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MIKAEL HILDÉN

Risk, uncertainty, indeterminacy and ignorance in fisheries management – an analysis of management advice

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5

Mikael Hildén

Risk, uncertainty, indeterminacy and ignorance in fisheries management – an analysis of management advice

Sammandrag: Risk, osäkerhet, obestämdhet och okunskap i fiskeriförvaltning – en analys av hur forskningen stöder förvaltningsbeslut

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Contents

Summarised publications

This thesis is based on the following papers, which will be referred to in the text by their Roman numerals:

I Hildén M. 1988. Errors of perception in stock and recruitment studies due to wrong choices of natural mortality rate in Virtual Population Analysis. *J. Cons. int. Explor. Mer*, 44:123-134.

II Hildén M. 1988. Significance of the functional response of predators to changes in prey abundance in multispecies virtual population analysis. *Can. J. Fish. Aquat. Sci.* 45:89-96.

III Hildén M. & Kaitala V. 1991. Comprehensive sensitivity analysis of a bioeconomic stock-recruitment model. *Ecol. modelling* 54:37 - 57.

IV Hildén M. 1993. Short term TAC Advice and multispecies effects. In: Nielsen J. R. & Helgason T. (eds.) *Multispecies fisheries management*, Nordic Council of Ministers, Nord 1993:3, pp. 49-65.

V Hildén M. 1993. Reference points for fisheries management: the ICES experience. *Can. Spec. Publ. Fish. Aquat. Sci.* 120:59-65.

VI Hildén M. 1997. Conflicts between fisheries and seabirds - management options using decision analysis. *Marine Policy* 21:143-153.

VII Hildén M. 1997. Boundary conditions for the sustainable use of major fish stocks in the Baltic Sea. *Ecological Economics* 20:209-220.

Author's contribution

The author of this thesis is fully responsible for all the original publications except (III). Dr. Veijo Kaitala proposed a joint study on the sensitivity analysis of a stock recruitment model. The sensitivity analysis was developed and carried out by the author, and the paper (III) was written jointly.

It is common knowledge that every effect has its cause, and this is a universal truth, however, it is impossible to avoid certain errors of judgement, or of simple identification, for we might think that this effect comes from that cause, when after all it was some other cause, beyond any understanding we possess or knowledge we think we possess.

José Saramago, The Stone Raft (transl. G. Pontiero)

Risk, uncertainty, indeterminacy and ignorance in fisheries management – an analysis of management advice

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Hildén, M. 1997. Risk, uncertainty, indeterminacy and ignorance in fisheries management – an analysis of management advice. Monographs of the Boreal Environment Research No. 5, 1997.

This study combines an ecological perspective on the exploitation of fish resources with an examination of fisheries management as planning. Through a conceptual, theoretical and empirical analysis I show that the combined perspective can provide additional insights into the problems of fisheries management and into the delivery of scientific advice for managerial decisions. A central theme is the lack of perfect knowledge and the consequences and implications for management advice. A more profound understanding of the nature of the imperfect knowledge that haunts fisheries scientists may help both producers and users of management advice to actively strive for the sustainable use of fish resources.

Keywords: Fisheries management, management objectives, fisheries science, management advice, planning, risk, uncertainty.

1 Introduction

All fisheries are based on the exploitation of ecological systems; fishing is therefore an ecological process. Fishers use ecological knowledge, but they also depend on technology to locate and catch fish. As producers of food and other raw material they are economic actors who react to signals from markets and to other incentives. Fishers are also members of societies whose traditions, social conditions and culture regulate and direct their activities. Fishing is thus much more than the act of catching fish. Fisheries are a diverse mixture of ecological, technical, economic, social and cultural variables involving many actors and interest groups.

Resource management, which can include resource enhancement, is a general term for active regulatory interference with the exploitation of natural resources. Managerial decisions concerning natural resources thus influence a wide range of processes and phenomena both directly and indirectly. This multidimensionality is particularly evident in fisheries management. Those who make managerial decisions use many different types of knowledge of the structure and functions of the system to be managed, although the starting point is usually some ecological knowledge concerning the resources. Here and subsequently I use knowledge in a wide sense, to cover understanding, information and data.

In many small fisheries the same individuals possess and accumulate knowledge of the state of the resources, manage the resources and do the fishing. In large fisheries, especially in those that are internationally managed, the tasks are usually separated. In such fisheries we can identify the fishing industry, managers, and the scientific institutions whose task it is to supply the managers and other actors with knowledge of the fisheries. This differentiation into users and producers of knowledge demands communication, which makes the knowledge explicit. The actors can then question the knowledge: is it relevant, is it sufficient and is it valid?

These questions are justified and important, because the knowledge that fisheries managers and other actors in fisheries have to rely on is far from perfect. Those who decide on managerial action do not have exact knowledge of the state of a fishery and its resources, nor can they be fully aware of the consequences of their decisions. I thus use the notion of imperfect knowledge in its widest sense, to cover imperfect data, lack of information and poor understanding of the systems being studied. An overview of management reality illustrates the importance of imperfect knowledge. For example, in 1993 the International Council for the Exploration of the Sea (ICES) could give no, or only crude, indicative quantitative information on half the stocks assessed for management advice (ICES 1994a). In other parts of the world poor knowledge is likely to play an even more dominant role in the managers' daily work.

I focus on knowledge, because its imperfections acquire specific significance in resource management. Arguments concerning the structure, function and dynamics of the system to be managed constitute not only an academic debate over different interpretations of reality, but also a dispute over whose reality and world view will determine managerial decisions. Thus imperfect knowledge enters all the tasks and phases of the elaborate processes for deriving decision-making material and decisions that have been set up to compensate for the lack of direct observation and control of fisheries and fish resources.

Although there are examples of early considerations of imperfect knowledge in fisheries research (Ricker 1958, Walters and Hilborn 1976), systematic interest in risk and uncertainty in fisheries is of recent origin (Walters 1986, Christie and Spangler 1987, Beamish and McFarlane 1989, Smith *et al.* 1993). Analysis of the effects of, and responses to, imperfect knowledge of fisheries have advanced rapidly.

Similar interest has arisen in environmental issues in general (Dovers and Handmer 1995). A practical response to imperfect knowledge has been the development of a precautionary approach (Garcia 1994), with guidance for management to make the approach operational (FAO 1995a). Examples of cautious management are readily available, but the question of how cautious is cautious enough will always be encountered.

Resource management scientists, who have noticed the existence of the imperfect knowledge of resources and its potential consequences for decisions, have arrived at different conclusions as to its significance. What might be called the pessimistic view holds that population models are poorly supported by available data (Hall 1988), or that the sources of uncertainties are so large that overexploitation of resources cannot be prevented (Ludwig *et al.* 1993). Alternatively, biological models have been based on a flawed paradigm that has led to the wrong type of management structure (Wilson *et al.* 1994). A more optimistic view suggests that the systematic analysis of risks will provide managers with the information necessary for achieving sustainable use of natural resources (Rosenberg *et al.* 1993). At a general level there is support for both viewpoints. The spectacular failures of management efforts in some extensively researched fisheries (Hilborn and Walters 1992, Finlayson 1993, Schrank 1995) support the pessimistic view. On the other hand, Frederick and Peterman's (1995) results support the more optimistic view. They showed that in a strictly quantitative sense, there are cases in which management decisions are not affected by lack of biological knowledge. It is also possible to analyse quantitatively when lack of knowledge matters.

The objective of this study is to examine the sources and nature of imperfect knowledge in fisheries management. In other words: I shall examine why knowledge is imperfect and why recommendations that are based on available knowledge do not materialise into action. This leads to a discussion on how imperfections of knowledge should be taken into account in compiling scientific support for management decisions. The perspective is that of the scientists producing material to help those making managerial decisions. Many of these scientists are biologists, but researchers of other disciplines are increasingly involved in producing material for managers and other actors in fisheries management. I refer to all these suppliers of knowledge as fisheries scientists.

The multifaceted nature of the issue of imperfect knowledge and of fisheries themselves calls for examination from different viewpoints. Integration of ecological theory with planning theory and decision analysis defined as "the careful deliberation that precedes a decision" (French 1988) sets the conceptual framework for the study. The extent of the subject is such that not all aspects can be equally covered. The individual studies (I-VII) on which this synthesis is based are examples of different problem areas. Combined with a broad literature review and the additional analyses presented in this synthesis, they outline some crucial features of imperfections of knowledge in fisheries management. My study thus contributes to fisheries management science in the sense of Stephenson and Lane (1995). At the practical level it is closely related to the idea of "ignorance auditing" (Dovers and Handmer 1995). It also provides a new perspective on the optimistic and pessimistic views of the potential for knowledge to support sustainable fisheries management.

After a theoretical and methodological overview (Sections 2 and 3) I shall examine imperfect knowledge of the biological system (Section 4). This analysis is based largely on an examination of models, because models are central tools in providing scientific advice for fisheries management. The models reflect and affect the perception and analysis of key processes, identification of means, assessment of outcomes and monitoring of consequences. In Section 5, I shall discuss management advice and the role of imperfect knowledge in formulating this advice. Management advice relates to boundaries and choice of action in the management process. This is followed by an analysis of how imperfect knowledge relates to the objectives of fisheries management (Section 6). Finally, I shall discuss the implications of the findings for the provision of scientific support for decisions aiming at sustainable use of fishery resources (Section 7).

2 Background

2.1 Theoretical foundations

The concept of imperfect knowledge is broad, and some categorisation is necessary before we can discuss it in detail. There is, however, no generally accepted terminology. The economic literature generally refers to uncertainty or to markets with imperfect information in contrast to the assumption of perfect markets that underlies a significant part of classical microeconomic theory (Kreps 1990). Uncertainty and risk are sometimes interchanged. Decision and risk analysts tend to give "risk" a very precise numerical meaning (Winterfeldt and Edwards 1986), whereas "risk" for some sociologists has become a very broad concept (Szerszynski et al. 1996). For example, Eräsaari (1993) suggests that the transformation of societal problem categories can be described as an evolution from "questions", for which an answer has been promised, to "problems", which can be regulated, and, finally, to "risks", for which there are no answers, not even regulation.

Dovers and Handmer (1995) list three different classifications of "ignorance", which they use as the general concept. Here I use imperfect knowledge as the general concept. Throughout this study I shall use Wynne's (1992) classification of different types of imperfect knowledge. Thus, when necessary, I shall distinguish between risk, uncertainty, ignorance and indeterminacy. The distinctions are that risk can be quantified and uncertainty delimited and reduced, but that ignorance escapes recognition by definition. Indeterminacy, or what Feldman (1989) at a political level calls ambiguity, is an expression for open causal chains or networks (Wynne 1992). The categories may differ in degree of imperfection of knowledge and the distinctions may be fuzzy but they still provide us with a way to conceptualise tendencies that are significant in developing management and management advice.

The other starting point of my study is ecological science and the concept of sustainability. The state of the resource is thus a key criterion in evaluating resource management. The premise of a recent definition of sustainability is that "biogeophysical sustainability is the maintenance and/or improvement of the integrity of the life-support system on Earth" (Munasinghe and Shearer 1995). In the Convention on Biological Diversity (Rio Convention 1992) sustainable use is defined as "the use of components of biological diversity in a way and at a rate that does not lead to the long term decline of biological diversity, thereby maintaining its potential to meet the needs and aspirations of present and future generations." This is the definition I refer to throughout the study.

With its reference to the needs and aspirations of present and future generations, the Rio definition of sustainability shows that socio-economic and political issues are relevant. An analysis of the role of imperfect knowledge must therefore address both ecological and socio-economic aspects of the exploitation of fisheries resources. To achieve this I view management in relation to several dimensions of ecological and planning theory. Such an approach leads to the use of conceptual extremes in the discussion. Real fisheries management obviously combines several different features. The extremes serve to characterise some key tendencies of fisheries management and they allow me to examine the differences in their relation to imperfections of knowledge.

Exploitation of fish resources can be viewed from the point of view of population dynamics, community ecology or population genetics. In its simplest form, exploitation is a factor governing the dynamics of a single population. When species interactions are introduced the exploitation becomes a community ecological phenomenon. Another option is to remove the assumption of a genetically static homogeneous unit stock and view exploitation as a factor influencing population genetics. When the two extensions are combined the exploitation of fish resources becomes an ecosystem issue.

Ranging from international negotiations to local community-based management, the many institutions and institutionalised processes that have evolved for different types of fishery show that fisheries management is a socio-political activity. The management process and its demand for and use of knowledge cannot be understood merely with reference to the ecological systems that it seeks to control. Management is usually a public activity and can be regarded as a form of planning for the wise use of resources. Planning theory can therefore be a fruitful perspective for gaining insight into management as a social and political process. Resource management is, however, a particular type of planning in that ecological processes strongly influence the planning agenda and the opportunities for control. Here I consider planning theory together with ecological theory to obtain a theoretical reference for resource management. The conceptual link between the ecological system and planning is clearly control of the exploitation of resources. Figure 1 illustrates the conceptual connections. Subsection 2.2 deals with the practical connections between planning and the ecosystems and the exploitation of these ecosystems in fisheries.

The theory of planning I refer to in Figure 1 is Sager's (1990, 1994) contingency approach to planning. The key feature of this approach is a conceptualization of planning in three dichotomies (Sager 1990): synoptic planning versus incrementalism, which reflects the degree of control over the system to be planned; technique versus communication, which is related to the underlying rationality of the plan; and flexibility versus rigidity, through which problems of efficiency and risk become meaningful. I specificly address questions related to control and rationality. Risk is discussed as one aspect of imperfect knowledge. Efficiency is not dealt with and flexibility is only briefly touched on in the concluding section.

