock has diminished more rapidly than in other stocks. For example, in the Moresby Gully stock of POP which has not had as high fishing mortality, the 1976 cohort still accounts for over 40% of the numbers of fish. Age truncation and falling catch rates indicate high fishing mortality, although an increase in F alone cannot fully account for the rate at which the 1976 year class is decreasing, relative to succeeding cohorts. Recruitment of POP to the fishery is primarily a behavioural rather than a physical process. Fish are physically vulnerable to the gear at approximately 6 yr old but are not fully recruited as a cohort until approximately 15 yr old (Fig. 9; Archibald et al. 1983; Leaman 1991). The rapid depletion of this 1976 cohort in the Langara Spit area suggests that it may have experienced higher partial recruitment than previous cohorts. A low biomass of adult fish may increase the rate of this behavioural recruitment, although this cannot be determined from these data.

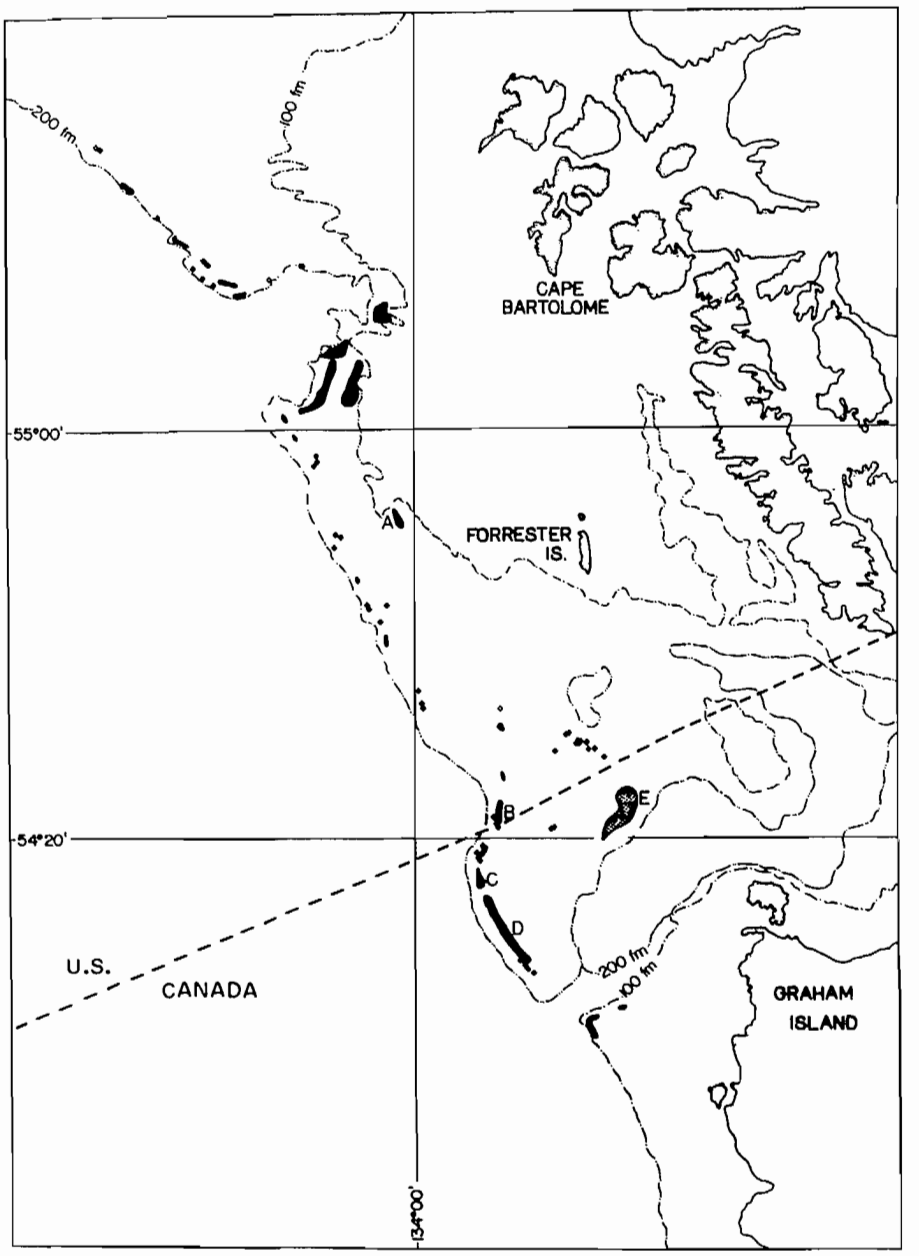


Fig. 7. Rockfish schools located during the 1983 Pacific ocean perch biomass survey of the Langara Spit-southeast Alaska area (from Leaman and Nagtegaal 1986). See text for discussion of schools B–E.

Implementation Problems

Scientists regarded the stock in the Langara Spit area as a unique experimental unit. However, industry tends to view stocks in a more integrated and opportunistic way, which reflects the normal conduct of fishing. Therefore, some vessel masters attempted to use a refusal to provide logbook information in 1984 as leverage to obtain concessions on rockfish quotas in other areas. Faced with this conflict, researchers recommended suspension of the experiment because one of the fundamental goals, data capture, was not being met (Leaman 1985). Managers were also concerned about these events but, rather than terminating the experiment, worked with industry to develop a compromise. A permit system was instituted which required detailed logbook data as a condition of the permit to fish in the area. Vessels were also required to temporarily surrender their licence to fish in all other areas of the coast.

The success of the compromise system required higher levels of surveillance than could be achieved with available resources. Surveillance is particularly difficult for rockfish fisheries, where enforcement staff require detailed knowledge of rockfish identification and trawling practices. The same lack of resources precluded direct enforcement actions, which might have improved compliance with the experimental design. The quality and quantity of submitted data continued to be uneven. However, further modification of the experimental design was not possible. In addition, and contrary to

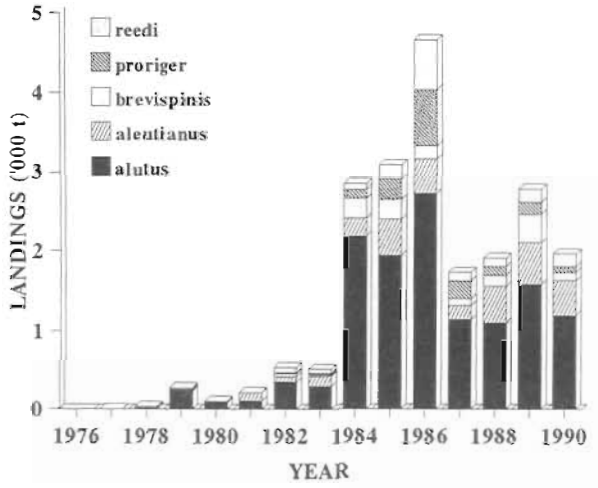


Fig. 8. Landings, by species, of the rockfish catch in the Langara Spit area, 1976–1990.

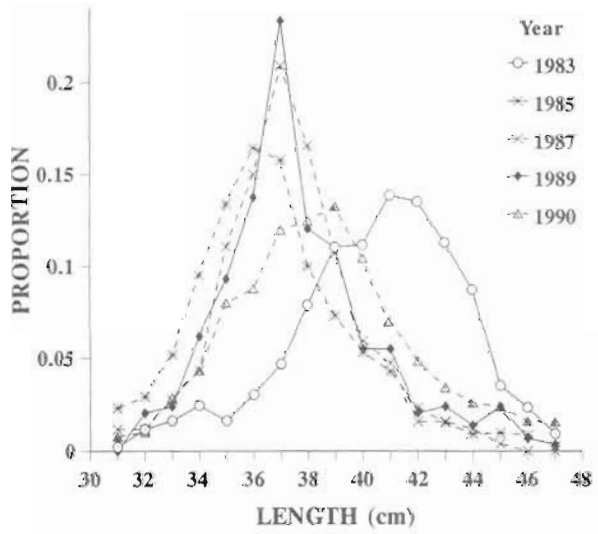


Fig. 9. Length and vulnerability to the fishery, by age, for Pacific ocean perch. Horizontal line represents the point where 50% of a cohort is vulnerable to the fishery.

concerns about accessibility expressed during the design of the experiment, the experimental area attracted vessels which landed to ports as far away as southern B.C. and Washington state.

The beginning of the second phase of the experimental design, closure, was scheduled for 1988. However, two factors made it difficult to implement this phase of the experiment. First, industry had come to depend on harvest levels achieved during the unrestricted fishing phase. Second, some fishermen were still able to obtain profitable trips from the area because of the aggregating behaviour of rockfishes, and the lowered number of participants in the fishery. This aspect has also been noted in other rockfish fisheries (Gunderson 1984; Leaman 1991). These individuals therefore argued for continuation of the experiment on the basis that little or no overexploitation had been demonstrated. When other fishermen ceased fishing in the experimental area, they also naturally ceased providing any data on the experiment. Lastly, the long lag in response of some stock indices to the effects of harvest,

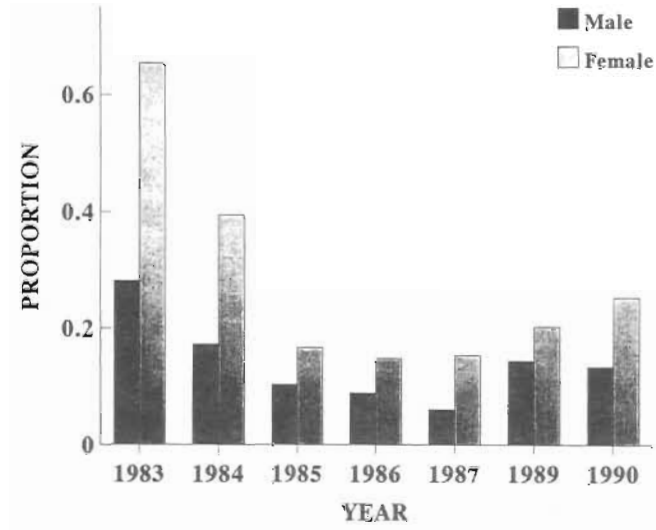


Fig. 10. Proportional length composition of female Pacific ocean perch catches in the Langara Spit fishery, 1983–1990, showing truncation of the initial size composition.

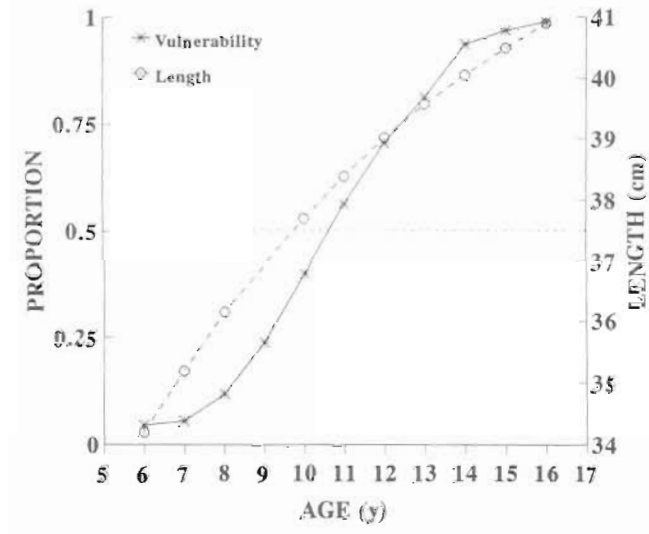


Fig. 11. Shift in proportion of fish > 40 cm in catches of the Langara Spit Pacific ocean perch fishery, 1983–1990.

because of the late age at recruitment of rockfishes, meant that not all indices from the experiment presented a consistent picture of stock status.