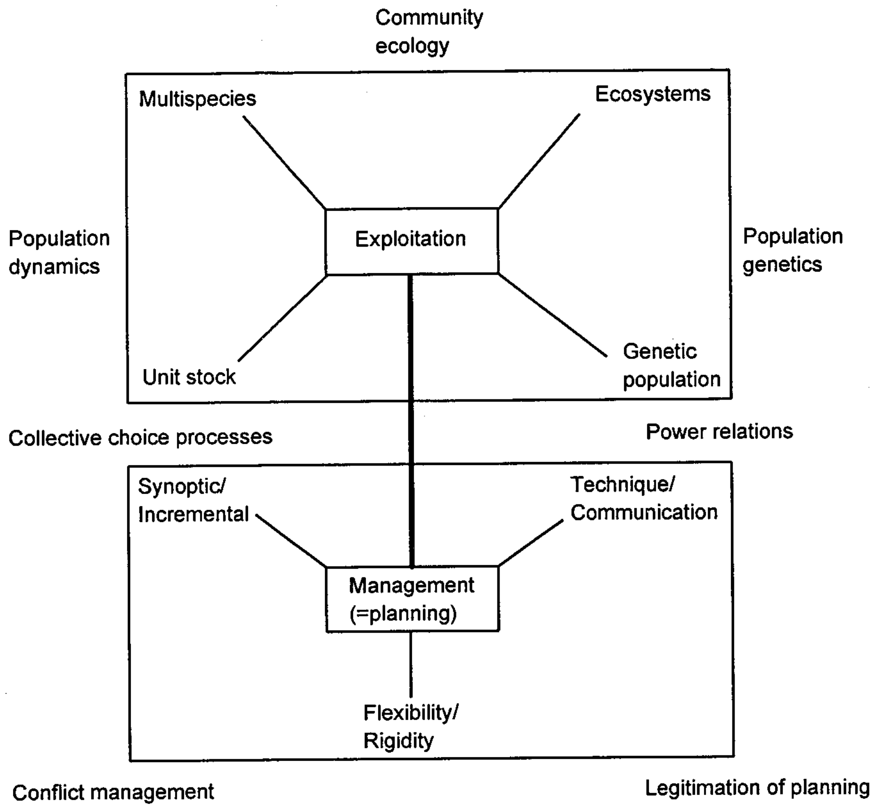

Fig. 1. The ecological concepts related to the exploitation of fish resources, and planning theoretical concepts related to the management of exploitation. Relevant ecological disciplines are noted outside the exploitation frame. Different aspects of the planning environment are indicated outside the management frame. The lower part of the figure is slightly modified from Sager (1990).

2.2 The management process

Before we can examine the sources and role of imperfect information we must place the analysis in the context of the management process. As noted above, management can be regarded as a form of planning that includes collecting and processing information on the future, delivering technical input to decision-making processes, arranging for communication among the parties involved in the decision-making, preparing for collective action, finding out by rational means how to achieve goals, making strategic proposals and managing conflicts over which future actions to choose (Sager 1990). When planning aims at rational action and the processing of knowledge for rational action we can identify some characteristics of "optimal" planning. Elster (1989) notes that an optimal amount of evidence should be available for a rational action, but the optimality depends on the desires the action should fulfill and on beliefs about the costs and benefits of gathering more information. This raises the question, what do we consider rational?

Being firmly rooted in biological sciences, "instrumental reason" in the sense of Habermas (1971) not surprisingly dominates fisheries management. Instrumental rationality provides guidance on the use of means to achieve stated objectives without attaching preferences to the means. The concepts of maximum or optimum sustainable yields demonstrate the existence of goal-oriented instrumental rationality in fisheries management. International agreements and fisheries legislation, e.g. the Finnish Act on Fisheries (286/82, Section 1), have codified instrumental rationality. But if this were the only rationality, many actions would have to be defined as irrational. Sager (1990) has shown that planning can be more deeply understood by introducing concepts for non-instrumental reason such as Habermas's (1984) "communicative rationality", which emphasises processes rather than the goal. In fisheries management this can simply mean that the management is motivated not so much by the nebulous concept of maximum sustainable yield as by the social activity and communication leading to management decisions. I shall show that the underlying rationality will affect the perception of imperfect knowledge and the responses considered appropriate even in assembling biological knowledge for fisheries management.

Table 1. Basic tasks of management and their interpretation in a fisheries context.

General management task	Examples of interpretations in fisheries management
1) Set the system boundaries	- Determine the geographical area of the resource(s) - Determine which interactions are considered relevant - Determine who are the stakeholders - Determine the "relevant considerations"
2) Determine the objectives and make them operational	General objectives: - Yield maximisation - Maximisation of economic rent - Supply of recreational opportunities - Conservation of resources Specific targets: - Level for fishing mortality - Short and long term yields - Escapement levels - Number of fishers
3) Analyse and understand key processes of the system	- Analysis and prediction of recruitment and stock size - Determination of growth and mortality patterns - Identification of interactions - Estimation of fishing effort and factors affecting it
4) Determine the means available for achieving the objectives in relation to understanding of the processes and the chosen objectives	- Regulation of catches (quotas) = output control - Regulation of fishing effort (temporal, spatial, technological) = input control - Regulation of entry and exit (licensing, a form of input control) - Resource enhancement
5) Assess possible actions and outcomes in relation to 3) and 4)	- Which quota will bring the fishing mortality down to the target level? - Can resource enhancement produce expected yields or economic returns?
6) Decide on action conditional on 1), 2), and 5)	- Decisions by local, regional, national or international management body
7) monitor the consequences	- Scientific sampling of catches, collection of fisheries statistics - Population monitoring - Modelling of population response - Analysis of response in fishery
8) repeat steps 1 to 7	

The actual management can be carried out by very different groups such as authorities, communities, private owners of waters or individuals with fishing rights, but the process typically involves the same basic tasks (Table 1). These tasks are similar in any planning or management problem (Rosenhead 1989) and they all involve making decisions using available knowledge. The making of decisions should here be understood in a wide sense. It is a question not only of setting quotas or gear restrictions but also of determining what should be investigated and what objectives should steer the process. The concept of rational action underlies the whole structure and also each task individually. When strictly instrumental considerations dominate, the tasks are considered to form a linear sequence. In the ideal case, perfect knowledge leads to omnipotent or synoptic planning (Sager 1990). Sager (1990) suggests that the appropriate contrast to synoptic planning is "social incrementalistic planning", which is rooted in Lindblom's (1959) concept of "muddling through" but expanded to include unlimited communicative rationality. In incrementalistic planning, knowledge of the past, present and future is imperfect, and the exact nature of expectations concerning future states of the world is unknown (Schiller 1987).

Fisheries management is influenced by the biological system. The cyclic nature of the underlying biological system often emphasizes repeated measures and regular reassessments. There are always some new recruits to a fish stock and the fish are continually growing. In a limited sense stocks and fisheries have been sustainable up to the present, a factor that influences the type of adjustments managers are willing to consider. At the same time flexibility is important. Despite the cyclicity of the biological system there is no guarantee that the next year will be like the present. Recruitment may differ by an order of magnitude and migration may lead to an apparent decline in a stock.

As in all decision problems, feed-backs and iterations between tasks are possible (Raiffa 1968) (Fig. 2). For example, new understanding of key

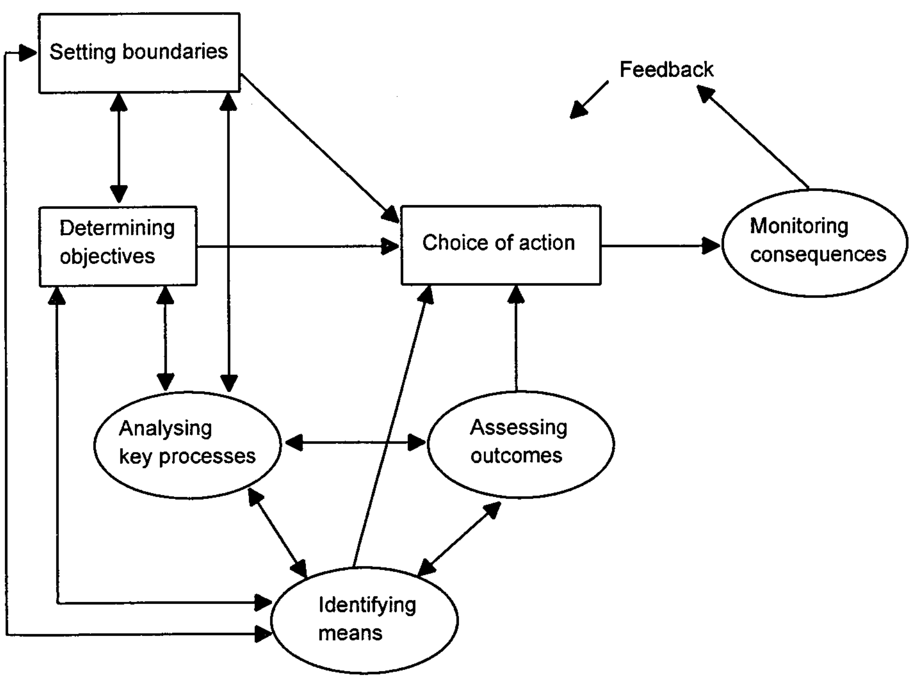

Fig. 2. The relationship between management tasks. Tasks dominated by decision making are given as squares whereas tasks dominated by scientific analysis are given in ovals. The feed back can affect all the tasks.

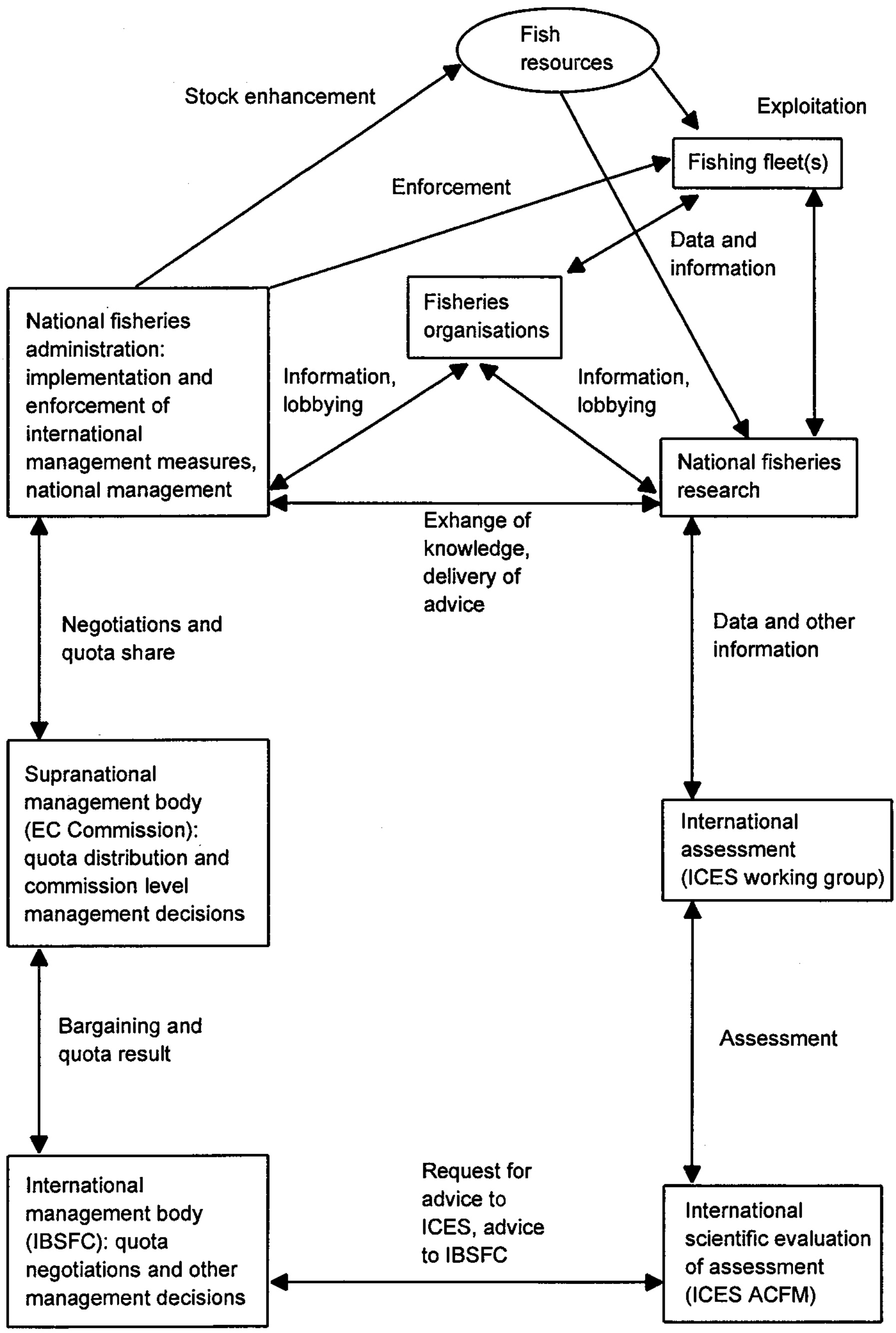

Fig. 3. A generalisation of the management structure for international stocks in the Baltic Sea. Similar structures occur throughout the ICES area. The many negotiation levels emphasise the political nature of knowledge.

processes may lead to a revision of boundaries and objectives, and the results of monitoring to a revision of any other task. In fisheries management, biological considerations have dominated tasks 1), 3) and 5), but feed-backs and iterations cause biological considerations to be taken into account in all tasks. This is also indicated in Figure 2. I shall explore imperfections in our knowledge of the biological system and its connection with other items of knowledge and the managerial decisions in greater detail in the following sections. At this stage it is sufficient to note that two roles dominate. First, imperfections of knowledge play a role in the instrumental concern for sustainability of the fishery resource: Sustainability is a specified objective, but how do we know whether we are approaching or moving away from sustainable conditions? Second, imperfections of knowledge have a role in the planning environment, with its collective choice processes, questions of legitimation, power relations and conflict management. The planning environment involves many different actors, because the management systems, especially of internationally managed fisheries, include many partly or wholly separate functions (Fig. 3). In such an environment knowledge easily becomes political.

3 Material and methods

3.1 Terminology and abbreviations

Throughout the study the following abbreviations have been used for standard concept in fisheries research.