Managers were faced with a classic impasse of adaptive management programs and an additional compromise among the participants was sought. Managers favoured a controlled harvest at a level lower than during the first phase, the "conservative compromise" of Walters (1986). Industry wished to continue to benefit economically from the stock. Scientists recommended that, since virtually any harvest would prevent rehabilitation and be uninformative, two options could be considered: closure, or continuation of the experiment to more thoroughly document the status of the stock. If the latter were adopted, it would be important that industry continue to apply similar levels of fishing effort, such that commercial CPUE would not only remain comparable among years, but also be

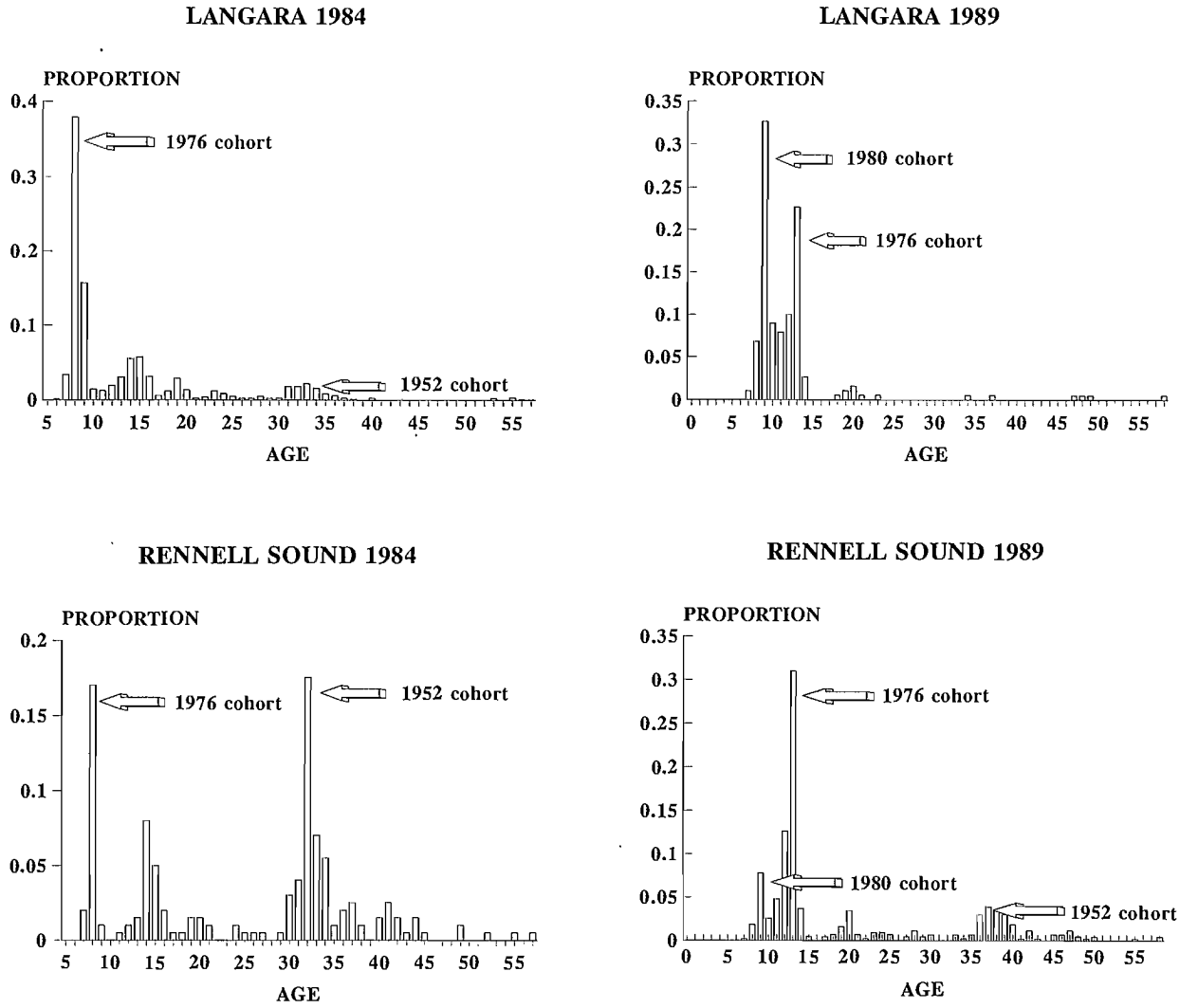


Fig. 12. Age compositions of Pacific ocean perch stocks from the Langara Spit area and from Rennell Sound in 1984 and 1989.

based on a reasonable number of observations (Stanley 1992). Therefore, if the second option were to be adopted, it was requested that the minimum level of effort from the previous years of the experiment ($\approx$ 1100 h) be applied.

The fishery was kept open for 1988–1989. The target level of effort was achieved in both years (Table 4). Surveillance efforts were increased to improve compliance and assure data quality. Finally, a mandatory observer program, funded by DFO, was established in late 1989, and in 1990 modified into a user-pay program. Industry's response to this program was largely negative. Trawl effort for POP in 1990 lagged seriously behind that for the comparable 1989 period, until fishing for rockfishes in all other areas of the coast was closed in November, due to quota attainment. Trawling effort in the Langara Spit area then increased substantially and the total figure for 1990 was approximately equal to that for 1989.

Analyses of reliable data from the observer program indicated noteworthy declines in some indices of stock status. Standardized CPUE for vessels previously suspected of noncompliance was only 76% of that for other vessels. These suspect vessels had reported CPUE well in excess of other vessels, during the period when observer verification was not required. The relationship between catch rate and effort for these vessels was also negative and highly non-linear, implying strongly diminishing returns with increasing effort.

For 1991–1992 there was again debate about instituting the second phase of the experiment. Researchers, managers, and the primary industry advisory body to DFO recommended closure of the Langara Spit area, with various options for the areas immediately adjacent. However, economic issues concerning fish plants and communities benefiting from the unrestricted harvest levels continue to play a role in the conduct and evaluation of the experiment.

The future of the Langara Spit experiment is somewhat uncertain. A compromise closure adopted for 1991–1992 permits fishing very close to the experimental area, rather than providing a large buffer zone. Hence, there is potential for illegal fishing. A commitment for a closure of five or more years, consistent with the provisions of the original experimental design, has not been made although there is optimism that the experiment will be completed as designed.

Some alternative management scenarios have been suggested, including incorporating this area into a coastwide quota, with no specific provisions to restrict yield from it. This approach would be unlikely to yield information of value in monitoring the stock. Even if it were to be closed for a period of five years, the late age at recruitment for this species (≈11 yr) implies that 10–20 yr will be required for the stock to show rehabilitation, and for scientists to be able to quantify it. Therefore, intensive monitoring, i.e., annually, of the stocks in the area during the closed period will not be informative. Even the catch required for a comprehensive annual survey could represent an undesirable level of fishing mortality. The present design therefore calls for intensive surveys in 1993 and 1996.

Discussion

These two experimental fisheries have provided valuable data and insight into population processes. They have also provided some confirmation that present rockfish stock assessments and management policies are appropriate. The simplest message from these two POP fisheries is that the application of fishing mortality at levels in excess of the estimated natural mortality, results in measurable and negative impacts on stock characteristics and apparent productivity.

The experiments have neither provided absolute estimates of stock biomass, nor confirmed that the absolute values of population parameters used presently in stock management are correct. Rather, the experiments have shown that the parameter values in use appear correct, relative to each other and to stock productivity. This distinction is important. For example, an estimated natural mortality rate (M) of 0.05 may be inaccurate due to ageing errors, sampling artifacts, selection by the fishery, differential availability with age, and other factors. The true value might be one-half or twice as much as this estimate. Other estimates used in population analyses are also subject to error, and any single parameter may also produce compensatory effects on other parameter estimates. However, use of the present suite of estimates in population analyses and management does produce results sufficient for sound management and development of biological insight. This corroboration is an important result of these experimental fisheries.

The experiments have also provided additional cause to question the traditional trawl survey methodology for rockfishes. They suggest that estimates of rockfish biomass from swept-area surveys should be used with extreme caution. Strong diel and tidal behavioural patterns can yield CPUE changes of two orders of magnitude at the same site over a 24 h period (Leaman and Nagtegaal 1986). In order to use swept-area surveys for such aggregated species, it is important to characterize within-site variance more thoroughly than has been done previously. This presents significant problems for surveys of large geographic areas. Analyses based on biological characteristics appear to provide a more reliable basis for conclusions about stock status. This is not to say that surveys have no value. Standardized surveys do capture large changes in stock characteristics. For example, the majority of the initial decline in the Goose Island Gully stock of POP documented by Archibald et al. (1983) with a catch-at-age analysis, was mirrored by relative changes in trawl surveys (Leaman 1985). However, there was a substantial difference in the estimated stock biomass from the two techniques, and the survey data ceased to reflect stock changes as biomass declined further (Leaman 1991).

A third result of these experiments is the finding that cooperative programs with industry can be frustrated or even destroyed by the actions of only a few individuals. When population responses to experimental manipulations are subtle, even small amounts of data contamination can significantly diminish the ability to interpret observations correctly. It was expected that industry would value the learning effects of the Langara Spit experiment, since it directly addressed their concerns that researchers underestimated abundance and productivity. However, fishermen did not provide quality data consistently, and cooperating vessel masters did not appear to pressure non-cooperating masters to adhere to the experimental program. Researchers did not develop a mechanism to guarantee that observations from the fleet on the status of stocks in the experimental area would continue to be received. The lack of a formal record on the reasons why fishermen ceased to participate in the fishery, resulted in an information gap.

Researchers failed to anticipate that the Langara Spit experiment would assume an economic life of its own. Once established, this experimental fishery came to be regarded as a necessary part of the groundfish industry. In other areas, managers were responding to declining stocks by restricting yields. It became very difficult to manage this fishery because its landings, though well in excess of the sustainable level, were 10–20% of the coastwide total of POP landings. The landings also had a localized impact because the majority went to a single port. To reduce or eliminate the Langara Spit fishery therefore carried much greater economic significance than the absolute level of landings would imply. In addition, despite declining catch rates and participation, industry initiated actions outside of the framework of the experimental program to keep the Langara area open indefinitely, and to create more open fishing experiments, without reference to the Langara results.

The interpretation of the Langara experiment differs from the perspectives of research/management and the fishing industry. As the stock declined, fewer vessels were willing to fish in the area and total effort decreased. As noted for the Vancouver Island experiment, a reduction in fishing effort on aggregating species often results in higher CPUE for the vessels still fishing. Fishery managers are then faced with conflicting arguments from scientists and fishermen. One of the lessons from these experiments is that fishery managers must be made aware of the probable results well in advance. For our experiments, several results are important in this regard.

Catch rates for aggregating species are, in general, inversely related to the amount of fishing effort. This means that

the interpretation of CPUE changes in such experiments must be considered relative to the effort expended. As real biomass declines due to fishing mortality, CPUE may be maintained or record smaller declines, if fishing effort declines proportionately. Total catches during unrestricted fishing may also exceed substantially any biomass estimates from swept-area surveys. A failure to apprise managers of these potential results can generate an erosion of confidence in other aspects of the stock assessment process. Researchers did not expend enough effort at the start of this experiment in ensuring that all levels of management were well versed in the potential results. As the experiment was to move into the second phase, the fishing industry approached senior management directly with their concerns. The resulting internal discussions required to detail the experimental results to senior management added to the delay and disorder in the implementation of the second phase. This process also tended to move the evaluation of the experiment into an arena different from that within which it had been planned and conducted.

The ability to monitor stock changes requires complete and accurate data and, conversely, is very susceptible to contamination by incorrect data. Managers and researchers should understand the limited ability to perceive changes in a biological system with high variance, even when data are fully reliable. All participants should be aware of the large cost of even small amounts of misreporting.

Lastly, the underlying biology of the target species should be considered in the design of such experiments. Our experiments concerned rockfishes because they are the major component of the trawl fishery on the west coast of Canada. However, for long-lived (30–50 yr) and late-recruiting (11–15 yr) species such as the rockfishes, the time and funding required to conduct such experiments may be a major factor in their feasibility. They are also poor candidates because the normal indices apparent to fishermen (e.g., CPUE) provide limited opportunity to observe stock changes. Species with a high rate of turnover, and for which observed indices of abundance provide adequate understanding of stock status, will be more amenable to such fishery experiments.

Several strong lessons, which may help to guide future experiments, have emerged from these two programs:

1. There should be a clear statement of objectives at the outset. These objectives should be endorsed by all levels of participants.