F = Instantaneous rate of fishing mortality ($time^{-1}$)

$F_{0.1}$ = The instantaneous rate of fishing mortality at a point where the slope of the curve relating yield per recruit to the fishing mortality rate is 1/10 of the slope at the origin.

F_{max} = The instantaneous rate of fishing mortality at a point where the curve relating yield per recruit to the fishing mortality rate reaches its maximum.

M = Instantaneous rate of natural mortality ($time^{-1}$)

MSVPA = Multispecies Virtual Population Analysis

R = Recruitment, usually defined as recruitment to the exploited part of the population at a particular age.

SSB = Spawning Stock Biomass, i.e. biomass of the mature part of the population.

TAC = Total Allowable Catch, i.e. maximum quota set for a fixed period of time, usually a year.

Y/R = Yield per Recruit, i.e. the expected yield from a fish recruiting to the exploited part of the population given its growth and mortality pattern as a function of age

Z = $F + M$

3.2 Model sensitivities

I have analysed the sensitivity of three simple fish population models to imperfect knowledge (I,II,III). The general form for the models is derived from ecological principles governing population change. The differential equation $dN/dt=g(N)$ or its difference equivalent $N(t+1) = u(N(t))$, where N stands for population or year class and g and u for functions, are the basic equations of most assessment models (Beverton and Holt 1957, Seber 1982, Ricker 1975, Gulland 1983, Hilborn and Walters 1992). Their origins can be traced to the 1920s (Ricker 1977). I have further analysed models for the fate of individual year classes without (I) or with (II,IV) species interactions, and a classical model of the relationship between the spawners and subsequent recruitment (III).

The virtual population analysis or cohort analysis examined in (I) and (II) underlies nearly all the scientific management advice based on "analytical assessments" provided by, among others, the International Council for the Exploration of the Sea (ICES). The same equations are also found in the core of advice on the western Atlantic and several Pacific fish stocks. The stock recruitment model of (III) is extensively used in Pacific salmon management (Hilborn and Walters 1992). In fisheries management other models such as those based on the Leslie matrix (Seber 1982), biomass models (Laevastu and Larkins 1981) and delay-difference equations have also been used. For management the fundamental problems are, however, the same:

do the models provide a plausible description of the dynamics of the resource and its exploitation, can reasonable parameter values be determined and do the models provide a tool for predictions that can be used in management? These problems are of practical and theoretical importance in the management process.

Sensitivity analysis can be performed using many different techniques (Swartzman and Kaluzny 1987), but the objective of providing a deeper understanding of the models and their results is common to all. A sensitivity analysis cannot falsify or provide support for a model because "it is impossible to tell from the picture alone whether it is true or false" (Wittgenstein 1921, 2.224). It can, however, deepen our understanding of the picture and thus point towards avenues for research and examination of particular features of reality.

In (I) I combined analytical derivation of the effects of bias in the natural mortality rate in the classical virtual population analysis (or cohort analysis) with simulations and examined the feasibility of removing distortions that the choice of natural mortality rate can create. A feature of the virtual population analysis used in the argumentation is its tendency to be conservative with respect to the total mortality rate. Furthermore a "tuning" procedure, i.e. adjustment of the fishing mortality rates for the oldest age group using historical data on fishing effort and mortality rates, will tend to compensate for overestimation of the natural mortality rate with underestimation of the fishing mortality rate, and vice versa (ICES 1993).

In (II) and (III) I used the Fourier Amplitude Sensitivity Test (FAST) developed by Cukier *et al.* (1978) for the sensitivity analysis. The FAST technique permits effective and systematic examination of different combinations of deviations of parameter values from the nominal values. It has the advantage that the effects of a particular parameter can easily be identified in the output by examining the frequency spectrum of the output variability.

In the applications no correlations were assumed between the parameters. The notion of a "joint probability distribution" in (III, p. 41) is thus somewhat misleading, because the parameter values are picked from the individual uncorrelated distributions, ensuring, however, that all possible combinations are effectively covered. I did not consider the lack of correlation between the parameters to be a major drawback in the examination of the models I have focused on. In a model of several species with correlated growth patterns, however, the independence assumption would clearly be unrealistic. Another theoretical criticism of the method concerns the use of truncated uniform distributions, as these may neglect the effects of extreme but infrequent parameter values. In these studies I considered that the effective mapping of all parameter combinations gives a sufficient picture of the model behaviour. I solved the problem of truncation by using ranges of parameter values wide enough to cover most values that could be realistically assumed or estimated.

A practical disadvantage of the FAST technique is that it requires a rapidly increasing number of simulations to keep pace with the growth in number of parameters. I therefore had to simplify the multispecies population analysis to the examination of a single cohort (II). In so doing I followed the suggestion of Holling (1978) that the essence of a complex model should be investigated in a reduced model. Another option would have been to set up a fractional factorial design for the model simulations (Cochran 1977, Swartzman and Kaluzny 1987). Such a design relies on a statistical analysis between parameter input values and model results to identify the most important parameters, and avoids the problem of too many simulations. This technique has been used by the multispecies assessment working group of ICES (1994b). They did not, however, systematically examine the effects of different functional responses until 1993.

In (III) I introduce an economic dimension in the form of the discount rate and the catch value. Although incomplete, this dimension permits a crude analysis of the possible consequences of economic factors on decisions concerning resource management. In the same context I examined some of the features that may affect entry and exit decisions by looking not only at the variability and the mean, but also at the frequency of low yield (income) situations.

3.3 Factors affecting management advice

Imperfections of knowledge, and especially what in everyday usage are called risks, are partly social constructs (Merkhofer 1987, Freudenburg 1988). The perception of the quality of knowledge influences the decisions taken by all actors in a fishery, managers included. Sometimes imperfections of knowledge are seen to centre on biological issues and models, sometimes on social, technological or economic issues. Whatever the case it is clear that management processes and decisions cannot be understood through quantitative analysis of model behaviour alone.

In (IV), actual management advice is contrasted with available information. ICES, which produces the advice, has used class decision rules for deriving management advice from available evidence. Attempts have been made to formalise the decision rules for producing fisheries management advice in relation to specific reference points. The earliest approaches were based on the concept of maximum sustainable yield, but these have been dismissed in favour of more fuzzy biological reference points (V). This suggests a deliberate loosening of the connection between the biological information derived from population models and the management advice.

ICES produces scientific management advice in a partly subjective and partly objective process in which the final result is an outcome of negotiations between nationally appointed experts using model estimations and other available knowledge as basic material (Fig. 3). To search for possible signals of key information in the final advice I examined the relationship between model output and advice using historical data series. In the exploratory phase I also used simple regression models. No claim of causal explanation is, however, made for the models, and in the original contribution (IV) parameter values were not reported for the models. Here I give a fuller account of the statistical analysis of the data. Due to the nature of the process, however, no statistical analysis can detect all relevant aspects. Full analysis of how management advice evolves in the process would demand a sociological analysis of all key actors. Such an analysis, showing the considerable importance of personal relations, subjective judgements and idiosyncracies, has been presented for the case of northern cod in Canada (Finlayson 1993).

3.4 The effects of management advice

If the knowledge provided by scientists were accepted, and managers and politicians were to follow biological advice to the letter the state of fish resources should improve. This is in essence the justification for the elaborate systems of assembling knowledge that are part of fisheries management throughout the world. The hypothesis that scientific knowledge is the key to management can be tested using empirical data on management advice and the development of the fish stock status (V). I did this using a simple overview of all stocks for which advice has been given at two moments in time. The time interval I chose was long enough to allow changes in stock status to take place, but short enough to prevent methodological development or management structure from having too great an impact. This method could yield erroneous conclusions because factors other than the fishery might lead scientists to recommend stricter management measures for the protection of the stocks. I therefore supplemented the general analysis with a detailed examination of two stocks that have been of major concern to ICES scientists, namely, North Sea haddock and cod (V). I chose these stocks because their development has significantly affected the nature of the scientific advice given by the ICES. For example, in 1990, the ICES ceased giving management advice in the form of a specific total allowable catch (TAC) for these stocks (ICES 1991a). I further supplemented the analysis with data on advice concerning other stocks compiled from the reports of the Advisory Committee on Fishery Management (ACFM) 1979-1995 (Appendix 1).

To gain deeper insight into the use of advice I have examined the statistical associations between relevant variables. One of these is recruitment variability. One might argue that advice on stocks whose abundance varies greatly would be difficult to follow. As a consequence, the management advice would also have to display greater variability simply because the need to advice a reduction in fishing mortality rates might arise more frequently than for stable stocks. As an indicator of recruitment variability I have used Myers et al.'s (1995a) results based on log transformed recruitment data. In addition to the ecological variable I have examined the total value of the catch and the price per weight. Data on prices are available at a crude level from the FAO fishery statistics year book (FAO 1995b). I obtained the average value of the catch, which indicates the value of the resource, by multiplying the average catch as reported by ICES (1996) by the unit price.

I analysed the management advice, recruitment variability and price information using log-linear models (Bishop et al. 1975), because the recommendations can then be examined as discrete classes. In the analysis I used three categories of recommendations: those to keep or accept the present rate of fishing mortality rate; those to reduce the fishing mortality rate; and strong recommendations to reduce the fishing mortality rate. The distinction between a recommendation to reduce the fishing mortality rate and a strong recommendation to do so is sometimes subtle. I have interpreted the recommendation as strong when the ACFM has recommended an reduction of at least 20 per cent.

The price, catch and recruitment variables are continuous, but the quality and availability of the data differ. To ensure a minimum level of quality I used only stocks for which a so called analytical assessment had been made and recommendations given for 1990 to 1995 (Appendix 2 and 3). The price data are areal averages. The mean catches and recruitment variability have been calculated for available time series by ICES (1995) and Myers et al. (1995a) respectively. The recommendations are reviewed yearly. For the analysis I discretised catch value and recruitment variability into four classes using quartiles of the data. For the price of fish I identified three classes (< 0.5, $0.5 \leq\ < 2$, ≥ 2 US dollars per kg). I analysed the data using the CATMOD and FREQ procedures of the SAS package (SAS Institute 1989).

3.5 Sources of conflict in management and management advice

The introduction of negotiations in the production of management advice as well as in management decisions (Fig. 3) indicates implicit recognition of the possibility of different interpretations of biological knowledge. I addressed this question by examining a specific conflict (VI). In this case I obtained crude quantitative estimates of the conflict of interest using the basic population and catch equations of (I), but in addition I analysed written material to find evidence of other conflict dimensions (VI). Previous analyses (Nettleship *et al.* 1984, ICES 1994c) have focused only on the quantitative aspect of the competition between seabirds and fisheries.

3.6 Assembling knowledge for decisions supporting sustainability

Sustainable use is specified as the general objective of resource management. This objective has been set by numerous international bodies, conferences and conventions, including the UN Convention on the Law of the Sea (United Nations 1983) and UNCED's Agenda 21 (United Nations 1992). To be meaningful the concept of sustainability has to be operationalised in specific sectors. In fisheries management sustainability becomes an issue in such matters as the rules on fishing methods and the size of catches; the protection and enhancement of fish resources; fishery development; and the mediation of conflict between actors and interest groups.

To examine the potential of management advice to promote sustainability I chose the major fish resources of the Baltic Sea as a case (VII). By systematically investigating the type of management advice given, the factors that appear to influence the fisheries of the major fish stocks in the Baltic Sea and available information on fisheries, I was able to suggest indicators for sustainable resource use referring specificly to fisheries. Methodologically the study (VII) is closely connected to the development of environmental indicators (OECD 1994, Hammond *et al.* 1995), with the difference that, in the case of the Baltic Sea, we can identify in advance the decisions for which the information should be usable.

4 Imperfect knowledge of the ecological system

In the following I use the observations in the individual studies (I-VII) together with other related published sources to examine the nature of imperfect knowledge of the ecological system. This section explores the imperfections of knowledge faced by scientists attempting to analyse a fishery. Since my starting point is the basic issues affecting the analysis of any biological system, I first examine the imperfections of knowledge that are related to the geographical boundaries, sampling and modelling. I go on to examine imperfections in the knowledge of key factors in the dynamics of an isolated single, or unit, stock. The analysis is then expanded by examining the effects of relaxing some central restrictive assumptions of the unit stock concept.

4.1 Basic issues

4.1.1 Geographical boundaries

Geographical boundaries are necessary for management. Ideally, boundaries delimit a unit such that interaction with other units is minimal. Management measures can then be focused, and so interfere minimally with neighbouring units. In reality only almost closed systems such as small lakes approach this ideal. In larger systems boundaries are always to a lesser or greater extent arbitrary. They are compromises because relevant processes operate on different spatial scales.

A single fishing fleet may exploit several separate fish populations at different rates, but it is not always possible to manage the stocks separately. Instead the area in which a fleet operates can be used to define boundaries. On the other hand, many different fleets may exploit a common resource, *e.g.* highly migratory or straddling stocks. The distribution of the stock would then seem to be the best way to define the boundaries. In reality, both cases often occur within the same geographical area. For example, in the Baltic Sea, the salmon population of the river Tornionjoki is exploited by many fisheries. At the same time Finland's coastal fisheries exploit several populations of salmon simultaneously, thus contributing to overexploitation (ICES 1994a). Similar problems can be found throughout the world, making management units matters of agreement and convenience as much as ecological entities.

The choices that have to be made concerning geographical boundaries introduce the subject of imperfections in knowledge of the resource. It may sometimes be possible to estimate the risks posed by the choice of particular geographical boundaries. In salmon management, considerable effort has been spent on stock identification, and probabilities can be determined for the distribution of catches of different populations (ICES 1994a). In other cases, boundaries (or lack of them!) may lead to uncertainty and ignorance. For example, the demise of vendace, *Coregonus albula* (L.) in the Finnish part of the Bothnian Bay, the Baltic Sea, defied explanation until tagging results revealed an influx of vendace from the Swedish side (Lehtonen and Enderlein 1984). The intensification of the Swedish vendace fishery reduced this influx (Hildén *et al.* 1984). Failure to observe the boundary between inshore and offshore cod fishery together with different developments in the two areas has been postulated as a reason for not recognising signs of decline in the cod stock off Newfoundland (Hutchings and Myers 1994).