2. Indices and criteria for evaluation of results should also be agreed on during the planning stages. For the experiments described herein, researchers did not detail how specific changes in indices would be interpreted, rather they assumed it would be 'assessment as usual'. Their expectation, that the same indices (or changes in them) that had not been previously convincing to industry, would somehow be convincing in an experimental framework, was probably unrealistic. The interpretation that will be placed on specific types of changes to indices must be agreed upon at the outset. There should also be agreement on what actions will taken in response to the changes in indices, prior to the time when these actions are required. In addition, a mechanism to guarantee that information will be gathered from individuals who cease to participate in the experiments should be in place.

3. Agreement on the forum in which the results of experiments will be interpreted, as well as when and by whom evaluation decisions will made, should be gained at the outset. Planning of future actions, and even the conduct of ongoing experiments, will be most effective within a single management and evaluation forum.

4. There should be frequent reviews with all participants during the course of the experiment. The reviews should not only examine experimental progress but should also be used to re-confirm objectives, as well as the design and evaluation criteria.

5. A commitment from all levels of industry and government to the process and the time frame of the experiments, and to how the results may affect the design of future management programs, should be acquired. Institutional or industry impatience should not compromise the proper conduct of experiments.

6. Participants should be made aware of potential results, both biological and economic, prior to undertaking the experiments. For one of the experiments we describe, the economic impacts of removing an unrestricted fishing program, once instituted, were greatly underestimated.

Acknowledgements

We are grateful to the many fishermen who approached these experiments in a positive manner, and for their help in the design and execution of them. The experiments were conducted in conjunction with the Offshore Division of the Canadian Department of Fisheries and Oceans. In particular, we thank E. Zyblut and G. Buechler for their cooperation and assistance. Many colleagues, including W. E. Barber, M. E. Fraidenburg, V. Haist, C. L. Halbert, J. C. Rice, and L. J. Richards provided thoughtful criticisms on drafts of the paper. S. J. Smith and two anonymous reviewers helped us improve our presentation significantly.

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A Brief Overview of the Experimental Approach to Reducing Uncertainty in Fisheries Management — An Extended Abstract

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R. M. Peterman and M. K. McAllister. 1993. A brief overview of the experimental approach to reducing uncertainty in fisheries management — an extended abstract. p. 419–422. *In* S. J. Smith, J. J. Hunt and D. Rivard [ed.] Risk evaluation and biological reference points for fisheries management. Can. Spec. Publ. Fish. Aquat. Sci. 120.

Fisheries managers face substantial uncertainty about future consequences of management actions due to at least five factors.

1. Ecological responses are complex, which can easily lead to incorrect models being used to identify optimal management actions.
2. Natural variability in fish population dynamics is often high, so past responses may not adequately reflect potential future responses.
3. Data are often imprecise, biased, or not available.
4. Because of concurrent changes in one or more management actions and naturally occurring variables such as oceanographic conditions, it is often difficult to uniquely attribute an observed change in a fish population to a particular disturbance.
5. Fisheries management agencies do not often have a strong tradition of documenting reasons for past successes or failures and relevant background data for judging why decisions were made (Larkin 1972). Therefore, it is difficult to pass from one generation of managers to the next the knowledge gained through experience (Hilborn 1992a), and this perpetuates uncertainty about the potential effects of management actions.

These five causes of uncertainty lead to foregone yield or potential overharvesting. To deal with these factors, several authors have advocated a deliberate experimental approach to fisheries management (e.g., Burkenroad discussed in Skud 1975; Larkin 1972; Walters and Hilborn 1976; Holt 1977; Silvert 1978; Peterman and Routledge 1983; Lee and Lawrence 1986; Sainsbury 1988; Collie and Walters 1991; Welch and Noakes 1991; McAllister et al. 1992). However, only a few implementations of experiments exist (e.g., Sainsbury 1988; Leaman and Stanley 1993), so the evaluation of the experimental approach is in its early stages.

This extended abstract is a brief overview of this experimental method of management; more complete reviews are found in Walters (1986), Hilborn and Walters (1992), and McAllister and Peterman (1992a). Here we discuss the merits and problems of this approach in the context of the methods presented at this workshop.

Fisheries Management Actions as Experiments

Principles of experimental design (Hairston 1989) can be applied to management actions, just as they currently are used to plan laboratory experiments or monitoring and assessment programs. In the latter cases, sampling programs are designed to have a high probability of detecting changes or to give precise and accurate estimates of parameters (e.g., Doubleday and Rivard 1981, 1983). Management actions should be considered in the same context so that they will generate valuable information to refine understanding of biological and physical systems (e.g., Sainsbury 1988).

Proponents of experimental management recognize that there are constraints to implementing ideal experimental designs. For instance, there are often too few populations to create replicates or controls. However, as detailed by McAllister and Peterman (1992a), there are many historical situations in which some elements of experimental design (such as contrasting treatments) could have been implemented if the fisheries managers involved had cast their actions in terms of experiments. This also applies to many current and future situations.

Benefits of an Experimental Approach

The experimental management approach complements the various approaches to estimating uncertainty and risk that are discussed elsewhere in this workshop volume (e.g., Claytor et al. 1993; Gavaris 1993). The utility to management of estimates derived from those approaches, e.g., bootstrapping,

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jackknifing, and Monte Carlo simulation, is ultimately limited by the quality of data that they use and the validity of the statistical and biological assumptions on which they are based (Francis 1993; Punt 1993).

Several authors (e.g., Walters 1986; Sainsbury 1988; Collie and Walters 1991; McAllister et al. 1992) have demonstrated through simulation that estimation of fishery model parameters is most efficiently improved through deliberate experimental management. This is because such manipulations of regulations and fish populations will generate wider ranges of conditions more quickly and under more controlled conditions than status quo regulations and natural variation. This will lead to more confidence in selection of models and estimates of underlying parameters that describe shapes of stock-recruit relationships, density-dependent functions, and other characteristics important to fishery management (Peterman and Routledge 1983; Walters 1986; Sainsbury 1988). Furthermore, increased contrasts, carefully chosen controls, treatments, replicates, and other features of experimental design will increase the chance of correctly detecting effects of management actions on harvested populations, i.e., will increase statistical power (Peterman 1990). Experimental approaches also facilitate improved documentation and institutional learning from past successes and failures through better evaluations of those actions (Hilborn 1992a; Leaman and Stanley 1993). Though not yet shown in practice because of the paucity of applications, several authors have calculated that experimental management will reduce long-term chances of foregone catch or low stock abundance and in some cases will generate higher economic value of the fisheries than nonexperimental approaches (e.g., Silvert 1978; Sainsbury 1988; Welch and Noakes 1991; McAllister and Peterman 1992b). Clearly, deliberate experimental management can be viewed as a promising approach to improving estimation and management performance via methods presented at this workshop.

Finally, an experimental framework will use budgets effectively by avoiding uninformative combinations of actions. In contrast, lack of an experimental design may perpetuate ineffective and costly management programs (e.g., Hilborn 1992b). For instance, artificial enhancement programs for salmon populations lack formal experimental design. This may preclude drawing conclusions about the degree of density-dependence in ocean survival rates that could make enhancement cost-inefficient (Peterman 1991).

Sainsbury's (1988) experiment with trap and trawl fishing of demersal species off the northwest coast of Australia has already supported some of the above claims about the potential benefits of experimental management. At a recent workshop (Anon. 1992), Sainsbury reported that results qualitatively support the hypothesis that the most valuable demersal populations were previously depleted in part because of the destruction of macrobenthos by bottom trawling. Leaman and Stanley (1993) also report many benefits of their experiments on Pacific ocean perch (*Sebastes alutus*) in British Columbia, including (1) a qualitative support of the notion that these fish are very unproductive and have a low natural mortality rate, and (2) the current survey methods for rockfish are unreliable for estimating stock biomass.

Methods for Evaluating Alternative Designs

Some authors have compared the expected statistical performance of alternative experimental designs because not all designs are going to be equally informative. They used statistical power analysis, which helps identify necessary sample sizes (years, stocks) and levels of treatment (magnitudes of manipulations such as changes in quota or abundances of juveniles released from artificial rearing areas). In this context, power analysis ranks experimental options highest that have the best chance of correctly detecting some effect (Bernstein and Zalinski 1983; McAllister et al. 1992). Bayesian methods enable evaluation of experimental designs when there are several alternative hypotheses (Walters 1986; Sainsbury 1988; Parma and Deriso 1990; Reckhow 1990). Unlike results from power analysis, results from a Bayesian approach can be directly used in decision analyses.

Methods such as statistical decision theory (Berger 1985) can help determine whether an experimental approach has economic merit. Ranking of management alternatives is usually based on expected economic value, which takes into account the probability of different events occurring and the probability distribution of economic value for each type of event. In this context, Monte Carlo simulations (e.g., Sainsbury 1988; Walters and Collie 1989; McAllister and Peterman 1992b; Lane 1993) permit incorporation of more "real world" complexity than formal optimization procedures (such as those used by Walters and Hilborn 1976; Walters 1986; Parma and Deriso 1990; Horwood 1993). Such simulations are thus likely to be more credible to managers for their decision analyses. In some analyses (Sainsbury 1988; Collie and Walters 1991; McAllister and Peterman 1992b), the expected economic value of an experimental approach was greater than the value of the current strategy, but in others, it was not (e.g., Parma and Deriso 1990).

Problems with an Experimental Approach

There are difficulties with experimental management. For instance, the design in Sainsbury (1988) has three spatial units: a replicated treatment but only a nonreplicated control (see McAllister and Peterman 1992a for more detailed evaluation of this design). The two experiments on rockfish in Leaman and Stanley (1993) had no spatial controls or replicates. More ideal designs (e.g., Collie and Walters 1991; McAllister et al. 1992) would be critical for more powerful tests of hypotheses and estimation of population parameters. Walters (1986) and Leaman and Stanley (1993) identified other challenges of experimentation. (1) Costly monitoring may be needed in order to detect responses of fish populations to contrasting treatments. (2) A high-fishing-rate part of an experiment may be difficult to reduce later because of growing economic dependence on high harvests. (3) Institutional

planning and communication must be improved so that all parties involved (fishermen, biologists, and managers) know potential outcomes and abide by the experiment's "ground rules."

Acceptability of the Experimental Approach

Major roadblocks to implementing deliberate experimental management include the lack of experience with the approach by managers and fishermen, skepticism, and institutional inertia. These have been overcome when compelling incentives for experimentation have arisen (e.g., perceived underestimation of abundance) and have led either fishermen or managers to become advocates (Leaman and Stanley's example with British Columbia rockfish). In Sainsbury's (1988) case, experimentation was an attractive means to resolve biological uncertainty about community interactions among Australian groundfish and other processes.

Leaman and Stanley (1993) point out many difficulties with implementation but provide helpful recommendations to overcome them. However, the few existing implementations of experimental management in fisheries corroborate the theoretical work indicating that this approach has potential to improve fishery management. Management agencies should therefore more seriously apply experimental design principles to the choice of management actions and to the design of monitoring and assessment programs.

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Fisheries Management Organizations: A Study of Uncertainty

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Since the implementation of fisheries management early in this century, there has been a dramatic evolution of the structures and functions of the institutions involved in fisheries management. Nevertheless, success of these institutions has not been great, due seemingly to a variety of unrelated problems. Not least of these are problems associated with the interaction of the individuals and groups (scientist, economist, fisherman, etc.) that compose the management process. These can lead to institutional uncertainty that could surpass other more "quantifiable" sources of variability (e.g., stock assessment). Recognizing that institutional problems exist is made more difficult by the lack of consistent approach and terminology used by management authorities around the world. An idealized model of a fisheries management organization is presented which is composed of the interaction of functional (objective setting, regulation and monitoring) and structural (technical advisory and decision making groups) components. This idealized organization is then used to examine the management approach employed on Canada's East Coast. Sources of institutional uncertainty are identified and solutions proposed.

Depuis l'implantation, au début du siècle, de la gestion des pêches, on a observé une évolution spectaculaire des structures institutionnelles qui s'occupent de gestion, et de leur mode de fonctionnement. La réussite de la gestion est toutefois restée modeste, ce qui semble dû à une vaste gamme de problèmes sans lien entre eux. Parmi ces problèmes, les moindres ne sont pas ceux qui sont liés à l'interaction des individus et des groupes (scientifiques, économistes, pêcheurs, etc.) qui composent le processus de gestion. Ces problèmes peuvent causer une incertitude institutionnelle qui pourrait dépasser d'autres sources de variabilité plus «quantifiables» (p. ex. l'évaluation des stocks). Il est difficile de reconnaître l'existence des problèmes institutionnels du fait de l'absence d'une approche cohérente et d'une terminologie uniforme parmi les responsables de la gestion du monde entier. Nous présentons un modèle idéalisé d'un organisme de gestion des pêches qui consisterait en une interaction de composantes fonctionnelles (établissement d'objectifs, régulation et surveillance) et structurelles (groupes de consultation technique et de prise de décision). Cet organisme idéalisé sert ensuite à examiner le mode de gestion en vigueur sur la côte est du Canada. Nous repérons les sources d'incertitude institutionnelles, et nous proposons des solutions.

The management of fisheries resources throughout the world has been the responsibility of various national and international organizations since the early days of this century. These bodies have become very elaborate and expensive, but have had considerable difficulty in meeting stated objectives (see Kirby 1982; Finch 1985; and Frost 1984 for Canadian, US and European experience, respectively). Given the amount of time and effort that has been spent managing marine resources, it is legitimate to ask why success has been so illusive. There is a wealth of knowledge on what to do but translating this into practical reality always seems to be a problem.

It is my contention that the root cause of many of these difficulties is the lack of a systematic approach to the design of management organizations. While much historical attention has been paid to examining particular aspects (e.g., the stock assessment), this has generally been done in isolation of the potential influences on and from other parts of the organization. Although some authors (Anderson 1984; Hannesson 1984; White and Mace 1988) have endeavoured to document the interaction between harvesters and regulators, these are still only partial views. It is the interaction among all components of the management organization, not the components themselves, that is often the cause of problems. As May (1984) noted, "It is in the translation of biological understanding into effective management that things keep going wrong". Therefore, one might consider that the uncertainty generated by problems in component interactions is at least as important as that generated by activities within any one component.

This paper presents a framework for the comprehensive examination of management organizations. It first describes the functions of these and how these interrelate. This is followed by discussion on the institutional structures required to carry out the functions. Groundfish management on the east coast of Canada is then examined using the above developed conceptual framework and problems identified. Solutions to the identified problems are then presented.

A Model of A Fisheries Management Organization

Management can be defined as the control of an entity to achieve specific goals. One generally establishes what it is

Table 1. The functions of a fisheries management organization.

Function		Description
Goal Setting	-Objectives	Policy statements that are broad and publicly understood; they are valid for long time periods and can be assessed objectively.
	-Strategies	Targets based on analytical models which describe how the entity being managed will respond to control.
Control	-Regulation	Control measures required to attain strategic targets. Note that this includes both regulatory measures and their enforcement.
	-Monitoring	While normally associated with stock assessment, this function involves the monitoring of all system elements (biology, economics, etc.) to evaluate whether or not the regulation is effective in achieving the strategic targets.

that you want to manage (the entity), define its future desired state (the goals), and then manage to get there (the control).

The definition of the entity to be managed is closely associated with the structure of a management organization. When management organizations were first created, control focused on the fleets who harvested the biological stocks. The entity being managed was the fish-harvester complex. It was Anderson (1984) who first noted the necessity of considering the management regulators as part of the complex being controlled. He argued that exclusion of the regulators would result in considerable uncertainty of the outcome of fisheries management actions. In his words, "regulation is the result of interactions between government agencies and market actors, where both operate so as to maximize their individual utilities subject to appropriate constraints rather than some notion of social utility". Through the use of a deterministic 'Bioregunomic' model, he showed that inclusion of the regulators could significantly change the equilibrium levels of MSY and MEY.

While Anderson (1984) emphasized the importance in considering the regulators as part of the management process, he did not provide any analysis of what an appropriate management organization should look like. We can receive guidance in the design of management organizations from General Systems Theory (GST) (Rosen 1974a, b). The GST considers organizations as being composed of structures that carry out particular functions. This theory states that structures exhibit the 'Principle of Function Change' (Rosen 1974b) in which particular structures can exhibit one of a number of different functions, dependent on external influences. This behaviour is intrinsic to all organizations. Through the careful design of the structural components, their functions and the linkages, the extent to which "undesired" functions will occur for a particular structure can be controlled. This requires explicit definition of the functions, how these relate to each other and consideration of what functions are carried out by whom.

We will return to the issue of structure after a review of the functions of management. A clear understanding of these is necessary before considering the structure of a management organization.

The Functions

The two major functions of a management organization are to set goals and to establish controls to attain these. Halliday and Pinhorn (1985) consider goal setting to be composed of objectives and strategies. Objectives are broad, general statements on the intent of management and here are considered equivalent to policy. Strategies are the quantifiable targets used to attain the objectives. Controls are generally restricted to regulatory measures. Here, I have included monitoring activity (e.g., stock assessment) as an integral part of the control function.

I describe these two major functions below (Table 1). The intent here is not to present a comprehensive review, but rather to provide the reader with a basic understanding of the functional definitions that are used in this paper.

1. Objectives

Historically, the objectives of fisheries management have been stated in vague terms, and have not been assigned priorities. This has sometimes compromised the choice of strategic targets and has resulted in a decoupling of the objectives as stated by the policy makers from the strategies adopted by managers (Cunningham 1980; Kirby 1982; O'Boyle 1987). Part of the problem is the need for management to address the often conflicting desires of various client groups. For instance, while companies wish to maximize profits, coastal communities may want to maximize employment. Nevertheless, there is a requirement to have the objectives of fisheries management, a) based on the input of all client groups and b) prioritized for all to see (Miller 1990). This implies that objectives have to be stated in non-technical terms so that all clients can understand the basis for management.

The often stated objectives involve the fish (conservation), the industry (economic performance) and the communities (social impacts) (Fricke 1985). Failure to recognize all these objectives can have severe consequences for management performance. For instance, managing solely towards conservation goals will ultimately be self-defeating through lack of control on the economic side of the ledger. On the surface, some objectives appear contradictory (conservation vs. resource allocation; Lane 1989b), but this depends on the priorities assigned to the objectives.

The most fundamental objective of fisheries management is conservation of the fish resource. Interestingly, explicit

Table 2. Strategic models and targets currently in use in fisheries management.

Model	Conservation Target	Economic Target
Surplus Production	MSY	MEY
Yield-Per-Recruit	F_{max} (Prevent Growth Overfishing)	$F_{0.1}$
Stock-Recruitment	SSB_{min} (Prevent Recruitment Overfishing)	N/A

definitions of conservation are rare. For instance, in his analysis, Kirby (1982) focused on the economic objectives and assumed that it these were met, conservation of the resource would be achieved. In other words, fishing would become uneconomical before the stock failed biologically. Unfortunately, experience has shown that fish stocks can be driven to biological extinction before fishing becomes uneconomical. This is particularly true of schooling species such as herring where catch rates and thus profit can remain high while stock sizes decline. Sinclair (1990) recognized the need for an explicit definition of conservation for the management of Canadian East Coast fisheries and recommended adoption of the UN's statement:

> That aspect of renewable resources management which ensures that utilization is sustainable and which safeguards ecological processes and genetic diversity for the maintenance of the resources concerned. Conservation ensures that the fullest sustainable advantage is derived from the living resource base and that facilities are so located and conducted that the resource base is maintained.

Economic objectives have come to play a prominent role in fisheries management (Cummingham 1980; Kirby 1982). These have always been stated in terms of profit maximization. A unstated element of these is the need to consider protection of the market share. This in turn depends on the stability of supply to this market, a point raised by the Haché Task Force in Nova Scotia (Haché, 1989). This point becomes relevant when considering the strategies required to achieve these objectives. In the case of economics, a strategy that minimizes interannual variability in market supply may be preferable to one that does not.

Social objectives have often been stated in terms of maximizing employment (Kirby 1982). This raises the contentious issue as to where (i.e., what areas) employment is maximized. This can lead to debate on inter-fleet allocations of catch, the location of processing plants and so on, all of which involve politics. For the communities involved, these are very real issues that require explicit consideration during the objective-setting process.

2. Strategies

To achieve the management objectives, one needs to define targets (e.g., stock size, exploitation rate, profitability) that will guide management action. In other words, if the target to achieve the conservation objective is a 20% exploitation rate and stock assessment estimates the current rate to be 40%, then, through regulation, fishing exploitation has to be reduced by 50%.

The definition of a strategic target requires the development and availability of a mathematical model which comprehensively describes the behaviour of the entity being managed under different regulatory conditions. By changing the regulatory conditions, one can evaluate that set of conditions which allows attainment of the management objectives.

Strategic model building has historically been dominated by biological considerations with subsequent enhancement to include economics. The three models commonly in use are provided in Table 2, along with their most often quoted conservation and economic strategic targets.

The Surplus Production Model (Schaefer 1954) was initially developed to provide estimates of a resource Maximum Sustainable Yield (MSY). This model has been extensively used by economists to describe the cost-revenue performance of a fishery. The addition of economics to a biological model is not straightforward however, and can generate unforeseen problems (O'Boyle et al. 1991).

The Yield-Per-Recruit model (Beverton and Holt 1957) was developed to define the exploitation rate (F_{max}) which maximizes the yield from a given year-class of a fish population. This prevents growth overfishing, a biological objective. Reduction of the exploitation to a lower level would in principle generate higher revenue per unit cost than fishing at F_{max} and thus meet economic objectives. Unfortunately the model is only a crude approximation of the complex interactions involved in a fishery (Lane 1989b). For instance, holding the exploitation rate at a constant level may meet the conservation objective but result in less benefits vis á vis economic performance (Mohn 1991; Hannesson 1989) due to market supply difficulties.

Stock-Recruitment models (Beverton and Holt 1957) have been used to define the minimum Spawning Stock Biomass (SSB_{min}) necessary to ensure the long-term viability of a resource. The conservation objective for this strategy is the prevention of recruitment overfishing. Unfortunately the analysis of stock-recruitment relationships has proven to be exceedingly difficult (Miller 1990; Peterman 1990; Sissenwine and Shepherd 1988; Walters and Ludwig 1981) and thus these models have not been extensively used in the management of marine populations.

The above models are primarily biological in nature and have only secondarily considered economic objectives. None address issues of social impact (employment and resource sharing). There is therefore a need for strategic models that allow evaluation of all objectives simultaneously. Recently, there have been some encouraging efforts (Charles 1989; Lane 1989a) to develop strategic models that explicitly incorporate biological, economic and sociological considerations, but these have yet to be used by management agencies.

Strategic models are built by scientists, in a process similar to that described by Tyler (1992). Much of the data required by this process is generated by the monitoring function to be discussed below. The linkage between strategic model building and monitoring is thus very close. Scientists have generally viewed these models as a means to enhance our understanding of the complex processes involved and have been wary of their predictive capabilities. Regulators, on the other hand, are often forced to make controversial decisions which require predictions of the consequences of management actions. These conflicting requirements have often been the source of tension between scientists and regulators and may be at the root of some of the organizational problems presented later in this paper.

The link between strategies and monitoring also raises the issue of risk evaluation. There are a number of current approaches to the incorporation of risk in the management process. These generally require Monte Carlo simulation of the monitoring results to define the measurement error and then determination of the likelihood that the strategic target will be exceeded as a result of a particular regulatory action. This quantification of risk is essential for appropriate choice of control options by regulators but has been missing from many current management approaches.

Once the strategies and targets have been established, it is then necessary to control harvesting activity through regulation, to achieve the targets and thus ultimately the management objectives.

3. Regulation

Regulations are those measures established to control behaviour of the entity being managed in order to achieve the desired goals. Without regulation, there can be no management. Consequently, the importance of regulation cannot be disputed. Nevertheless, compared to other aspects of fisheries management, research on regulatory methods has been surprisingly limited. Although there has been considerable evolution in regulatory methods, this has generally been through the process of trial and error rather than the result of systematic study.