4.1.2 Sampling

The theory of sampling ecological phenomena and other relevant fisheries variables is well developed. Standard texts on sampling (Cochran 1977) give the general structure that is applied in the sampling of variables relevant to an analysis fish population dynamics (Doubleday and Rivard 1983). The distinction between precision and accuracy (Cochran 1977) is important. The precision obtained in sampling can be quantified with standard techniques. In well designed stock surveys coefficients of variation as low as 10 per cent can be obtained for some variables, *e.g.* the proportion of fish of a particular age in the total catch (Schweigert and Sibert 1983, Hildén *et al.* 1984, Kimura 1989), although coefficients of variation of 20 per cent or higher are more common (Doubleday and Rivard 1981). Recreational catches can be estimated with coefficients of variation down to 10 per cent (Leinonen 1988), although in some situations they may exceed 100 per cent (Meredith and Malvestuto 1991). A danger of bias is inherent in all sampling and thus coefficients of variation do not necessarily reflect true imperfections of knowledge. Leinonen (1988) observed that bias was typically large in precise catch estimates but less so in imprecise estimates, showing that low coefficients of variation may create a false perception of certainty. The danger of bias emphasises the role of the sampling design in reducing the imperfections of knowledge connected with the sampling process. Unbiased sampling can be dealt with as a quantifiable risk. Bias can sometimes be delimited and then treated as a risk, but in most cases it will be a source of uncertainty or ignorance in management advice.

To reflect the lack of knowledge that sampling causes I used ranges of values for input variables and parameters in the fisheries models (I, II, III) and in the probabilistic statements for the state of fish stocks and their productivity (VI). In practice it is, however, difficult to separate the lack of knowledge of parameter values caused by sampling from that caused by, say, choice of geographical boundaries. Migration patterns and mortality rates are easily confused.

4.1.3 Models for assessment and management

Models of biological phenomena are the intellectual tools underlying all resource management, even when they are only expressed at a heuristic level in the form of assumed dependencies between variables. A model represents a particular choice of imperfect knowledge.

The use of models in resource management differs conceptually from the use of models in basic ecological research. In ecological research, understanding and prediction are major modelling objectives (May 1981). Resource management is based on the idea of active intervention. Prescriptive assessments of present and future states dominate management modelling. The models include control variables that management can manipulate. A key task of modelling is considered to be a reduction of uncertainty (Varis *et al.* 1994). This is true in the sense that loose arguments are given a clear structure. Risks associated with the use of a particular model can be assessed in relation to historical data. But since a management model also imposes its structure on decisions governing future states of the system it can become a source of ignorance. This is especially true of the large number of models (Allen 1953, Beverton and Holt 1957, Palm 1975, Clark 1976; 1985) devoted to the optimisation of resource exploitation.

Much has been written about the identification and choice of appropriate model structures (Vemuri 1978, Halfon 1983a,b, 1985, Bartell *et al.* 1988, Kettunen 1993), but models cannot be definitely verified (Swartzman and Kaluzny 1987). In resource management an adaptive approach aiming to produce data that can help to refute some model structures and corroborate others may provide opportunities for reducing uncertainties without increasing ignorance (Holling 1978, Walters 1986, Walters and Collie 1989, Sainsbury 1991).

The inherent imperfections of knowledge concerning both models and available data cast doubt on the use of deterministic mechanistic models. Explicit recognition of a lack of knowledge led me to the use of crude probabilistic modelling (VI). However, a true reduction of the risks and uncertainties involving the biological components of a fishery requires that the two modelling approaches should complement one another as in (VI).

4.2 Population dynamics of a single stock

The perception of the dynamics of exploited populations is critical for management. Mortality and recruitment, which determine the size and productivity of exploitable resources, are key ecological processes. Imperfections of our understanding of these processes are due to the sampling of biological phenomena but they are also caused by the choice and fitting of models.

4.2.1 The rate of natural mortality

Together with the fishing mortality rate, the natural mortality rate is a central parameter determining the dynamics of age-structured fish population models. Historically the natural mortality rate has been chosen using auxiliary information on species longevity, linear regression of total mortality rate against fishing effort (Gulland 1983), tagging data (Ricker 1975, Seber 1982) or the known number of stocked fish and catches (Salojärvi 1991). In a few cases knowledge has been available from unfished or lightly exploited populations (Ricker 1975). Since the natural mortality rate is an aggregate of a wide range of ecological phenomena from predation to disease and life history strategies the possible variation around any point estimate is clearly considerable. For example, Salojärvi (1992) reported variations in instantaneous rates of between 0.07 and 0.37 from his studies of whitefish, and Lapointe *et al.* (1989) consider variations of ± 50 per cent of the point estimate.

The development of multispecies models has allowed estimates to be made of natural mortality rates (Sissenwine and Daan 1991). Calculated coefficients of variation for model outputs of predation mortality rates varied from 0.15 to 0.36 for prey species such as herring and sandeel (I). Estimation of the natural mortality rate has reduced some ignorance but opened up several new sources of imperfect knowledge. Poor knowledge of the predation process is one example (II). Another is inadequacies in the estimation of the amount of food consumed by predators, which depending on the submodel can result in major differences in predator effects (ICES 1991b, Hansson *et al.* 1995). These issues represent uncertainties, however, not ignorance, because they can be addressed through experimental work and through analysis of monitoring data.

4.2.2 Recruitment

Recruitment to existing fish stocks is a strong driving variable in determining the dynamics of exploited fish populations. Recruitment levels can be determined through catch rates of surveys or through direct enumeration using a method such as hydroacoustics, or they can be estimated *post factum* using population models. Ever since Hjort (1914) recruitment variability has been of interest to fisheries scientists. Recruitment data show considerable variation (Cushing 1977, Rotschild 1986, Myers *et al.* 1995a). The interpretation of the processes determining variability has a direct influence on the evaluation of the consequences of different management options.

It is commonly assumed that the variability observed in recruitment levels is largely exogenously driven by climatic factors (Cushing 1982). Alternatively one can assume that the variability arises endogenously from deterministic chaos (Wilson *et al.* 1991, Wilson *et al.* 1994). So far there is scant evidence for chaotic behaviour in exploited fish populations (Fogarty 1995). If recruitment variability is modelled and interpreted as a consequence of random environmental effects the evaluation of long-term strategic decisions should be based simply on expected values, possibly corrected for risk attitudes, or equilibrium analysis. If recruitment variability is modelled as deterministic chaotic behaviour (Wilson *et al.* 1991, Wilson *et al.* 1994) one might conclude that active management intervention might, at least in theory, remove some of the variability.

The different interpretations of recruitment variability also affect the perceived role of the spawning stock biomass. Some management agencies have explicitly or implicitly modelled the variation in recruitment omitting the effects of stock size, thus assuming that the size of the spawning stock can be ignored in practical management (Hilborn and Walters 1992). An extensive analysis of relationships between spawning stock biomass and recruitment suggests, however, that depletion of spawning stocks can result in a decline in recruitment levels (Myers *et al.* 1995b). Thus the dynamics of the spawning stock biomass should be included in the identification of key processes (Myers *et al.* 1994).

My studies have shown that lack of knowledge of stock and recruitment relationships can arise from the assumptions made in the analysis of available data on fish stocks. The choice of natural mortality rate (I) and the assumed type of predatory interactions in models with species interactions (II) can distort recruitment variability and so affect interpretations of the stock recruitment data. Later studies (Bradford and Peterman 1989, Lapointe *et al.* 1989) have confirmed the significance of the choice of natural mortality rates. Multispecies assessments have examined evidence for prey shifts and functional responses, but evidence falsifying the use of the present model in the North Sea multispecies analysis has not been found (ICES 1994b).

Several different models have been developed for stock and recruitment data. Two or three parameters have been used to describe a wide range of deterministic relationships between the spawning stock and subsequent recruitment (Ricker 1954, Beverton and Holt 1957, Shepherd 1982). Probabilistic models have used discrete stock size and recruitment intervals (Getz and Swartzman 1981, Evans and Rice 1988). In the predictive recruitment modelling in my studies, lack of knowledge was reflected in wide ranges of parameter values (III) or in probabilistic formulations (VI). For well studied fish stocks lack of knowledge of recruitment patterns (Table 2) can often be characterised as risk, *i.e.* it is possible to state something about the probability distributions of recruitment (VI, VII). For less well studied stocks recruitment is a source of uncertainty.

4.3 From unit stock to communities

The unit stock has been a key concept in fisheries science (Gulland 1983). It has imposed a uniformity constraint on the fish to be managed and has restricted their interactions with the outside world to single parameters. The concept has simplified quantitative estimates and even the considerations of management objectives and measures. An ana-

Table 2. Levels of recruitment variability

Context	Measures of variability	Source
Parameter uncertainty in stock-recruitment model	Range of variability, ratio of high to low recruitment value: from 1.5 at low stock levels to >2.5 at high stock levels	III
Variability of recruitment of sandeel in Shetland n=16 (background for VI)	Coefficient of variation (%) = 55	Myers et al. 1995a
Empirical material (n=274)	Coefficient of variation (%), range: 14 - 273	Myers et al. 1995a

Table 3. Dichotomies of perceptions of the ecological system, their appearance in different management problems and examples of methods and assumptions. In each cell the dominant types of imperfect knowledge are identified.

Stock characteristics	Single species	Community
Unchanging homogeneous unit stocks	**Problem:** Optimisation of exploitation, quota management **Methods:** Analysis of yield per recruit, population analysis, stock-recruitment, production models **Assumptions:** - Fixed M - Constant growth - Recruitment variability "external" => Risk, ignorance	**Problem:** Management of interacting fisheries, trade-offs, gear and entry restrictions **Methods:** Multispecies population analysis, lumped parameter production models **Assumptions:** - Part of M estimable - Constant growth - Recruitment variability mostly external => Risk, uncertainty
Diverse, potentially evolving population	**Problem:** Artificial enhancement, preservation of genetic diversity **Methods:** Population genetic and evolutionary models **Assumptions:** - Differences between individuals - Selection patterns cause changes in population parameters => Risk, uncertainty	**Problem:** Ecosystem management, biological diversity management **Methods:** Transformation models for habitats/biotopes **Assumptions:** Critical habitats/biotopes and driving variables identifiable => Risk, uncertainty, indeterminacy

lysis of the models for unit stocks provides insights into the knowledge used for management in its simplest form. But the unit stock is clearly not the only possible way to examine the ecological basis of management.

The entity for which knowledge is to be compiled for management can be viewed in the dichotomies of single stock vs. community and unchanging unit stock vs. evolving genetic population(s) (Table 3). In the table the upper left hand corner represents "traditional" management, which until recently was a driving force behind the development of management approaches and methods.

The ecological foundations of the original unit stock concept are weak. There are several ways in which the restricting assumptions can be relaxed. The population genetic argument against the concept leads to a recognition that all fish are not equal (Allendorf *et al.* 1987). In their simplest form, the population genetic models deal with a single-locus two-allele system (Kaitala and Getz 1995). Detailed examination of these modelling approaches is beyond the scope of this study, but I note that they can provide a new perspective on exploitation patterns and levels. Moderately high fishing mortality rates may turn out to be unsustainable because of the selection pressure they exert on a nonhomogeneous population (Nelson and Soule 1987, Kaitala and Getz 1995).

The community ecological arguments against the isolated unit stock have led to the development of methods for multispecies assessments (Sissenwine and Daan 1991). Fisheries research has tended to focus on unit stocks and so has not systematically collected information on changes in fish communities. Such changes may then appear erratic and incomprehensible. For example, stock replacement, *i.e.* a shift in dominance in the fish community, cannot be fully analysed with data from research on unit stocks (Daan 1980). Extensive research on unit stocks spanning decades turns out to be insufficient even for understanding changes in the abundance of single stocks (Daan *et al.* 1994, Serchuk *et al.* 1994). Yet the consequences of major shifts in abundance are significant from a management point of view.

An ecosystem approach to management regards the ecological system to be managed as consisting of several diverse and interacting populations. Competition, predation and evolutionary change are all possible in such a system (Ryder *et al.* 1980). The number of questions increases and therefore they have to be answered with a broad brush. Attempts to construct whole ecosystem models for management purposes have met with scepticism, because the number of implicit assumptions is large relative to the amount of data available. At a conceptual level they may, however, open up new avenues for discussion. In fisheries management, the North Sea model of Andersen and Ursin (1977) was seminal for the work on more tractable multispecies virtual population analysis methods.

An interesting question concerns the manner in which imperfections of knowledge change when the perspective shifts from the unit stock to a genetically nonhomogeneous stock, multiple interacting stocks or whole ecosystems. At a general level the imperfections that dominate the knowledge of single unit stocks can be characterised as risk and ignorance. The risks arise from sampling and other data sources and their effects can be calculated. Ignorance arises from the deliberate exclusion of factors affecting stock dynamics. When the management broadens to non-homogeneous stocks or to multiple stocks, some of the self-imposed ignorance is replaced by uncertainty but the sources of risk remain essentially the same. Full evolutionary ecosystem management also transfers some ignorance to uncertainty, but it also expands indeterminacy. The possible pathways for change are numerous and many of the possible cause and effect relationships, *e.g.* the top-down versus bottom-up control of ecosystem dynamics in the Baltic Sea (Rudstam *et al.* 1994), are indeterminate.

5 Imperfect knowledge of fisheries and the management advice

In many parts of the world scientists deliver management advice through institutions specificly founded for this purpose. These institutions filter knowledge of fish stocks and fisheries and provide interpretations in the form of management recommendations. The focus is generally on the biological characteristics and population dynamics of the fish stocks. This section investigates imperfections of knowledge in the delivery of advice to managers. First I address the conceptual boundaries that affect the framing of management advice, second the relation of different planning approaches to imperfect knowledge in the advice, and third, the relation, if any, between management advice and fish stock status.