Regulation can be split into two broad categories — catch and effort controls. Catch controls limit the output of fishermen's activity. They thus control the flow of gross revenue. Initially, quotas were set for the entire fishing fleet — the so-called competitive quota. With experience, it became evident that quotas did not effectively reduce the "race for the fish" (Hardin 1968) and thus separate fleet quotas were imposed. Economists, (Hardin 1968; Keen 1983; Pearse 1982) argued that the tendency towards overcapitalization that plagues quota regulations could only be addressed if a property rights system was adopted. Kirby (1982) therefore recommended quasi-property rights for Atlantic Canada's large trawler fleet. The "right" to fish a certain proportion of the quota was allocated to each business or enterprise and therefore termed an Enterprise Allocation. In the other fleet sectors, the enterprise has been defined as the individual vessel-boat owner unit, and the "right" referred to as the Individual Transferable Quota (ITQ). Evaluation of and how effective they are in meeting management targets, has not, to my knowledge, been undertaken. They are costly to enforce and while they may facilitate the meeting of economic objectives, it is less clear what are their social consequences (Koslow 1982).

Effort controls limit a fishermen's activity and increase the cost of fishing. They include gear and vessel restrictions (gear type and mesh size) as well as limitations on fishing operation such as area and season closures. While the study of gear impacts on the resource has received considerable attention, such is not the case with regulations that influence the time and location of fishing. Indeed the notable exceptions (Allen and McGlade 1986; Crecco and Overholt, 1986; Halliday 1988) emphasize the need for models describing how and why fish move.

The above is just a sample of the regulatory tools available to the fisheries manager. Unfortunately, this variety can in itself cause problems. It is important to keep in perspective the intent of the various regulations being considered. At the very least, this will minimize the danger of implementing regulations that are either redundant or at cross purposes. One way to do this is to explicitly link a regulation with a management objective, be it conservation, economic or otherwise (Table 3). This process can also be useful in determining which regulations receive the highest funding priority, the latter set by the management objectives.

The definition of regulatory control is closely linked with the management organization's enforcement capability. Stating that log books need to be filled out accurately and having this done are two entirely different things. Needless to say, if the budget for enforcement agencies was infinitely large, virtually any regulation could be imposed. Such is not the case and therefore, the definition of regulatory controls is limited by enforcement cost. There is no sense in establishing regulations that are too expensive to enforce. Due to cost, enforcement activities are normally carried out on a sampling basis with the level of sampling inversely related to the level of imposed sanction. These relationships are complex and have not generally been the subject of examination by research groups.

A way to decrease enforcement cost is to improve the level of voluntary compliance with the regulations (Hardin 1977). It is first necessary to educate the clients with the need for regulation and then listen to their solutions to the enforcement problem. Those approaches proposed by clients have the greatest chance of being implemented. This was observed in the Southwest Nova Scotia groundfishery where there was a generally recognized need to limit the catch of

Table 3. Definition of regulatory controls in relation to management objectives.

Management Objectives	Regulatory Controls	
	Catch	Effort
Conservation	Competitive Quota (escapement level)	Gear Restrictions (size and type) Area Closures (nursery, spawning, total)
Economic Performance	Fleet/Company/Vessel Quota	Vessel Restrictions (licensing, capacity, etc.)

small fish (Haché 1989). The industry proposed the use of large, square mesh codends in their trawl nets which was accepted and implemented by Canadian fisheries authorities. To date, voluntary compliance with this regulation has been high.

The effectiveness of enforcement is also critically dependent on the process whereby an offender receives judgment on his/her actions. If courts are used, a conviction requires "beyond reasonable doubt". Responsibility is on the management agency to prove wrong doing. Surveillance activities are by their nature a highly variable process and it is often difficult to quantify "reasonable doubt" (Smith and Robert 1992). Recently, the DFO has moved toward a process whereby the Minister, not the courts, is responsible for dispensing justice. Here the onus is on the violator to prove innocence.

Closely associated with the regulation function is that of monitoring (e.g., biological assessment). If log book and landings information collected as part of the regulatory process are incorrect, then the monitoring function will be adversely affected. If monitoring is affected, then improvements to strategic models are also in jeopardy. It can be seen that problems in the regulatory function can dramatically impact the other functions of management. Let us now consider more closely the monitoring function.

4. Monitoring

The monitoring function provides the management organization with the information required to judge whether or not a strategic target, and thus objective, has been achieved. An equally important, but less obvious activity conducted in the monitoring function is that of research. This leads to improvements of our understanding of nature and how man impacts it and ultimately can lead to improved strategic models, regulations and even monitoring activities. Note here that I include all aspects of research (biology, economics, regulations, etc.) as part of the monitoring function.

The monitoring function of most current management organizations is dominated by research on the biological resource. For instance, stock assessment methodology continues to be a very active field of scientific study (Fournier and Archibald 1982; Gavaris 1988; Pope and Shepherd 1985; Sinclair et al. 1991). This is understandable as many of these agencies were initially established to conserve the fish populations. However, as other objectives have risen in importance, complementary monitoring activities have not generally been initiated. This is unfortunate for, as with strategies and regulations, monitoring activities should be conducted to assess progress of management against all objectives. To judge economic performance, ongoing surveys of the industry's profitability should be undertaken. If an objective of management is to improve employment, then the latter should be monitored.

Summary

Objective setting, strategic target establishment, regulatory control, and monitoring are the functions of a management organization. All are closely linked (Fig. 1). Objectives define the strategies. Regulations are used to attain the strategic targets. The latter are evaluated through monitoring. If the targets are not being achieved, then changes are made to the regulatory function. Through research, changes in strategic models and regulatory controls can also be made. No one function can be considered in isolation of another and problems in any one have ramifications for the others. Interestingly, much of the research conducted on the uncertainty in management has focused on a particular function, i.e., problems in strategic model or monitoring activities. It is obvious from the above, however, that the interaction among functions is as important an aspect of management and could be a significant source of uncertainty. This in turn is related to the institutional arrangements needed to implement the functions.

The next section considers the structure of an idealized management organization, taking into consideration the functions discussed above.

The Structure

There is a considerable amount of literature (Rosen 1974a, b) on organizational behaviour that can guide us in the design of administrative structures necessary to implement the management functions. For instance, the work of Karl Popper (Magee 1982) has stimulated the creation of the Centre for Institutional Studies, University of East London, devoted to research on this topic. I have used the observation of Rosen and Popper as well as my own experience in fisheries management to design what I consider to be an ideal structure.

Administrative Groups

The first issue to consider regarding the structure of a management organization is the number and mandate of administrative groups required to carry out the functions. One

could create a group with responsibility for each of the functions, i.e., an objectives committee, a strategic committee, etc. Alternatively, groups could be structured by objective conservation group, economics group, etc. The problem with either of these approaches is that both require a large number of lines of communications. As these increase, so too does the probability that there will be communications problems and that the groups will start to act independently. One committee will not understand the basis for decisions made by others. Experience has shown (Miller 1990) that such an organization could be subverted by the individuals working in it. This leads to a breakdown in the linkages among the functions necessary for effective management. Keeping the number of groups small reduces this risk and improves understanding of all involved (Lieberman 1986).

The simplest structure would consist of one organizational body, composed of representatives from all client groups, responsible for the development and implementation of all management functions. In this way, all issues would be open to the input from everyone affected by management action. Indeed, it has been shown (Fricke 1985; Peyton 1987) that it is essential that all clients contribute to management decisions. However, decisions have to be based on the best information available. The issues involved can be complex and highly technical. It is unrealistic to have one body both analysing the issues and making decisions. I advocate the creation of two groups, the first a decision-making group which defines the objectives, strategies and regulations, and the second a technical advisory group which conducts analysis on options for strategies and regulations which would be given to the decision-making group for consideration. This latter group would also conduct the work of the monitoring function. Objective setting is not a mandate of this group. This is solely within the terms of reference of the decision-makers.

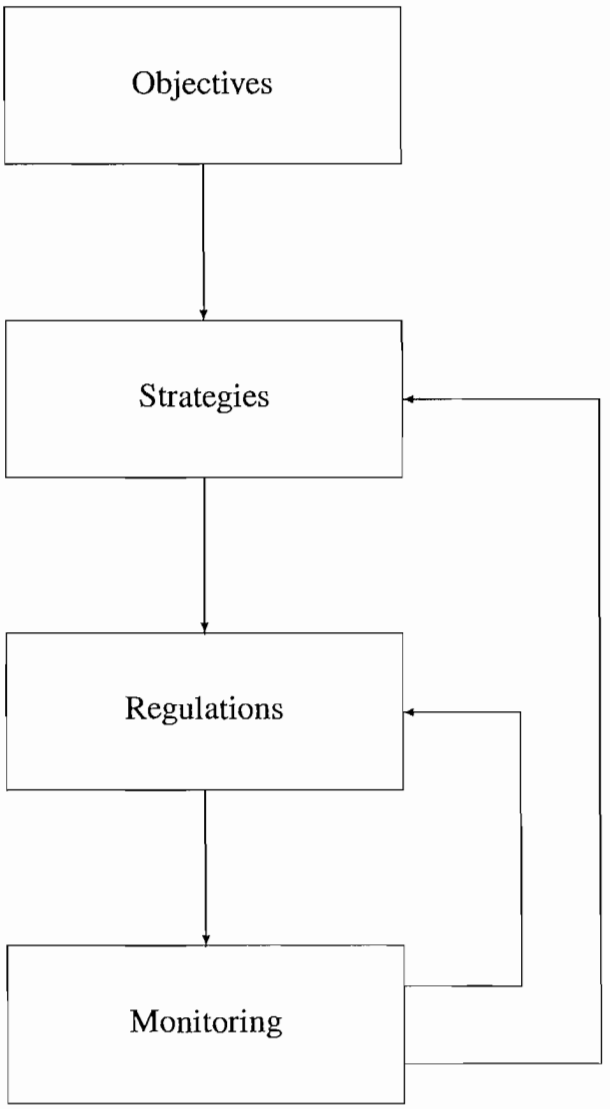


Fig. 1. Linkages among functions of a Fisheries Management Organization. Note the feedback of monitoring to both the strategic and regulatory function.

Membership and Operation

Defining the membership of the administrative groups is not a trivial question (Anderson 1984; Rosen 1974b). It strongly influences the decision-making process and thus the mix of objectives, strategies and regulations established (see for instance White and Mace (1988), for an illustration of how membership affects the regulatory package for the Nova Scotia herring fishery). Traditionally, membership has been based on natural divisions or associations already present in the fishing industry. This is to be encouraged. However, it is important to establish who these associations represent through a system of accreditation. It is not uncommon to witness debate between associations only to discover later that the protagonists represented either a very small group or indeed only themselves. Organization accreditation would inform the other members of the group what interests are being represented. As well, there would be an onus on the part of the representatives to truly represent their clients. Within the decision-making group, the process whereby decisions are made (consensus or vote) determines the nature of participation by membership. For instance, a voting system would require representatives based on population. Consensus with veto power may require representation by region or fleet sector. There are a number of options to consider in defining the decision-making process, too numerous to cover here. Each has their strengths and weaknesses and suitability for particular situations. Which approach should be adopted is a very important starting point for discussion on administrative structure. Whatever method is adopted should be accountable. A way of ensuring this is to establish a secretariat that documents all decisions of the management organization. My experience has shown that a documentation trail is one of the main tools to guarantee long-term "memory" in the organization. While individuals change, documents which record previous decisions do not.