5.1 Conceptual boundaries in management advice

In efforts to manage natural resources, the focus of management advice is determined by the boundaries between the managed system and the outside world. Such boundaries are not only the geographical boundaries of the stock and their exploitation as noted in subsection 4.1, but also the boundaries between disciplines and issues. In a wider sense, all decisions on system boundaries specify the extent to which processes are regarded as parts of the system for which knowledge is needed and management measures devised. All knowledge will be interpreted in relation to these boundaries. The choice of boundaries thus introduces and modifies imperfections of knowledge.

5.1.1 Management advice beyond unit stocks

Management advice for a unit stock exploited by a sole "owner" (the manager) is conceptually simple. For a sole owner whose only interest lies in the expected yield in weight a maximum sustainable yield is a valid objective. Scientists can focus on measures such as recruitment, spawning stock biomass, growth rate, fishing mortality rate and its distribution over age groups to compile management advice. The average yield can be used as a measure of success as the variability does not matter. Given the basic biological knowledge of the stocks as perceived by the scientists, the management advice should then be reasonably predictable. In (IV) I show that this is indeed the case.

When the unit stock assumption is relaxed by introducing species interactions, the conditions for producing the management advice change along with the perception of stock dynamics. Any reduction in ignorance achieved by broadening the perspective can thus increase uncertainty, because many new questions arise, for example, concerning the nature of the species interaction (II). These uncertainties are particularly important in simple systems in which species interactions may have considerable impact on the short-term dynamics of individual stocks. Different advice will be delivered when possible species interactions are taken into account. In a retrospective analysis the new perception of the stock dynamics will therefore make the previous advice seem inconsistent. The models for actual TAC recommendations fit the multispecies data less well than they do the single species data on those stocks whose apparent dynamics have changed, *e.g* those of Baltic herring and sprat (Table 4). The models should, however, only be taken as crude exploratory tools. The processes of producing advice are likely to introduce non-linearities in the relationship between variables. Thus models that include interaction terms may display counterintuitive parameter values for some variables. For example, the spawning stock biomass ends up with a negative parameter value in Baltic herring and North Sea cod (Table 4).

Simple management advice cannot be given when the ecological basis of the management advice has expanded to include communities, populations under selection pressure and whole ecosystems. As noted above, the number of possible pathways multiplies and there are no longer optimal solutions, only different trade-offs based on subjective preferences. Indeterminacy now enters management advice. Managers have not always found this helpful (Brugge and Holden 1991). To reduce the indeterminacy, boundary conditions such as a minimum viable population size can be introduced. Explicit recognition of trade-offs in the advice may also provide a way of reducing the indeterminacy through stakeholder negotiations. Now the relevant risks are not those related to achieving a particular target but those of violating boundary conditions.

5.1.2 Economic issues

In most parts of the world fish resources are exploited by private actors, which may be individual fishers or firms. Thus a complex set of factors relevant at the level of individuals determines the evolution of a fishery (Hartwick and Olewiler 1986, Hanna and Smith 1993, Gillis *et al.* 1995). Possible issues are changes in market structure and the price of fish or cost of fishing, including capital costs and variable costs as well as subsidies and taxes (Flaaten 1988, Conrad 1995). To be meaningful for management, socio-economic aspects should be considered in relation to ecological features of the exploited populations (Charles 1983, Plourde and Bodell 1984, Flaaten 1988, III, Welch and Noakes 1991).

The inclusion of fish prices is a simple extension of the biological models. The introduction of prices raises, however, the issue of time preferences with respect to income. Hence a discount rate must also be included (Clark 1976). Clark (1973) demonstrated that the discount rate can play a critical role in determining the level of exploitation to the extent that extinction can become "optimal" from the point of view of the firms exploiting the resource. The effects of the discount rate on the exploitation of a resource are, however, complex if the rate applies to society at large, because then it influences the costs of fishing (Charles 1983, Hanneson 1987). Although a universal discount rate cannot be specified (Stiglitz 1994) an analysis of possible discount rates can illustrate how time preferences translate into the way resources are exploited, by affecting, among other things, decisions on entry and exit and the preferred level of the resource (III, Campbell *et al.* 1993). It turns out that the discount rate can be important in determining the variability of the value of a fishery even when the biological variability is high and poorly known (III).

The time preference of income is only one aspect of fishery economics. Complex ecological systems permit many different types of response to changing conditions. These responses then affect the success of management alternatives (Fletcher *et al.* 1988, Hilborn and Walters 1992). A detailed analysis of these issues using *e.g.* production functions or game theory is beyond the scope of this study. My empirical data on major Baltic Sea fisheries show, however, that exploitation patterns change in response to economic conditions and that uncertainties concerning the future of a fishery can be mitigated by increasing understanding of economic issues (VII). This finding also supports the claim that concepts of economic risk and risk attitudes are important in management (III, Pearse and Walters 1992). Thus a simple maximisation of the expected yield in weight is no longer a valid measure of sustainability. The probability and length of periods of low income are important for the fishery and its managers.

5.1.3 Expanding the fisheries sector

The introduction of socio-economic issues is only one of many possible extensions. The choice of boundaries between fisheries management problems and other concerns is a decisive but subjective step in setting the framework for management advice. It is a value based judgement that cannot be objectively determined. The problem can be seen as an example of a dispute between world views in a collective decision making context (Amy 1987, Merkhofer 1987). Some actors see fisheries simply as a biological or technical activity producing catches of fish, whereas others would like to see fisheries as part of a general environmental issue. The significance of the choice is that it affects real or perceived imperfections of knowledge concerning processes in the system to be managed (Friend 1989).

Traditionally, fisheries management has been considered a separate activity from general aquatic resource management and environmental protection, but several tendencies have worked against this division. The recognition that species interactions are relevant in fishery management (Mercer 1982, Flaaten 1988, Sissenwine and Daan 1991) has raised the question of impacts of fisheries on other ecosystem components (VI, European Commission 1995, Johnson and Martinez 1995). Concern for protected species and the need to conserve biological diversity has added new elements to management advice (Kimball 1993, Schnute and Richards 1994, Lackey 1994, Martin 1994, Sherman 1994, VI, VII). This process has probably been encouraged by discussions within the fishery sector demonstrating that objectives restricted to the domain of high sustainable fish yields are too narrow (see below, Jentoft 1985, Nielsen 1992, Charles 1992).

I have shown (VI) that the perspective on resource management and the delivery of management advice changes as the field of management broadens. A narrow definition of the fisheries sector leads to self imposed-ignorance in fisheries management advice. This ignorance matters, because the managers have to act in a much broader context than that covered by narrow advice. The broader context, covering political, social, technical and economic issues, is likely to affect many societal decisions. Examples include those related to environmental protection, as these may eventually influence fisheries through restrictions. A broadening of the boundaries permits new interest groups to enter the discussions on fisheries management, but unless the management advice is broadened accordingly, the discussions will contain a large amount of uncertainty, ignorance and indeterminacy.

Table 4. Extended presentation of the simple linear models of IV. DF = degrees of freedom; *F* and *T* = test statistics; *P* = probability of accepting null hypothesis, SS = sum of squares, type III (SAS Institute 1989); Est. = parameter estimate; SE = standard error of estimate. The row under the subheadings (multispecies, single species) gives statistics for the whole model, also reported in IV. Other symbols as in subsection 3.1.

Baltic sprat

Analysis	Multispecies R^2=0.48; DF=11; *F*=4.08; *P*<0.05				Single species R^2=0.80; DF=11; *F*=18.9; *P*<0.001			
ANOVA	SS	DF	***F***	*P*	SS	DF	***F***	*P*
F	4481	1	4.32	0.06	5278	1	13.9	0.005
F·SSB	8030	1	7.74	0.02	14366	1	37.8	0.000
Model parameters	Est.	SE	***T***	*P*	Est.	SE	***T***	*P*
Intercept	25	23	1.08	0.31	52	12	4.27	0.002
F	-275	130	-2.10	0.06	-186	50	-3.73	0.005
F·SSB	1170	420	2.78	0.02	795	130	6.15	0.000

North Sea herring

Analysis	Multispecies R^2=0.81; DF=15; *F*=28.4; *P*<0.001				Single species R^2=0.80; DF=15; *F*=25.8; *P*<0.001			
ANOVA	SS	DF	***F***	*P*	SS	DF	***F***	*P*
R	72749	1	8.08	0.013	56842	1	5.85	0.031
F·SSB	97191	1	10.8	0.006	61433	1	6.32	0.026
Model parameters	Est.	SE	***T***	*P*	Est.	SE	***T***	*P*
Intercept	-61	41	-1.48	0.16	-22	38	-0.59	0.56
R	0.013	0.004	2.84	0.013	0.0086	0.003	2.42	0.031
F·SSB	0.47	0.14	3.28	0.006	0.44	0.28	2.51	0.026

Baltic herring, subdivisions 27-29

Analysis	Multispecies R^2=0.55; DF=11; *F*=3.30; *P*<0.08				Single species R^2=0.70; DF=11; *F*=6.12; *P*<0.02			
ANOVA	SS	DF	***F***	*P*	SS	DF	***F***	*P*
R	0.0032	1	3.69	0.091	0.0099	1	17.1	0.003
SSB	0.0040	1	4.63	0.064	0.011	1	18.4	0.003
R·SSB	0.0046	1	5.33	0.050	0.010	1	17.2	0.003
Model parameters	Est.	SE	***T***	*P*	Est.	SE	***T***	*P*
Intercept	0.65	0.21	3.06	0.16	2.2	0.44	4.87	0.001
R	-0.016	0.0084	-1.92	0.091	-0.18	0.044	-4.13	0.003
SSB	-0.31	0.14	-2.15	0.064	-2.0	0.48	-4.29	0.003
R·SSB	0.014	0.0059	2.31	0.050	0.19	0.046	4.15	0.003

Table 4, continued.

North Sea cod

Analysis	Multispecies R^2=0.70; DF=14; ***F***=8.61; *P*<0.003				Single species R^2=0.65; DF=14; ***F***=11.31; *P*<0.002			
ANOVA	SS	DF	***F***	*P*	SS	DF	***F***	*P*
F	1586	1	10.7	0.008				
SSB	8722	1	8.82	0.013	7249	1	6.86	0.021
F·SSB	7611	1	11.0	0.007				
CATCH					2451	1	2.32	0.15
Model parameters	Est.	SE	***T***	*P*	Est.	SE	***T***	*P*
Intercept	1227	360	3.40	0.006	5.41	40.6	0.13	0.90
F	-1493	460	-3.27	0.008				
SSB	-9.83	3.3	-2.97	0.013	0.735	0.281	2.62	0.021
F·SSB	14.1	4.3	3.31	0.007				
CATCH					0.373	0.245	1.52	0.15

Baltic cod

Analysis	Multispecies R^2=0.36; DF=10; ***F***=5.06; *P*<0.05				Single species R^2=0.40; DF=10; ***F***=6.00; *P*<0.04			
ANOVA	SS	DF	***F***	*P*	SS	DF	***F***	*P*
SSB	7195	1	5.06	0.051	7997	1	6.00	0.037
Model parameters	Est.	SE	***T***	*P*	Est.	SE	***T***	*P*
Intercept	109	42	2.61	0.028	98.8	42	2.35	0.043
SSB	135	60	2.25	0.051	158	65	2.45	0.037

5.2 Treatment of imperfect knowledge in advice

The recognition that fisheries and their management are not a biological activity alone but also a form of social planning brings economic variables, private actors and new sources of imperfect knowledge into the management process (Clark 1973; 1976, Hartwick and Olewiler 1986, Hilborn and Walters 1992, III, VI, Stephenson and Lane 1995). The non-biological knowledge that is relevant for resource management can be specified. In (VII) I showed that this knowledge is available, but we have to ask how this additional knowledge influences management advice and how imperfections in that knowledge can be dealt with. To answer these questions it is necessary to refer to the different views of fisheries management as planning (Fig. 1).

Different planning approaches make different demands on knowledge but, in addition, the advice is implicitly expected to deal differently with imperfections of knowledge. Ideal synoptic planning is preoccupied with facts and does not readily recognise imperfections of knowledge; management advice aims at predefined decisions and any knowledge that has no direct bearing on these decisions is useless. Management according to synoptic planning can deal with imperfect knowledge that can be expressed as quantitative risk estimates in calculations of expected values. Uncertainty that explicitly leads to the compilation of new facts can also be handled. Other types of uncertainty as well as indeterminacy and ignorance do not fit conceptually into synoptic planning.

Social incrementalism in the sense of Sager (1990) is process oriented. A mixture of qualitative and quantitative knowledge can be used in the process, the end points of which are indeterminate. Uncertainties and indeterminacies are the fuel for the communicative interactions that make up the planning. In (VI) I showed that communication between interest groups prompts questions that differ from those usually faced by fisheries management. In social incrementalistic planning management advice should highlight options and possible avenues of change rather than provide normative facts. Quantitative estimates of risk may lose a significant part of their meaning when dominated by uncertainties and indeterminacies. The determination of what is significant knowledge becomes a matter of debate. Taken to its extreme, the process can become a merry-go-round that demands continuous input of new knowledge only to keep up the momentum. Such communication for the sake of communication is not, however, what Habermas (1984) meant by communicative rationality, which always aims at common action.

5.3 The role of advice in achieving sustainability

A compilation of actual advice for typical problem stocks that I identified in (V) suggests that true improvement in the status of a fish stock is a rare event (Table 5). Recommendations to reduce the fishing mortality rate tend to be repeated year after year. For these stocks biologically based advice is unambiguous: the fishing effort should be reduced. Yet little improvement can be observed. This suggests that there are fundamental conflicts between the conservation and exploitation of fisheries resources and that these conflicts are only weakly related to imperfections in knowledge of the status of the fishery resources. In (V) I noted that improvement may occur, but usually only after a crisis leading to *e.g.* the complete closure of a fishery or some other such a dramatic event. Only then do all actors appear able to accept biological advice and take a reduction in effort as a starting point for reopening the fishery.