The technical advisory group has to be composed of individuals skilled at addressing the technical needs of each client

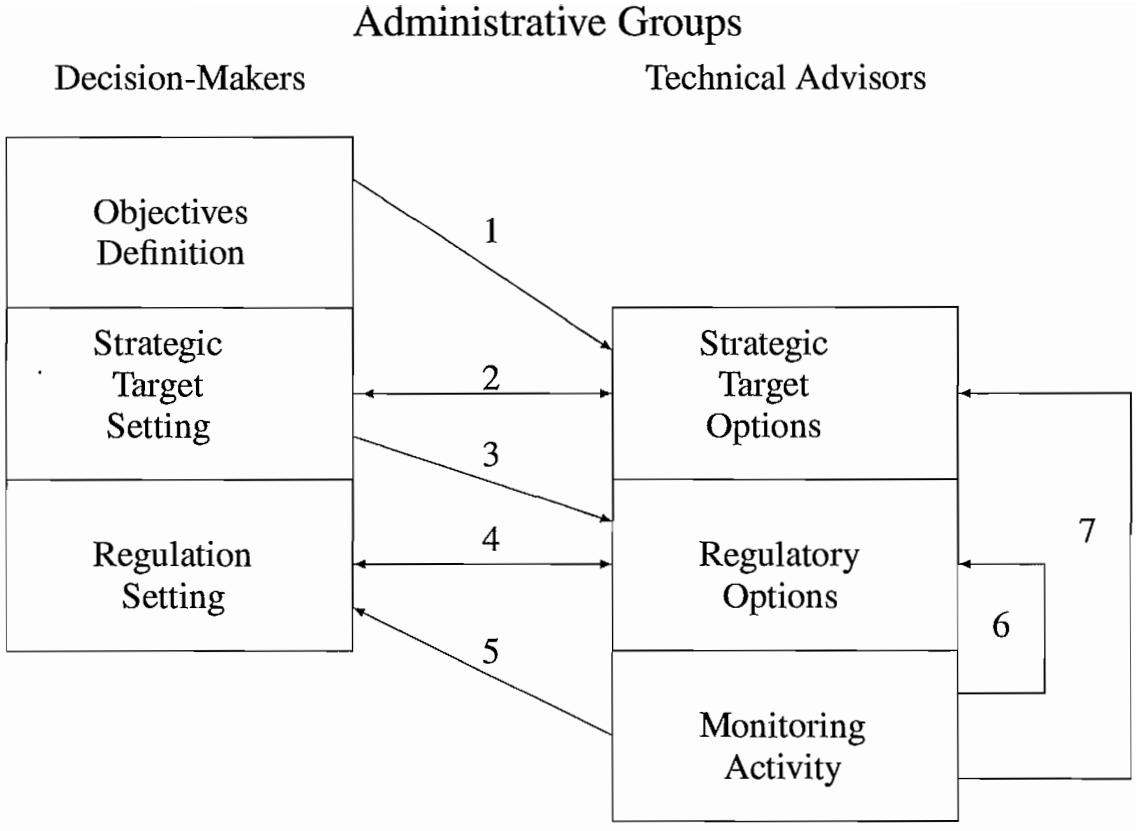


Fig. 2. Institutional arrangements of an ideal Fisheries Management Organization. Arrows represent linkage and direction of information flow. The lines between boxes indicate information flow. The numbers are explained in the text.

group, which is sometimes very difficult to ensure. This is particularly true of fishermen when it comes to biological stock assessment. In these cases collaboration between the biologists and members of the fishing industry to facilitate joint studies and analyses can go a long way towards ensuring that fishermen's observations are discussed in the technical group. Ultimately though, technical discussion has to be conducted within an organization which draws upon the greatest base of expertise possible in a peer review forum. International scientific organizations, such as ICES, operate in this mode and are a good model to use in establishing technical groups.

Link Between Structure and Function

An overview of the relationship between structure and function is given in Fig. 2.

At the on-set of management, the decision-making group would develop the prioritized list of objectives. This could be through a broad-based consultative process (e.g., public hearings). Once these are available, they would be given to the technical group (arrow one, Fig. 2) for it to develop the necessary strategic models, targets and options. There would then be dialogue between the two groups to allow informed decision-making on the appropriate strategic options to adopt (arrow two, Fig. 2). Having decided this, the technical group would then develop a package of regulations necessary to achieve the strategic targets (arrow three, Fig. 2). Again, there would be dialogue between the two groups before the decision-makers would approve a final regulatory package (arrow four, Fig. 2). Once this is established, management performance is monitored by the technical group and the status provided to the decision-makers on an ongoing basis (arrow five, Fig. 2). As well, when necessary, the regulations (arrow six, Fig. 2) and strategic models (arrow seven, Fig. 2) would be updated, again with the approval of the decision-makers.

The above is an idealized model. In the next section, the Canadian East Coast groundfish management system will be compared to this ideal to see where problems may exist and allow proposal of solutions.

The Canadian East Coast Groundfish Management Organization

Canada's East Coast groundfish resources have been under some form of management since the early days of this century. During the 1950's and 1960's, the objective was stock

Table 4. An overview of the functions of Canada's East Coast Groundfish Management Organization.

Function	Client		
	The Fish (Conservation)	The Industry (Economic Performance)	The Community (Employment)
Objectives	None identified	The Atlantic fishing industry should be economically viable on an ongoing basis.	Employment in the Atlantic fishing industry should be maximized subject to the constraint that those employed receive a reasonable income. Fish within the 200 mile Canadian Zone should be harvested and processed by Canadians.
Strategies	F_{MAX} SSB_{min}	$F_{0.1}$ and 50% rule	None identified
Regulations			
Catch Control	None identified	Three-year plan Competitive Quotas (fixed gear) Boat/Enterprise Quotas (mobile gear)	
Effort Control	Gear Type/Size Nursery and SpawningClosures	In-Season Closures Trip Limits Vessel Capacity Guidelines	Limited Entry
Monitoring	Biological Stock Assessment	Industry surveys	None identified

conservation, this being pursued through an F_{max} or MSY strategy. Regulations were effort-based being predominantly gear type and size restrictions. During this period, the International Commission for North Atlantic Fisheries (ICNAF) was the organization responsible for management. By the late 1960's, it became evident that further regulations would be required if the conservation objective was to be attained. Consequently quotas were introduced in 1970 in an attempt to limit exploitation. As well, spawning area closures were introduced for the haddock stocks.

In 1977, Canada extended its territorial boundary to 200 miles. This harkened a new era of management in which domestic fishery agencies became responsible for resource husbandry. Economic objectives were stated and pursued through a strategy of $F_{0.1}$. However, since extended jurisdiction, the regulatory package has grown in complexity. What started as a competitive quota system has evolved into one dominated by enterprise and boat quotas. Effort-based regulations have also been added (e.g., vessel size limits, trip limits, fish landed size). In spite of this, exploitation rates are in many cases higher than the $F_{0.1}$ strategic target (Anon. 1992).

Let us now examine the function and structure of the current management process using the model outlined in the previous section.

The Functions

An overview of the functions is given in Table 4. The functions are considered by client. The fish are considered a "client" of management and the objectives, strategies, regulations and monitoring of interest to this client presented. Similarly, the fishing industry and community are represented as clients and activities in relation to the four functions presented. It should be noted that the other users of the marine environment are not included here. Specifically, there are developing recreational interests and other users of the habitat. These are at an early stage of development for which little, if any, management has as yet been enacted.

No conservation objectives have been explicitly identified. It has been assumed that these would be attained as part of the objectives associated with economic performance. This is unfortunate because, as stated earlier, short term profit maximization can lead to long-term conservation difficulties (Mahon et al. 1985). In his review of the East Coast's management process, Sinclair (1990) recognized the need for a statement on conservation and suggested adoption of the UN definition described earlier. This has yet to be adopted by DFO.

Kirby (1982) evaluated the other objectives of management and found them too vague to be useful. Consequently, as a result of an extensive series of consultations and public hearings, he established a prioritized set of objectives which have subsequently been the basis for management. He was careful to fully describe each objective to leave no misunderstanding of his intent. He considered economic viability as the number one priority and therefore considered the industry's concerns of paramount importance. The next two objectives — employment maximization and "Canadianization" — were priorities two and three respectively and were generally intended to improve long term community viability.

The current strategies are developments of those used in the 1960's by ICNAF. At that time, conservation was the main objective and consequently the strategies were based on the Yield-Per-Recruit and Surplus Production Model, the targets of exploitation being F_{max} and Maximum Sustainable Yield

(MSY) respectively. ICNAF realized that there were conservation and economic benefits to limiting fishing effort below F_{max} or MSY and developed the strategy of $F_{0.1}$ (Gulland and Boerema 1973) to achieve these. This was adopted by Canada in 1977 when it took over management of the coastal fisheries as a result of extended jurisdiction. It had been assumed that $F_{0.1}$ would be enough to ensure long term stock conservation. However, by the mid-1980's it became apparent that more was required. Therefore in 1987, the Groundfish Management Plan (Anon. 1986) was amended to include the following statements:

> If the stock assessment provides evidence of levels of spawning stock biomass likely to endanger recruitment, fishing effort (and thus fishing mortality) in the coming year be reduced to allow immediate growth in spawning stock biomass.
>
> In addition, if F_{next} (fishing mortality in coming year) is higher than twice $F_{0.1}$, F_{next} is set to twice $F_{0.1}$ to prevent growth overfishing. The twice $F_{0.1}$ level is proposed as an approximation to F_{max} due to difficulties in reliably estimating the latter.

To date, there has been no quantification of a minimum spawning stock biomass to prevent recruitment overfishing.

It is worthwhile to note that the strategy of $F_{0.1}$ was not developed to meet the objectives of Kirby (1982) which in fact were stated much later. Rather, $F_{0.1}$ was employed in a new role. Nevertheless, it has only recently been evaluated (Lane 1989b) in terms of its economics and has been found deficient in a number of areas. A critical missing element is the lack of consideration of the effects of interannual variability in yield on profitability, and market share. This was raised as an issue by the Haché Task Force (Haché 1989). One source of variability was created by DFO's attempts to reduce fishing effort from above $F_{0.1}$ to $F_{0.1}$ in one year. This was generally recognized as disruptive to fishing operations and thus a phase-in strategy was adopted in 1987. This strategy was implemented when the quota advice between two years varied by more than 10%. It stated:

> If the spawning stock biomass is high enough as to not be of concern, the fishing mortality in the coming year of the plan would be set at a value halfway between the current fishing mortality and $F_{0.1}$ according to:
>
> $$F_{NEXT} = F_{CURRENT} - \frac{(F_{CURRENT} - F_{0.1})}{2}$$
>
> where the F_{NEXT} is the fishing mortality for the coming year of the plan and $F_{CURRENT}$ is the fishing mortality estimated for the most recent year of the current assessment.

This has come to be known as the 50% rule. Although two of the main objectives of management are related to employment, no strategic targets have been identified. In other words, there are no defined minimum levels of employment or maximum levels of plant capacity to guide management action.

The current regulatory package is complex and is characteristic of a system attempting to limit the effort of an overcapitalized fishing fleet. Conservation-directed regulations include limits on trawl codend mesh size (to 145 mm diamond mesh), fish landed size (to 17 inches) and various closed areas and seasons to protect spawners and juveniles. Prior to 1985, global fleet or competitive quotas were used to limit overall exploitation and thus achieve the economic objectives. As elsewhere in the world, these proved inadequate in limiting overcapitalization and resulted in quota overruns, disputes among gear sectors and so on. This has been replaced by an enterprise quota system for the large companies (1985) and individual boat quotas for the independents (1991). The fixed gear fleet is still regulated by in-season closures, competitive quotas, trip poundage limits and vessel capacity guidelines. To further minimize interannual variability in landings, quota allocations are provided for a three year period. These are only changed if the biological assessment indicates a critical resource situation.

Regarding employment in the fishery, no new licenses are being issued. New entrants must acquire inactive licenses to legally fish. Many of these inactive licenses still exist, despite repeated attempts by DFO to retrieve them, which have always met with considerable opposition. This is indicative of the deep-rooted attachment to fishing that coastal communities have and the desire to keep the fishing career option open. With the implementation of IQ's, licensing is becoming a non-issue as access is controlled by who has the quota, not who has the right to fish.

Enforcement of this complex regulatory package has been costly. There are current attempts to have industry pay a share of these costs although these have met with opposition. There is a need to institute education programs in the communities on the basic principles of and the need for management, so that the level of voluntary compliance may be increased.

Monitoring effort has focused on biological stock assessment. Port and sea sampling programs have been established as well as a logbook system to record catch and effort information. Unfortunately, the quality of monitoring information (e.g., catch and effort data) is severely compromised by the regulatory function. Lack of consensus on the regulatory package has led to enforcement difficulties which in turn has produced problems in fishery data collection.

Various annual surveys are being conducted to monitor population trends and environmental parameters. Active research programs on population dynamics and modelling are ongoing. Research on improvements to currently used strategic models has however, been limited. The basis for management are still the Yield-Per-Recruit and Surplus Production Models.

Monitoring of economic performance or employment trends is limited. A notable initiative is the tri-annual economic survey of specific fleet sectors conducted in Nova Scotia (Anon. 1985). This survey has produced much valuable

information on fleet performance and has been used to evaluate current management practices (O'Boyle et al. 1991).

Overall, while the objectives are well documented (except for conservation), there is little continuity between these and the other functions of management. In some cases, the links are strong (e.g., regulation and biological monitoring). Nevertheless, there are significant gaps that endanger the capability of management to fulfil its mandate.

The Structure

The final authority for decision-making on management actions resides with the Minister of Fisheries and Oceans. The latter obtains information from a number of sources before making a decision. In comparison to the idealized model, the Minister represents the decision-making organization. For many decisions however, a government/industry group called the Atlantic Groundfish Advisory Committee (AGAC) is consulted to determine the option most acceptable to the clients of management. This body is composed of representatives of the fishing industry and various levels of government. There are no representatives of the communities or general public. It is chaired by a DFO senior official and is generally considered to be a DFO consultative group. The recommendation of the AGAC are further considered by DFO who then make recommendations to the Minister.

The DFO East Coast organization is split into four administrative regions, these being Newfoundland, Scotia-Fundy, Gulf and Quebec. In each of these, a Groundfish Advisory Committee (GAC) similar to AGAC is used to obtain industry input on management issues. These groups do not officially report to the Atlantic group. Rather, representatives of the regional bodies also participate in AGAC.

The issues discussed by the GAC's are predominantly regulatory in nature. The allocation of quotas receives the most attention although in recent years there has been discussion on effort-based regulations. Technical input on these issues is received either directly from DFO staff or from the technical groups to be discussed below.

Membership of the GAC's is based on the traditional divisions within the fishing industry, these being the large fish companies, mid-sized independent processors, mobile gear fishermen and fixed gear fishermen. The latter are composed of a number of different associations varying in size and representativeness. There are current efforts to encourage the development of more representative organizations, through an accreditation process. Problems persist in communication between the association representatives and the grass roots.

Development of advice by the GAC's is in principle by consensus although this often is not achievable. DFO takes all the discussion into consideration before making recommendations to the Minister.

There is no secretariat for the Advisory Committees, although DFO staff record and circulate minutes to participants. This is unfortunate in that it is very important to have administrative structures that provide long term continuity to decision-making — a secretariat would provide such continuity.

Recommendations on Total Allowable Catches (TAC's) are provided to the Minister and AGAC by the Canadian Atlantic Fisheries Scientific Advisory Committee (CAFSAC) and the Northwest Atlantic Fisheries Organization (NAFO). Both are successors to ICNAF which was disbanded in 1977. CAFSAC conducts analyses on those resources solely within Canadian jurisdiction while NAFO those considers resources off the Atlantic Coast which are of international fishing interest. Both organizations handle predominantly technical issues and from this point of view are equivalent to the technical advisory group presented in the idealized model.

CAFSAC and NAFO restrict their analyses and recommendations to the impact of harvesting on the resources. Economic and employment issues are not generally considered. As requested, advice on technical issues is provided to the GAC's and the Minister for further consideration.

Both CAFSAC and NAFO are composed primarily of scientists with a predominance of biologists. Input from industry is limited to the provision to catch/effort information through normal data processing channels. Consultation with industry is often conducted outside of CAFSAC and NAFO. Unfortunately, these tend to be limited due to the difficulties in obtaining technical input from fishermen who operate under a complex regulatory package.

CAFSAC and NAFO operate through a scientific peer review process, with a well established document trail and secretariat similar to that of ICES. Their structures have been emulated by other international agencies wishing to improve the quality of technical advise on fisheries matters.

The Link Between Structure and Function

An overview of the interaction of structure and function for Canada's East Coast groundfish management is provided in Fig. 3.

The objectives were developed outside the GAC organization by the Kirby task Force (Kirby 1982). To my knowledge, these have not been formally presented to the technical group for the purposes of strategy development. I consider this to be a significant gap in the management organization. There is thus no guaranty that the existing strategies will achieve the objectives of Kirby (1982).

There has been dialogue between the GAC's and CAFSAC/NAFO on changes to the strategies (Fig. 2, arrow two). This has, for instance, resulted in the 50% rule. Overall, however, the interaction has been limited and therefore this link is considered weak.

Once the strategic targets have been agreed to, the technical groups have responded by providing regulatory options necessary to achieve these (Fig. 3, arrow three). For instance, the quota implications of the 50% rule are a standard feature of technical advice. On the other hand, the technical group has been unsuccessful in dissuading the use of spawning closures as a means of protecting spawning stock biomass (Halliday 1988), which has received wide spread support without scientific rationale.

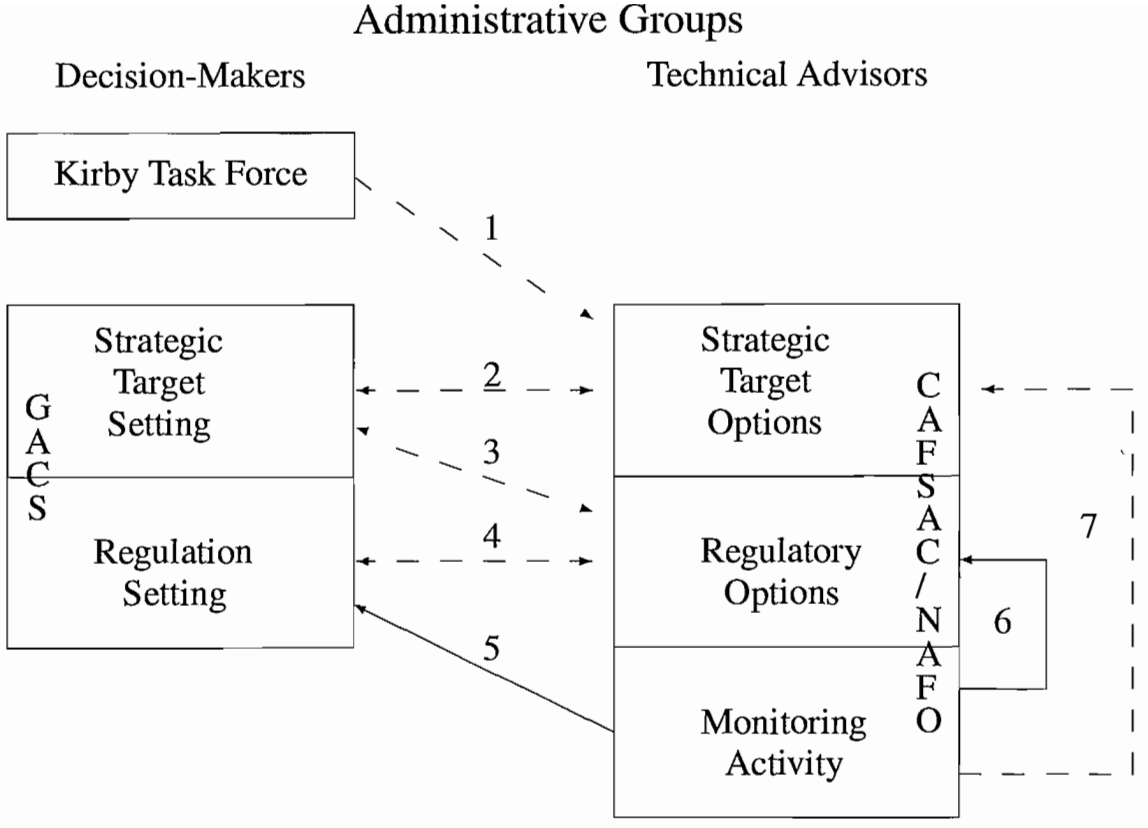


Fig. 3. Institutional Arrangements of Canada's East Coast Groundfish Management. The lines between boxes indicate information flow, with solid lines indicating strong linkage and dashed lines a weak linkage. The numbers are explained in the text.

This problem surfaces whenever regulations proposed by the industry are considered (Fig. 3, arrow four). For instance, the Haché Task Force recommended changes to existing mesh size regulations as a conservation measure. These were adopted by DFO without consideration by CAFSAC. These recommendations were subsequently modified to address the concerns of some sectors of the industry. This could have been avoided by more informed technical input at an earlier stage. Again, in 1989, the Scotia-Fundy GAC adopted multi-species catch quotas to avoid enforcement problems, without an impact analysis conducted by CAFSAC. Further, in 1985 and 1991 the boat/enterprise quota regulation was adopted without formal evaluation on its potential impacts. There is thus room for improvement of the linkage between decision-makers and technical advisors as far as regulations are concerned.

The feedback from the technical groups on monitoring is strong (Fig. 3, arrow five) although it is restricted to biological issues. The GAC's have come to respect the statements made on resource status which has provided the support for regulatory change (Fig. 3, arrow six). On the other hand, as stated earlier, the update of strategic models based on research (Fig. 3, arrow seven) has been slow.

Summary

The structure, function and linkages of the Canadian East Coast management organization are the result of an evolutionary process which started with extended jurisdiction in 1977. As a consequence, some elements are stronger than others while some require considerable improvement. By taking a systematic view of the entire process, the strengths and weaknesses are easy to identify. In the next section, I present some solutions to the identified problems which will hopefully lead to less uncertainty in organizational behaviour and therefore more predictable management.

Solutions

The Functions

In regards to objectives, there is a need for an explicit definition of conservation. Of all the objectives, this should be given the highest priority. Regarding the other objectives, in my opinion they are well defined and adequate to guide current management efforts.

A review of the existing strategies is required to ensure that these are adequate to meet the stated objectives. There is already some evidence to show (Lane 1989b) that $F_{0.1}$ may not be adequate. This needs further examination. This could be used as an opportunity to investigate models that simultaneously consider all objectives (e.g., Charles 1989).

There is a requirement to examine the regulation package in its entirety, rather than consider each regulation in isolation as has been previous practice. In this way, the compatibility of regulations can be evaluated. Also they could be related to the various objectives of management and their enforcement costs evaluated and prioritized accordingly. Work is also required to improve the enforceability of the regulations. Analyses are needed to define the interaction among deterrence, cost and penalty so that the enforcement budget can be optimized. Lower enforcement costs can also be sought in the longer term through education programs aimed at improving understanding and thus hopefully leading to higher levels of voluntary compliance with regulations.

Activities to monitor the economic performance of the fleets and the employment profiles in the communities are needed. This involves enhancement of the existing economic surveys, as well as initiation of new programs.

Efforts to improve the strategic models are to be encouraged. Where possible, these should consider the interaction of biological, economic and social processes.

The Structure

There is a need to modify the current Advisory Committee structure so that the regional groups report to the zonal body. The terms of reference of these organizations requires review as does its membership and meeting protocol. To be effective, this body should reflect the needs of all client groups and have the authority to make decisions. It could be autonomous from DFO but with veto power over decisions residing with the Minister. The group would require a secretariat to provide a long term record of decision-making.

The existing technical groups, CAFSAC and NAFO should continue providing analyses as required, however some changes would improve their input. It is desirable to have discussion and input on technical matters as broad as possible. This would be facilitated by combining the scientific aspects of CAFSAC and NAFO into one group. This group could also address technical issues raised during many bilateral discussions (e.g., Canada/USA, Canada/France, Canada/Japan) that have arisen in recent years. Such a body could fulfil a role similar to that of ICES in the North Eastern Atlantic. Such a proposal has already been made by DFO regional staff (R. Halliday, DFO, Dartmouth, N. S. pers. comm.).

Work in the technical group should be expanded to include consideration of economic and social issues. What work is being conducted in relation to the latter is generally being reviewed outside of CAFSAC/NAFO. Having technical discussion on all management issues conducted in one body would have the additional benefit of encouraging multi-disciplinary study and research.

Greater emphasis needs to be placed on providing communications avenues between the technical groups and members of the fishing industry. This could be facilitated by establishing collaborative studies which could then provide a nucleus for wider participation as interest and expertise grows. In most current management organizations, the data on harvesting activities provided by fishermen is analysed by scientists to determine the resource status and thus recommend regulatory actions. In the eyes of fishermen, provision of logbook information to scientists often translates into reduced quotas, not the sort of feedback that inspires science/industry cooperation. Closer collaboration would provide the means for more positive feedback and generate greater understanding for the resource, scientific requirements and the need for management actions.