As long as the situation appears bearable, imperfections in knowledge of the stocks may be used as an excuse for delaying action. One can argue that a reason for the lack of action is "structural imperfection" in the management advice, *i.e.* neglect of knowledge outside the realm of the biological status of the fish stocks. Biologically based advice only states the problem; it does not suggest how the problem should be solved. The lack of action despite unambiguous biological advice indicates that many management problems are related to management and stakeholder objectives, or lack of them. The following section discusses objectives in greater detail.

Table 5. Distribution of fifteen years of recommendation for selected stocks according to recommendation categories. Recommendation: 0=unknown stock status, 1=status quo, 2=keep F, 3=reduce F, 4=reduce F substantially, 5=build up stock or 0 catch. Stock codes are given in Appendix 3. Data from ICES ACFM reports for 1978-1993 (Appendix 1).

Stock	Recommendation category					
	0	1	2	3	4	5
cod2224	0	1	1	3	10	0
cod2532	0	0	3	1	9	2
codfarvb1	0	1	2	6	3	3
codiiia	0	6	0	4	5	0
codnearc	0	0	4	5	7	0
grhalsub12	0	2	3	8	2	1
grhalvxiv	2	4	4	5	1	0
hadnearc	0	0	10	2	4	0
her2224	0	2	3	3	7	0
mackewes	0	0	3	7	5	0
plaiceviie	5	3	3	3	1	0
saitfarvb	1	1	5	3	5	0
saiticelva	0	0	12	3	0	0
saitnearc	0	0	7	6	3	0
salmbal	2	0	0	3	9	1
sebmarnear	0	11	1	3	1	0
sebmennea	0	4	3	7	2	0
soleviie	2	3	6	1	3	0
whitviia	0	4	4	2	5	0

6 Imperfect knowledge and management objectives

Fisheries management is typically said to aim at sustainability, but at a practical level this formulation has little meaning. With respect to imperfect knowledge several related questions emerge. The first refers to the objectives that determine the use of management advice, in other words: what advice is considered relevant? The second concerns the relationship between knowledge of the ecological system and the objectives, and the third the manner in which imperfect knowledge of objectives affects the choice of means in management. All issues concerning objectives are closely related to conflicts between stakeholders. In dealing with conflicts I shall assume that their origin can be cognitive (instrumental knowledge) or lie in interests and world views, including values (Amy 1987).

6.1 The nature of imperfect knowledge of objectives

As a major producer of scientific advice for fisheries management, ICES has frequently commented upon the lack of specified objectives (V). This does not necessarily mean that there are no objectives. They may appear to be lacking when they are so conflicting that managers and other stakeholders cannot agree on them. Alternatively, the actual objectives may be too much in conflict with publicly announced principles for resource use and thus stakeholders do not wish to make them explicit. In both cases those who deliver scientific management advice face uncertainty, ignorance or indeterminacy with respect to objectives. Forester (1993) argues that objectives are often indeterminate in complex social issues. Those who produce advice for decision-makers do not know how it will be used and may therefore give "wrong" advice. To avoid this dilemma, without, however, solving it, producers of biological advice have frequently adopted explicit "biological" objectives (V). In a similar vein, Feldman's (1989) "bureaucratic analysts" keep on producing papers they know that nobody will use.

Without the imperfect knowledge of objectives, the production and use of management advice would be straightforward. One could then predict that only stocks with very erratic dynamics would turn out to be problematic for management. To test this proposal I examined all stocks for which ICES has recently provided advice, and for which the available ecological knowledge meets basic quality criteria. Assuming that the strictness of the advice reflects the state of the stock, we can use this material to find out if recruitment variability and other variables can be associated with the general state of the stock.

Recruitment variability, the price of the fish and the total value of the catch were associated with the strictness of the management advice (Table 6). On its own, recruitment variability is only tentatively related to the strictness of the management advice, whereas the unit price of fish is clearly associated with the strength of recommendations to reduce fishing mortality rates. The stocks for which recommendations to keep or accept the present fishing mortality rates are made have a higher frequency of "cheap" fish than those for which reductions are recommended (Fig. 4). The high proportion of stocks of intermediately priced fish in the strong recommendations reflects the large number of severely exploited gadoid stocks.

The log linear models give some additional information. They indicate that not only unit price but also the interaction between unit price and total catch value and that between unit price and recruitment variability are important (Table 7). Interactions between unit price and total resource value are understandable. Small resources of valuable fish may end up being overexploited as may large resources of relatively cheap fish. A further exploratory variable could be the distance from fishing ports and markets. For example, North-East Arctic gadoid stocks are in a better shape than North Sea stocks. The crudeness of the data and the simplicity of the models are not able to satisfactorily model the variability in management advice as shown by the likelihood ratios. The results nevertheless strongly suggest that fairly simple, non-biological factors determine the state of stocks. In other words, sustainable use of resources does not depend on ecological characteristics or the ecological knowledge of the stocks alone.

The conclusion of my analysis is that interests and world views influence heavily the interpretation and use of ecological "facts". Historically it is possible that a consensus of ignorance has led to the overexploitation or destruction of renewable resources (Howarth and Norgaard 1995). For the stocks examined by ICES, however, this explanation is unlikely; economic incentives compounded by socio-economic or political issues such as regional equity are more plausible. Interviews with stakeholders can reduce some of the imperfections of knowledge that "hidden" agendas cause for those who provide management advice. For example, Hildén and Kuikka (1990) showed that different stakeholders

The question is, do those providing scientific management advice have to worry about the imperfect knowledge of objectives? If management is seen as an exercise in synoptic planning, objectives are a necessary driving force. Without objectives the means-ends scheme of synoptic planning becomes mean-

Table 6. Test statistics for associations between variables using table scores. n=sample size. Data in Appendix 2.

Variables associated with recommendation categories (RECCAT)	Cochran-Mantel-Haenszel statistic for general association between RECCAT and row variable.	DF	Probability
Recruitment variability class (RECRCLA); n=296	10.165	6	0.118
Class of total value of catch (VALCLA); n=319	10.025	6	0.124
Class of unit price of fish (PRICCLA); n=319	73.173	4	0.001
PRICCLA controlling for RECRCLA; n=296	71.047	4	0.001
PRICCLA controlling for VALCLA; n=319	65.758	4	0.001

Table 7. Maximum likelihood analysis of variance based on fitted log-linear models and the likelihood ratios of the models. Data in Appendix 2. Abbreviations as in Table 6.

Model (parameters omitted)	Analysis of variance	χ^2	DF	Probability
RECCAT+ PRICCLA	RECCAT	59.34	2	0.000
	PRICCLA	26.69	2	0.000
LIKELIHOOD RATIO		72.72	4	0.000
n = 319				
RECCAT+ PRICCLA+	RECCAT	34.18	2	0.000
RECCAT*PRICCLA+	PRICCLA	13.90	2	0.001
PRICCLA*RECRCLA	RECCAT*PRICCLA	57.68	4	0.000
	PRICCLA*RECRCLA	58.56	4	0.000
LIKELIHOOD RATIO		11.93	16	0.749
n = 296				
RECCAT+	RECCAT	27.50	2	0.000
RECCAT*PRICCLA+	RECCAT*PRICCLA	60.21	4	0.000
PRICCLA*VALCLA	PRICCLA*VALCLA	38.29	6	0.000
LIKELIHOOD RATIO		19.76	19	0.409
n = 319				
RECCAT+	RECCAT	21.59	2	0.001
RECCAT*PRICCLA+	RECCAT*PRICCLA	34.85	4	0.000
PRICCLA*RECRCLA	PRICCLA*RECRCLA	26.60	6	0.000
LIKELIHOOD RATIO		66.94	55	0.130
n = 296				

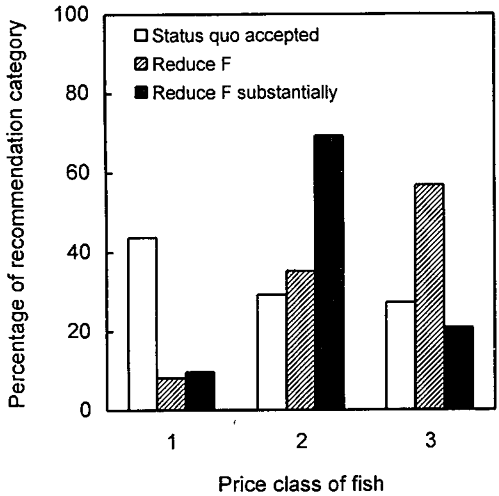

Fig. 4. The distribution of price classes of fish with categories of management recommendations.

ingless. If fisheries management is seen as an incremental social process stressing communication, objectives are not necessarily set *a priori*, but at some point they have to become explicit. Otherwise the scientific information on the fish stocks and fisheries does not have a meaningful reference. In (VI) I showed that neglect of relevant objectives led to a management conflict that could not be solved without better knowledge of the way in which the different stakeholders viewed the management problem. From the point of view of management advice the immediate effect of the broader perspective on objectives is that maximum sustainable yield of single fish stocks is replaced by reference points that are only indicative of directions of development (V).

6.2 Objectives and the ecological system

There is a close connection between the degree of complexity of the ecological system and the conceivable objectives. Maximum yield can be used as a dominant objective for a unit stock. A genetic stock removes the possibility of simple optimisation because at least two objectives are relevant: yields and the maintenance of genetic diversity. The problem of management is thus transferred to the sphere of "games of strategy" (Neumann and Morgenstern 1944). With the introduction of several species the dimensionality of the objectives is likely to increase even further. Attempts are sometimes made, especially in tropical regions, to regain simple objectives by analysing aggregate catches. Such aggregate objectives are, however, of doubtful validity in complex systems. For example, Laloë and Samba (1991) note that flexibility is a prominent feature of artisanal fishermen. The suggestion is that fishermen will readily respond to a number of signals in the exploited fish community and other external conditions such as those of the fish market; they will not adhere to a single (aggregate) objective defined in terms of catch. When one moves to whole ecosystem management all possibilities for simple objectives disappear.

Imperfections of knowledge are superimposed on the structural complexity of the ecological system. Lack of knowledge of the ecological system further reduces the opportunities for unique objectives. Risks are compatible with fixed predetermined objectives, but uncertainties, indeterminacies and ignorance concerning the ecological system make fixed objectives meaningless, even for the simplest of systems. A fish pond owner cannot manage her pond for a specific objective if she does not know what is going on in the pond ecosystem. The replacement of fixed objectives with biological reference points in recent fisheries management advice implies a greater acceptance of imperfect knowledge. The risks associated with the reference points can be calculated, but the sensitivity of different risk estimators to the assumptions (Cordue and Francis 1994) suggests that what has been called risk is in fact uncertainty, and too rigid a choice of risk estimators can lead to ignorance. This means that imperfections of knowledge at different levels interact. We will also find that uncertainty, indeterminacy and ignorance at the level of the ecological system generally create indeterminacies at the level of objectives.

6.3 Imperfect knowledge of objectives and the identification of means

Theoretically, fisheries managers have a wide range of means at their disposal. Hilborn and Walters (1992, p. 453) distinguish between the strategies of fish stock management, which determine how the catch taken from the fishery will be adjusted from year-to-year, and the actual tactical decisions, which set the rules for any particular year. Strategies and tactics deal with regulation, economic incentives and active intervention in the resource base. Regulation is commonly divided into output control, which is based on catch quotas, and input control, which is based on some form of regulation of the fishing effort through licensing or regulation of production through temporal, spatial or technical restrictions or economic incentives. Economic incentives can include subsidies or taxes and payments or the creation of new markets such as individual transferable quotas. Fish stocking, pest control and physical restructuring are examples of active interventions in the resource base. To what extent is imperfect knowledge of objectives related to advice on choice of means?

In an instrumental mean-ends scheme only the objectives are value driven (Forester 1993) and thus the choice of means would be an optimisation exercise once the objectives are known. Imperfect knowledge of objectives complicates this kind of optimisation. In (V) I showed that producers of instrumentally oriented management advice have demanded more explicit objectives, the idea being that clearly expressed objectives would simplify the production and delivery of management advice. In a planning perspective this illustrates a strongly instrumental view of management. When the perspective shifts to include communicative rationality, means are also value laden. Agreement on what is sustainable for a particular fish stock is not then sufficient for guiding the production of management advice. Salmon management in the Baltic Sea is a case in point.

The International Baltic Sea Fishery Commission has agreed that safeguarding naturally spawning salmon stocks is a key objective. In (VII) I showed that, for all practical purposes, perfect knowledge is available showing that this means a substantial reduction in the fishing mortality rate. With synoptic planning it would immediately be possible to introduce whatever restrictions were found necessary to reduce the mortality rates. Any attempts to do so have, however, encountered substantial difficulties, because the means available to achieve the reduction are highly value laden. In a synoptic view this could be interpreted as an indication of additional objectives such as the number of fishers and the regional distribution of catches. Scientists could solve the problem by taking these additional objectives into account with the help of, say, multi-attribute decision support. According to this view, problems arise because knowledge of the other objectives is less than perfect. Instrumental synoptic management thus recognises, but does not accept, uncertainty and ignorance with respect to objectives. Indeterminacy has no place in this approach except in the form of poorly defined optima in relation to given objectives.

In a social, incrementalistic perspective it is impossible to agree once and for all on well defined objectives. Knowledge of objectives remains less than perfect because every choice is dependent on interests, world views, past events and changing expectations. Conflicts cannot be neatly delimited to a discussion concerning the ends of management. Even if one could agree on a catch quota for a region, decisions on who should take this quota remain. Decisions on who will be able to fish may actually influence prospects for reaching an agreement on the quota. Means and ends become intricately linked and the main task of management is to reduce ignorance and resolve indeterminacies. Stakeholder involvement thus becomes a key in reducing imperfect knowledge concerning objectives. A social incrementalistic process demands this type of interaction in the choice of means. The task of management advice is then to facilitate and provide input to the process by analysing alternatives. Statements restricted to a “preferred catch level” or to observations that a stock is considered to be within or outside “safe biological limits” reflect instrumental rationality and are difficult to reconcile with social incrementalistic view of fisheries management. Such statements add relatively little to an examination of alternative courses of action.