An administrative mechanism used to encourage science/industry collaboration in other countries (e.g., Australia) is a cost-shared research fund. All clients of management would contribute to the fund which would be managed by the decision-making group. On an annual basis, issues and studies that need funding would be identified and requests for proposals circulated. The decision-making group would than distribute the funding after an evaluation process. Such a fund would make the industry a vested interest in research and go a long way towards fostering industry/scientist interaction.

The Link Between Structure and Function

The problems in communication between the decision-makers and technical advisors have been presented above. Many of these can be very simply addressed by clearly defining who is responsible for what and when. It is imperative that before strategic targets and regulatory measures are accepted, impact analyses by the technical group are conducted. There has to be continuing dialogue between the decision-makers and the technical advisors to ensure effective implementation of management action. As long as this simple rule is adhered to, problems in communication can be avoided.

Concluding Remarks

Current fisheries management organizations are by necessity complex structures required to meet the needs of resource harvesting. Generally they have evolved into their current state and have not benefited from reference to an overall design or plan. Experience has shown that considerable uncertainty can been introduced into management actions solely as a result of the organization's behaviour. This paper presents the structure and function of an ideal management organization to which the Canadian East Coast groundfish situation is compared. A number of problem areas are identified and potential solutions proposed. These are general in nature and are provided in the spirit of initiating broader dialogue on this topic.

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Appendix A: List of Participants

R. Arnason (Iceland)
M. Basson (England)
J. Beckett (Canada)
M. Bergh (South Africa)
B. Bogstad (Norway)
M. Chadwick (Canada)
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W. Clark (U.S.A.)
R. Claytor (Canada)
L. Coady (Canada)
R. Conser (U.S.A.)
W.G. Doubleday (Canada)
S. Farlinger (Canada)
R. I. C. C. Francis (New Zealand)
W. Gabriel (U.S.A.)
P. Gagnon (Canada)
S. Gavaris (Canada)
J. Glaister (Australia)
C. P. Goodyear (U.S.A.)
V. Haist (Canada)
M. Hildén (Finland)
J. Horwood (England)
J. J. Hunt (Canada)
T. Jacobsen (Norway)
R. Kronlund (Canada)
P. Kunzlik (Scotland)
D. Lane (Canada)
B. Leaman (Canada)
M. K. McAllister (U.S.A.)
P. Mace (U.S.A.)
J.-J. Maguire (Canada)
R. K. Mohn (Canada)
J. B. Morrissey (Canada)
R. A. Myers (Canada)
R. N. O'Boyle (Canada)
H. Palsson (Canada)
D. Pelletier (France)
R. Peterman (Canada)
E. Pikitch (U.S.A.)
A. T. Pinhorn (Canada)
J. Powers (U.S.A.)
A. Punt (South Africa)
V. Restrepo (U.S.A.)
L. Richards (Canada)
D. Rivard (Canada)
A. Rosenberg (U.S.A.)
L. Savard (Canada)
P. Shelton (Canada)
R. Shotton (Canada)
A. Sinclair (Canada)
M. Sinclair (Canada)
A. Smith (Australia)
S. J. Smith (Canada)
S. I. Steinshamn (Norway)
R. Stephenson (Canada)
G. G. Thompson (U.S.A.)
S. Tjelmeland (Norway)
D. Vaughan (U.S.A.)
X. Xu (Canada)
K. C. T. Zwanenburg (Canada)

Appendix B: Workshop Schedule

Schedule for
Risk Evaluation and Biological Reference Points
for
Fisheries Management

Convener: S.J. Smith (DFO, Dartmouth, Canada)

Tuesday, November 19, 1991.

08:30. Registration.

08:50. Welcome and General Information.

9:00. Opening Address: J.B. Morrissey, ADM Science, DFO

Session 1. Reference Points: current usage, track record, strengths and weaknesses (with the aim of providing an overview of experiences so far).
Chair: J.J. Hunt (DFO, St. Andrews, Canada).

09:40. Reference Points for Fisheries Management: The Western Canada Experience.
B. Leaman (DFO, Nanaimo, Canada).

10:00. Reference Points for Fisheries Management: The Eastern Canada Experience.
D. Rivard (DFO, Ottawa, Canada) & J.-J. Maguire (DFO-CAFSAC, Québec, Canada).

10:20. Coffee break.

10:40. Use of spawning stock size consideration in providing fishery management advice in the North Atlantic — A brief Review.
C. Goodyear (NMFS, Miami, U.S.A.).

11:00. Overfishing in the U.S. fisheries and evaluation of spawning stock biomass per recruit criteria.
P. Mace & M. Sissenwine (NMFS, Silver Spring, U.S.A.).

11:20. Evaluation of Gulf of Mexico King Mackerel stock assessment research: Benefits and costs to management.
J. Powers & V. Restrepo (NMFS, Miami, U.S.A.).

11:40. The behaviour of F_{low}, F_{med} and F_{high} in response to variation in parameters used for their estimation.
T. Jakobsen (IMR, Bergen, Norway).

12:00. Lunch

13:30. Reference Points for Fisheries Management: The ICES Experience.
M. Hildén, (National Board of Waters and the Environment, Helsinki, Finland).

13:50. The Barents Sea capelin collapse: A lesson to learn.
S. Tjelmeland & B. Rogstad (IMR, Bergen, Norway).

14:10. The development of a management strategy for the South African Anchovy resource.
D.S. Butterworth & M. Bergh (University of Cape Town, South Africa).

14:30. Session I discussion and wrap-up.

15:00. Coffee break.

Session 2. Identifying and Quantifying Uncertainties, Risks and Costs.
Chair: D. Rivard (DFO, Ottawa, Canada).

15:20. Monte Carlo Evaluation of the Risk for Biological Reference Points used in New Zealand Fishery Assessments.
R.I.C.C. Francis (MAF, Wellington, New Zealand).

15:40. Using jackknife and Monte-Carlo simulation techniques to evaluate forecast models.
R. Claytor, G. Nielsen (DFO, Moncton, Canada) & P. Shelton (DFO, St. John's, Canada).

16:00. Assessing the impact of sampling error on model-based management advice: Comparisons of equilibrium yield per recruit variance estimators.
D. Pelletier, (IFREMER, Nantes, France).

16:20. The utility of variance estimators for fisheries assessment and the performance of alternative variance estimation procedures for some assessment methods.
A. Punt (University of Cape Town, South Africa).

16:40. Model-Based estimates of uncertainty for the projected catch.
S. Gavaris (DFO, St. Andrews, Canada).

18:00–20:30. Mixer at Chateau Halifax.

Wednesday, November 20, 1991.

09:00. Incorporating uncertainties from research surveys: Bootstrap resampling versus random number simulation.
S.J. Smith (DFO, Dartmouth, Canada) & S. Gavaris (DFO, St. Andrews, Canada).

09:20. The variance-bias tradeoff when using catch-age models to calibrate indices of abundance — which entails more risk?
R. Conser (NMFS, Woods Hole, U.S.A.).

09:40. From uncertainties to risks.
J. Hoenig (DFO, St. John's, Canada) & V. Restrepo (NMFS, Miami, U.S.A.).

10:00. Bootstrap estimates of ADAPT parameters and their projection in risk analysis.
R.K. Mohn (DFO, Dartmouth, Canada).

10:20. Coffee break.

10:40. Risk evaluation and biological reference points for capelin management in NAFO Division 3L.
P. Shelton, J. Carscadden & J. Hoenig (DFO, St. John's, Canada).

11:00. A comparison of event tree risk analysis to spawner-recruit simulation: An example with Atlantic Menhaden.
D. Vaughan (NMFS, Beaufort, U.S.A.).

11:20. Session II discussion and wrap-up.

12:00. Lunch.

Session 3. Possible strategies, including strategies which incorporate uncertainties: where are we going?
Chair: A. Sinclair (DFO, Mont Joli, Canada).

Biological Strategies

13:30. Risks and uncertainties in the management of a single-cohort squid fishery: the Falkland Islands Illex fishery as an example.
M. Basson & J.R. Beddington (Imperial College, London, UK).

13:50. Groundfish exploitation rates based on life history parameters.
W.G. Clarke (IHPC, Seattle, U.S.A.).

14:10. Surplus production for cod (*Gadus morhua*) in the Southern Gulf of St. Lawrence, NAFO Div. 4T-4Vn (Jan.–Apr.), southwest Nova Scotia, NAFO Div. 4X and northeast Newfoundland, NAFO Div. 2J-3KL with implications for stock management.
W.G. Doubleday (DFO, Ottawa, Canada), G. Chouinard (DFO, Moncton, Canada), S. Campana (DFO, Dartmouth, Canada) and J. Baird (DFO, St. John's, Canada).

14:30. The comparative performance of a number of feedback-control management procedures for the stock of Cape hake off the west coast of South Africa.
A.E. Punt (University of Cape Town, South Africa).

14:50. A proposal for a threshold stock size and maximum fishing mortality rate.
G.G. Thompson (NMFS, Seattle, U.S.A.).

15:10. Coffee Break.

15:30. Stochastic optimal harvesting from a multi-cohort stock with non-linear dynamics.
J.W. Horwood (MAFF, Lowestoft, UK).

15:50. Possible Management Targets for Canadian Atlantic Coast Gadoid Stocks.
J.-J. Maguire (DFO-CAFSAC, Québec, Canada).

16:10. Use of SSB/R-based reference points in management of New England groundfish.
W. Gabriel (NMFS, Woods Hole, U.S.A.).

16:30. Management strategies for a complex of stocks.
Laura J. Richards (DFO, Nanaimo, Canada).

Economic Considerations

16:50. Bioeconomic reference points and associated risk analsysis.
B. Kaufmann & H. Palsson (DFO, Ottawa, Canada).

17:10. Market based Fisheries Management.
R. Arnason (University of Iceland, Reykjavík, Iceland).

Thursday, November 21, 1991.

09:00. Bioeconomic impacts of TAC adjustment schedules: The case of Northern cod.
D. Lane (Université d'Ottawa, Ottawa, Canada) & B. Kaufmann (DFO, Ottawa, Canada).

Experimental Management

09:20. Experimental design in the management of fisheries: a Review.
M.K. McAllister & R.M. Peterman (Simon Fraser University, Burnaby, Canada).

09:40. Experimental Management programs for two rockfish stocks off British Columbia, Canada.
B.M. Leaman & R.D. Stanley (DFO, Nanaimo, Canada).

10:00. Coffee Break.

Control Mechanisms

10:20. Fixed or variable catch quotas.
S.I. Steinshamn (Norwegian School of Economics and Business Administration, Bergen, Norway).

Evaluation

10:40. Fisheries Management Systems: A Structure for the examination of uncertainty.
R.N. O'Boyle (DFO, Dartmouth, Canada).

11:00. Choosing a management strategy for stock rebuilding when control is uncertain.
A.A. Rosenberg & S. Brault (NMFS, Woods Hole, U.S.A.).

11:20. What to do when SPA fails.
K.C.T. Zwanenburg & K. Frank (DFO, Dartmouth, Canada).

11:40. Evaluation of the impact of recruitment uncertainty on Pacific herring assessment and management.
V. Haist & J.F. Schweigert (DFO, Nanaimo, Canada).

12:00. Lunch.

13:30. Risks of over- and under-fishing in developing new resources.
A.D.M. Smith (CSIRO Division of Fisheries, Hobart, Australia).

13:50. Estimating the risk of ignorance.
E. Pikitch, M.K. McAllister & R. Hilborn (University of Washington, Seattle, U.S.A.).

14:10. Session III discussion and wrap-up.

15:00. Coffee Break.

15:20. Working Group Meetings.

Friday, November 22, 1991.

09:00. Working Group Meetings.

12:00. Lunch.

13:30. Plenary session.

Appendix C: Referee List

The editors thank the following for giving their time and expertise to referee the manuscripts submitted to this volume.

M. J. Armstrong
W. Barber
J. Carscadden
A. T. Charles
J. Collie
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D. Gray
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D. C. Hamilton
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S. I. Steinshamn
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L. Van Eeckhaute
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