The foregoing indicates that advice on means cannot be separated from objectives and the planning perspective. At a general level, output control is mainly related to a synoptic planning perspective whereas input control fits more easily in with both social incrementalistic planning and synoptic planning. The type of input control, however, obviously depends on the planning perspective. Synoptic approaches call for a central power that determines the choice of input control based on “objectively” calculated advice, whereas social incrementalistic approaches can be founded on cultural processes based on stakeholder communication and local enforcement.

7 Decision support for sustainability

In the preceding sections I examined the appearance and role of different types of imperfect knowledge in fisheries management. In this concluding section I return to sustainability and the question of how fisheries scientists can deal actively with imperfections in the knowledge used and presented in advice to managers. The task of the scientists is demanding. Several studies have shown that the human mind is susceptible to errors of judgement when knowledge is imperfect (Tversky and Kahneman 1974, 1981). Management advice and managerial decisions in fisheries are no exception.

A starting point for the discussion is that all aspects of imperfect knowledge covered in the individual studies (I-VII) and in this summarising analysis can be relevant in management for sustainability (Table 8). Their relative importance depends on the management context, including the nature of any management conflicts. The effects of imperfect knowledge concerning boundaries, objectives and key processes as they appear in input variables, parameters and model structure will depend crucially on the type of management decision (I, II) and on the particular characteristics of a fishery (IV). Similar conclusions have been reached by Bradford and Peterman (1989), Lapointe *et al.* (1989) and Frederick and Peterman (1995). Analogous situations arise in water quality modelling (Varis *et al.* 1994). Thus it is not meaningful to make general quantitative statements concerning the relative contribution of different sources of imperfect knowledge. Instead I first explore the role of the management context and then use the conclusions to examine some of the controversies concerning the prospects for sustainable fisheries. Finally I offer possible quality criteria for management advice.

7.1 Interactions between imperfect knowledge and the management context

A crude grouping recognises strategic and tactical decisions (Table 9). In fisheries, strategic decisions set the general rules for exploitation and usually have a time horizon of several years. Tactical decisions are concerned with year to year, or sometimes even seasonal, adjustments of the exploitation. An important characteristic of a management decision or plan is thus its temporal perspective. The distinctions between strategic and tactical decisions are, however, not alway clear cut. For example, closed areas can be used at a strategic level, but they can also be used as tactical decisions to direct exploitation in the short term (Table 9).

7.1.1 Strategic decisions

The longer time horizons of strategic decisions reduce the relative importance of numerically quantifiable risks. The chosen ecological perspective will further affect the relative role of different imperfections of knowledge (Table 10). The differences become particularly evident, when the general ecological perspective is combined with typical management decisions (Table 9). For example, restrictions on access in a system managed as a whole ecological community are typically indeterminate. No knowledge can, as such, give an unambiguous answer on how the restrictions should be enforced. Furthermore, even the quantitative level of the access limitation is indeterminate, as it depends on the type of exploitation. On the other hand, access limitations for a homogeneous fishery concentrating on a single major stock are subject mainly to uncertainties and risk. Similar differences arise in such matters as the development of subsidies for different types of fishery.

The complex interactions between the temporal and ecological perspectives, and the different kinds of managerial decisions, can fuel conflicts and create confusion. Stakeholders will find imperfections of knowledge at many different levels and exploit them fully in identifying and evaluating strategic decisions. For example lack of knowledge of ecological processes can be used as an argument for retaining the status quo or for implementing specific solutions. Both arguments were used in the Shetland case (VI). In other situations socio-economic issues become a central argument. In a conflict situation stakeholders can use lack of knowledge as an argument against any management action that does not serve their (perceived) interests or correspond to their world views. The suggestion is often that a reduction of risks and uncertainties is the main task of the scientific advice.

Although many arguments in fisheries management tend to focus on what may superficially appear to be risks and uncertainties, conflicts at a strategic level can seldom be solved by overcoming

Table 8. Summary with examples of different types of imperfect knowledge in different management tasks.

Management task	Risk	Uncertainty	Ignorance	Indeterminacy
Setting boundaries	Deliberately specified boundaries of system and fish resources	Poorly specified boundaries of resources and their exploitation	Boundaries based on flawed perception of system	Evolving system without recognisable boundaries
Specification of objectives	Multiple objectives with weights	Unspecified weights for multiple objectives	Major actors and interest groups are excluded	Ongoing debate on multiple objectives
Identification of key processes	Sampling of variables	Choice of variables and lack of data	Unrecognised processes and variables	Evolving processes
	Competing model specifications with prior probabilities	Competing model specifications with nonin-formative prior probabilities	Lack of knowledge of possible models	Future events determine course of events and appropriate models
Evaluation of means and assessment of outcomes	Alternative outcomes with prior probability	Alternative outcomes with noninformative prior probabilities	Surprising outcomes	Evolving system
	Choice of risk levels	Specification of unacceptable risk	Lack of concern for the "unthinkable"	
	Random variability affects possibilities for control	External events limit opportunities for control	External events and attempted control cause unexpected effects	Choice of means does not on its own specify course of development in any way

such imperfections of knowledge. One reason is that difficult ethical questions tend to emerge whenever management becomes operational (Donaldson 1992). For example, the question of who should be allowed to exploit a resource cannot be solved by analysing risks and uncertainties. Ethical questions are also encountered in the simplest of risk assessments. Even if the risks can be numerically calculated, the question of what is considered an acceptable risk will still be a matter of debate.

These observations suggest that, from the managers' point of view, ignorance and indeterminacy are particularly relevant at the level of strategic decisions. For fisheries scientists the situation easily becomes frustrating. Scientists are trained to compile data and have developed methods for the transparent treatment of uncertainties and risks. However, the risks and uncertainties are often interpreted as proof of ignorance by those who feel that the managers' strategies make them losers. The overall result may be a strengthening of indeterminacy. The observation that conflicts involving fishers, managers and politicians typically remain for long periods at the status quo deplored by all stake-

Table 9. Examples of tactical and strategic decisions in fisheries management.

Focus of managerial decision	Tactical	Strategic
Catch and distribution of catches	- Yearly or seasonal quota	- Quota strategy: Fixed *F*, fixed escapement, minimum stock biomass - Distribution strategy: First come - first served, individual transferrable (share) quota, personal permanent share, areally or temporally distributed quota - Risk level/degree of precaution
Effort	- Short-term effort limitations: days at sea, quantity of gear, closed seasons, varying closed areas based on by-catch restrictions, protection of young fish etc.	- Access regulation: Open access, licence, transferrable licences. - Permanent gear control (type/amount) - Permanent temporal and spatial restrictions - Risk level/degree of precaution
Resource enhancement	- Number of fish to stock - Operational restoration	- Choice of species and stock - General stocking levels - Restoration strategies
Development of fishery	- Yearly management procedure - Amount of taxation or subsidy	- Management institutions and procedural rules - Type of taxation or other payment for resource use - Type of subsidy - All above strategic decisions

Table 10. Types of imperfect knowledge in relation to the characteristics of planning and the resource system

Temporal perspective in planning/ management	Unit stock population	Evolving stocks	Multiple	Community
Short term	Risk	Risk, uncertainty	Uncertainty, indeterminacy	Indeterminacy
Long term	Risk, uncertainty	Uncertainty, ignorance	Ignorance, uncertainty	Ignorance, indeterminacy

holders, supports this conclusion.

The persistence of the status quo is shown by findings concerning the use of management advice (V, Table 5) and by an examination of the production and use of salmon management advice (Hildén 1990). Reducing uncertainties and risks through improved knowledge of fish stocks or their exploitation may thus not bring any relief unless it is accompanied by action that deals with ignorance and indeterminacy. This is not simple when the reasons for the ignorance and indeterminacy are poorly defined or unknown objectives, interests and world views.

7.1.2 Tactical decisions

Short-term tactical management decisions tend *per se* to be less sensitive to the overall perception of the dynamics of a fishery and to other lack of knowledge than strategic decisions. The reason is simply that short-term management decisions have a strong tendency to aim at the status quo using simple predictive models (Pope 1983). In systems comprising a large number of species with partially overlapping ecological functions, most models predict small changes from year to year, and the systems appear to be resilient at an aggregate level. For these systems lack of knowledge of say, species interactions may be of minor concern in short-term decisions. This is not true of systems dominated by a few predators and prey with considerable fluctuations in year-class strength (IV). In these fisheries short-term tactical decisions can have a decisive effect on the dynamics of the fish community. Model uncertainty, including uncertainties related to species interactions, is significant, because alternative perceptions of reality may lead to qualitatively or quantitatively different short-term management advice (IV). Similar changes may occur when new methods are introduced for estimating stock size, fishing effort or mortality rates. Some of the variations in management advice shown by Table 5 are related to this type of uncertainty.

As long as the status quo is a starting point, ignorance and indeterminacy play a smaller role than risk and uncertainty in tactical management decisions. The status quo neutralises, at least temporarily, many of the conflicts at the level of interests and world views. If, however, managers attempt to use short-term tactical decisions as part of a long term strategy, then other types of imperfect knowledge gain importance. In the Baltic Sea, for example, many years passed before the International Baltic Sea Fishery Commission could agree on quotas for cod and salmon, showing that tactical decisions were interpreted as having strategic importance. Therefore they also unleashed conflicts of interests and world views.

7.2 Types of imperfect knowledge and the sustainability controversy

The collapse of many large pelagic stocks in the 1960s and 1970s has been explained by referring to the tragedy of the commons (Hardin 1968) and the concurrent neglect of proper scientific advice (Gulland 1987, Rosenberg *et al.* 1993). This line of thought suggests that sustainability could have been achieved if biologically based management advice had been taken. Alternatively one can argue that overexploitation is inevitable because there is no consensus on the state of resources (Ludwig *et al.* 1993) or because the wrong type of management has been proposed by biologically based scientific assessments using a flawed paradigm (Wilson *et al.* 1994).

The arguments concerning the nature of fisheries management problems have a fundamental impact on the perception of decision support for resource management. If current fisheries science cannot provide guidance for sustainable use, a major reorientation of the research effort is called for. If, on the other hand, non-sustainable exploitation is mainly a problem of communication, the main emphasis should be on the education of decision-makers and on promoting discussion between stakeholders.

The sustainability controversy can be set in perspective by examining the role of imperfect knowledge in relation to the ecological and planning aspects of management. For managers, the justifiability of assumptions concerning the dynamics of the resource (single unit stock, evolving populations, limited number of interacting species, or whole communities) is a central ecological aspect. Planning is closely related to the type of feasible control and decision-making. Synoptic instrumental planning suggests full control of both ecological and socio-political variables, whereas social incrementalistic planning represents the conceptual opposite.

Using the different concepts of imperfect knowledge, I suggest that those who argue that present fisheries science and management cannot achieve sustainability, are claiming that ignorance is the key problem. Managers cannot do the right thing because knowledge of all relevant variables is not available and cannot even be made available. In other words: fisheries managers are unable to act as synoptic planners. Those who see sustainability as a difficult, but nevertheless attainable goal point out that risks, and to some extent, uncertainty are the dominant problems, and that fisheries science is learning to address them. Here, too, synoptic planning is an implicit assumption. Scientists direct their advice to managers or decision makers, who are assumed to be able to control exploitation if they "want to". The questions are then: what evidence is available to support the different claims and what management situations produce the evidence?

As subsection 7.1 shows, risks often dominate short- and intermediate term tactical resource management decisions concerning well researched fish stocks. About 60 of the fish stocks assessed by the International Council for the Exploration of the Sea, or half of the total stocks, fall in this category (ICES 1994a). For these, risks can be quantified (Varis *et al.* 1993, Rosenberg *et al.* 1993, VII) and set in relation to specific reference points (V). However, few of these stocks provide opportunities for truly synoptic planning. In reality, managers' control over exploitation is often weak and to justify their actions they have to negotiate with and convince numerous stakeholders. This is foreign to synoptic planning, which, by definition, is conflict free at the level of values and world views, and which would deal with *e.g.* risks in the form of unidimensional expected utilities. The necessity of negotiations hints at indeterminacy and ignorance, allowing for value conflicts between stakeholders. The question, risk to whom, becomes a complement to any unidimensional risk assessment. As long as risk and uncertainty dominate over ignorance and indeterminacy, synoptic planning can, however, be used as a conceptual reference, and systematic analyses of the resources and their (potential) variability can support decisions towards sustainability.

Ignorance and indeterminacy are generally dominant in long-term strategic decisions and all decisions concerning poorly investigated and understood resources (V). In the case of ICES (1994a), half the stocks were in a category for which no, or only unsatisfactory, quantitative estimates were available. In long-term decisions ignorance and indeterminacy arise partly because of lack of understanding of long-term changes in the resource, but also because of the interaction of different historical trends causing unanticipated changes. These situations do not even remotely correspond to synoptic planning. For example, problems related to Finnish salmon management can, with hindsight, be attributed to a historical development involving decisions on ownership of waters and fishing rights, subsidies for rural occupations and general rural policy, exploitation of rivers for hydro power, artificial stocking and institutional solutions within the fisheries administration (Nybacka *et al.* 1991). An additional aspect of ignorance is its subjective nature. Separation of fisheries management from other environmental management issues has in the past led to a self-imposed ignorance about other values; the outcome may be serious conflict (VI).

A formal risk analysis may give a false sense of security and controllability when ignorance and indeterminacy are dominant. More useful decision support is based on precaution, for which criteria and rules of thumb can be specified (Garcia 1994, FAO 1995a), or an adaptive approach (Holling 1978, Walters 1986), which can gradually reduce ignorance and indeterminacy. When ignorance and indeterminacy are dominant, scientists cannot give *a priori* quantitative estimates of the degree of precaution. Quantitative risk analysis as advocated by Rosenberg *et al.* (1993) can provide additional insights but cannot resolve the fundamental problems of management. The problems are not due to risks but to issues connected with collective decision making, power, legitimacy of management and flexibility in the face of changing conditions.

The conclusion of this discussion is that the two seemingly opposite arguments concerning the role of science in supporting sustainable fisheries management and its capacity to do so focus on different types of situations. These differences arise at ecological, socio-economic, political and scientific levels. At the ecological level the stocks for which risk analyses are relevant characteristically have a sufficient dominant or independent role in the ecosystem. Scientists and managers can therefore treat them as unit stocks. The opposite conditions prevail in exploited ecosystems with numerous stocks interacting in many ways. At the socio-economic level the stocks whose management benefits from risk analyses are distinguished by well defined and controllable fleets with considerable investments in the fishery. The other end of the scale is represented by a wide range of small-scale exploiters, who are an anonymous crowd from the point of view of centrally appointed managers. Sophisticated risk analyses have little meaning for this type of fisheries. Instead, community-based management may be the key to sustainability. At the scientific level the available data and understanding of the stock determine the differences. Quantitative risk analyses have little to say when next to nothing is known about the stock(s). The scientific level interacts with the two other sources of differences. It is easier to obtain good data from large dominant stocks exploited by well defined fleets than from numerous small stocks exploited by a wide range of fishers.

My analysis shows that approaches to sustainability cannot have universal validity. Systematic analysis of the imperfections of knowledge and sources of conflict can, on the other hand, provide a coherent and simple explanation as to why decision support for management can fail to improve the status of the fish stocks. Thus Wilson *et al.*'s (1994) assumption of deterministic chaos is not needed in an evaluation of management paradigms and relevant decision support. Careful analysis of the true nature of imperfect knowledge is, however, necessary. If scientists or managers confuse the different types of situations, problems are inevitable. Too strong an emphasis on ignorance or indeterminacy when controllability is reasonable leads to confusion and inefficient use of available knowledge. Presentation of knowledge as if synoptic planning were feasible, when an incrementalistic approach is the only one appropriate, may lead to serious errors in managerial decisions.

7.3 Decision support, interactions of imperfect knowledge and collective choice

Providing support for resource management would be easy if all problems could be neatly categorised so that optimal solutions could be found for each type *vis-à-vis* decision support. My study, the cases examined and the review of the relevant literature have shown that management for sustainability is characterised by complex combinations of risk, uncertainty, ignorance and indeterminacy interacting with different conflict dimensions. Ignorance and indeterminacy come on the scene even though accumulated wisdom indicates that the management issues can be handled in a quantitative risk framework. For example, the discovery of a syndrome affecting reproductive success of the Baltic salmon (ICES 1994a) made earlier assessments of sustainable exploitation meaningless. Another example is technological innovations in fishing, which introduce indeterminacy into the implementation of specific management measures (Suuronen 1995).

The interactions of different types of imperfect knowledge complicate the question as to the kind of support one can demand for resource decisions aiming at sustainable use of natural resources (Charles 1994, VII). An obvious starting point is that all decisions that can significantly affect the utilisation of resources should be based on "best available knowledge". The UN Convention on the Law of the Sea demands "best *scientific evidence* available" (United Nations 1983, my italics), but this analysis and the underlying studies (V, VI) have shown that instrumental scientific rationality may not be sufficient in conflict-laden resource decisions in the face of risk, uncertainty, ignorance and indeterminacy.

My study indicates that the multidimensionality of the problem of imperfect knowledge precludes universal solutions. Efforts to reduce imperfections of knowledge will depend on the role of management in a planning perspective. In fisheries management, the decision analysis includes collective decision-making with incompatible world views (von Winterfeldt and Edwards 1986, Merkhofer 1987). Arrow's impossibility theorem states that there are no rules of social choice that could solve the problems and at the same time satisfy four generally accepted properties of rationality and democracy (Sen 1988). This led Sager (1990) to suggest that a trade-off between some of the properties, notably a compromise between logicality and fairness, is necessary. However, before rushing to this conclusion we should examine the conditions of Arrow's theorem and the properties of resource management planning.

Sen (1982) distinguishes between different kinds of aggregation in collective decisions. Much of fisheries management is an exercise that demands the aggregation of individual interests into social decisions. Fisheries managers deal with interests that can at least partly be measured on a common scale, as shown by the Shetland case (VI). The classical framework of the impossibility theorem does not seem appropriate for this type of problem and Sager's (1990) compromise may not be necessary. Instead, bargaining becomes an issue. Only certain special cases of strategic fisheries management may be subject to the full gloom of Arrow's theorem. Examples include the choice between different fisheries policies, *e.g.* should recreational fisheries be favoured over commercial fisheries? Such a decision involves an aggregation of individual welfare judgements to a collective decision. Fairness will be an important criterion for decisions of this type.

What do the foregoing points of view mean for the decision support that scientists are asked to provide for fisheries management? One observation is that the stakeholders in fisheries management will seldom be able to accept normative statements of *the* correct solution as a basis for decisions. Bargaining is an important part of management and therefore the decision support should give stakeholders a chance to examine the consequences of different options. To facilitate the bargaining, stakeholders and scientists may be able to agree on the type of knowledge that is needed and also on the criteria that the decision support should meet. This will obviously not stop different stakeholders from arriving at different conclusions when they use the material provided by scientists.

7.4 Quality criteria for decision support

Keeney and Raiffa (1976) proposed the following criteria for examining the quality of decision trees: completeness, operationality, decomposability, absence of redundancy and minimum size. These quality criteria are applicable to knowledge serving instrumental rationality and means-end schemes. Merkhofer (1987) suggested that decision-aiding approaches for social risk management can be evaluated by considering a broader set of quality criteria: logical soundness, completeness, accuracy, practicality and acceptability. Of these, logical soundness, or rather consistency, completeness and accuracy refer to instrumental rationality in the management of natural resources. Practicality and acceptability also include communicative rationality. Both instrumental and communicative rationality appear to be necessary to cover the full range of questions and situations that arise in efforts to achieve sustainability.

The findings of my study support a combined list of quality criteria for management advice:

- Completeness
- Parsimony (absence of redundancy)
- Logical consistency (soundness)
- Accuracy
- Practicality
- Transparency
- Neutrality

I have excluded acceptability as a criterion of management advice since acceptability is something that can be sought in the management process, before or after managerial decisions. Acceptability cannot be required of the advice *a priori.* Advice immediately accepted by all stakeholders is not really advice at all, only a confirmation of past actions and prejudices. Instead I introduce neutrality as a criterion. It is based on the observation that indeterminacy is important in fisheries management. The advice cannot as such resolve this indeterminacy, but it can provide input into the collective processes that choose courses of action. To achieve this it must avoid creating or enhancing conflicts by presenting one-sided views or by rejecting alternatives without justification. The criterion is coined neutrality rather than objectivity because it is not a question of rejecting subjective statements. Instead this criterion demands a fair presentation of different and possibly conflicting subjective conclusions. Neutrality of advice may thus be a prerequisite for acceptability.

The completeness criterion is highly demanding in fisheries science. Many reference points and approaches have been developed for the biological subsystem (Smith *et al.* 1993, Hutchings and Myers 1994, Myers et al. 1994, Caddy and Mahon 1995), but in (VII) I argue that this is not sufficient. The biological reference points cover only a limited part of the system to be managed. Furthermore, lack of knowledge of the status of the resource in relation to the biological reference points may give an inconclusive assessment where other variables could give a better evaluation and ranking of management options. The crisis in the Canadian Atlantic cod fishery supports this argument.

In the Newfoundland cod fishery the biological reference points were so uncertain that managers were receiving ambiguous signals almost up to the moment the fishery collapsed (Doubleday 1993, MacKenzie 1995) whereas socio-economic information had already shown clearly in the 1970s that any management decision leading to an increase of fishing effort would worsen the situation for the fisheries (Copes 1983). My own findings that the state of the resource, as revealed by the strength of the recommendation, is related to economic variables also point to the importance of variables outside the biological realm (subsection 6.1). Results showing that better biological analysis would have revealed the true situation (Hutchings and Myers 1994) do not invalidate the finding that several types of reference points are needed for proper evaluation of management options (VII).

The problem with the completeness criterion is that the number of variables that have some bearing on the sustainability of fisheries and that are or can be measured exceeds the capacity of any actor in the resource management process. The parsimony criterion demands removal or aggregation of redundant variables. It is not possible to state universally how the aggregation should be carried out or how many variables should be retained. Olli Varis (Helsinki University of Technology, personal communication) has pointed out that this amounts to finding the correct level of resolution, which may or may not be achieved through aggregation. The issues depend on the context, and different levels of aggregation are appropriate for different decisions, *e.g.* strategic decisions on the control of effort and tactical decisions on the next year's quota demand different resolution.

The logical consistency criterion requires that the advice follows logically from stated premises. This criterion is demanding because there are competing theories concerning both the ecological system and the development of the fishery. As scientific advice is generally produced in a process that combines objective calculations and subjective judgements, that advice may be inconsistent.

The accuracy criterion is a double-edged sword. I noted above that demands for accurate knowledge can be made indefinitely. The criterion must therefore be viewed in the light of the consequences of the decision. The reversibility of the consequences is a particularly apt reference. If the consequences may be irreversible, the demands placed on accuracy will be greater than they would be if all consequences were easily reversible. By stating and justifying the chosen level of accuracy, scientists can recognise the accuracy criterion without succumbing to demands for the endless collection of minutiae. Accuracy also means that the variables used in the methods for assessing the state of the fish stocks and the fishery accurately reflect what they are thought to measure. For example, there are several ways of measuring fishing effort, from days at sea to hours of fishing, and their relation, if any, to the fishing mortality rate may differ substantially. This is critical when the development of fishing effort is being estimated or when advice on appropriate fishing effort is delivered.

The practicality criterion recognises the possibilities and limitations of fisheries management. In the Baltic Sea, the fish community and fisheries alike could benefit from biomanipulation that would reduce the stocks of sprat (Rudstam et al 1994); but the advice to increase fishing for sprat is useless as long as there are no markets capable of absorbing the products of an extensive sprat fishery.

Transparency is a key to stakeholder participation. The reasoning behind the advice must be understandable. In addition, transparency requires explicit recognition of lack of knowledge (Smith *et al.* 1993). My study has provided three justifications for an analysis of lack of knowledge in the identification of means and the assessment of outcomes. First, the analysis can provide insights into how lack of knowledge and its subjective resolution in, say, models can distort the perception of reality and interpretations of observations (I, II, Lapointe *et al.* 1989). Second, it permits an initial assessment of the feasibility of management decisions. For example, in the case of the Shetland Islands, uncertainties with respect to recruitment, on which the fishery is largely dependent excludes quota management (VI). Third, an assessment of the lack of knowledge of consequences gives a fuller understanding of the limits of management, *i.e.* the controllability of the system. For many complex systems, the actual possibilities of control are often limited to crude directional control relative to the present. It may, for example, be possible to state that scrapping subsidies will decrease the fishing effort rather than increase it, but the magnitude of the change cannot be determined with any reasonable accuracy.

7.5 Applying the criteria

When the focus of decision support goes beyond instrumental synoptic planning with criteria for practicality, transparency and neutrality, the decision support can no longer be separated from the institutional framework for management. Ideally, management institutions should create incentives to reduce ignorance, uncertainty and indeterminacy (Pearse and Walters 1992). Ignorance arises especially from the interaction of different types of imperfect knowledge, and is often fuelled by conflicts related to different world views. Therefore the first step towards improved knowledge involves a broad definition of actors and interest groups coupled with opportunities for participation. This may require changes in the way management decisions are taken.

A detailed analysis of how different institutional arrangements can support participatory fisheries management is not the task of this study. I simply note that in democratic societies broad stakeholder participation appears to be the only way to reduce indeterminacy concerning objectives and to increase acceptability of advice. Furthermore, the multidimensionality of complex issues effectively blurs the difference between lay and expert knowledge (Wynne 1996). For example, a biologist who is an expert in the dynamics of fish populations is often a lay person when it comes to the operation of fishing gear or functioning of fish markets. Accepting communicative rationality can, paradoxically, allow a strengthening of instrumental resource management, *i.e.* a move from ignorance to uncertainty and risk. This should make managers move towards what Schön (1983) calls the "reflective practitioner". On the other hand, fixation on instrumental synoptic planning as a model for all fisheries management is likely to produce additional support for Ludwig *et al.*'s (1993) pessimistic conclusion on the inevitability of overexploitation.

Examination of the criteria in actual management advice suggests the following. Short-term biological advice in the format provided by *e.g.* the Advisory Committee on Fishery Management (ACFM) of ICES (1996) fulfils many of the criteria. ACFM gives information on the main characteristics of the state of the stock and the fishery, and provides explanatory notes on any lack of knowledge. ACFM also provides material that allows for an examination of options in biologically oriented short-term tactical managerial decisions. The persistent problems of many fish stocks and fisheries suggest, however, that the main problems are found at a strategic level (Table 5). The production of knowledge for strategic managerial decisions is much weaker, being restricted to general statements on trends and to biological reference points. Individual examples of comprehensive analyses can be found, but the decision support is often severely limited in scope and unable to meet more than a few of the quality criteria. Developing both the institutional setting and the decision support for strategic decisions in fisheries is likely to be one of the keys to sustainability in fisheries.

8 Conclusions

Management of fishery resources cannot avoid dealing with risk, uncertainty, ignorance and indeterminacy. These imperfections of knowledge are relevant in the different tasks in the management process. They affect the setting of boundaries, the specification objectives, the analysis of key processes, the identification of options, the choice of means and the assessment of outcomes. They interact and stakeholders exploit them in conflicts concerning managerial decisions to turn decisions in particular directions. The conflicts can confuse the different types of imperfection. Stakeholders find it, for example, easy to raise doubts about ecological systems, and thus conflicts may appear to focus on knowledge of the fish communities, even when conflicts concerning interests and world views would be more important. Those who provide material to support managerial decisions may end up providing irrelevant details if they do not grasp the true nature of the missing knowledge.

It is possible to systematically examine imperfections of knowledge and sources of conflict in fisheries. My study has shown that this must be done through all the tasks of the management process if the problems of management for sustainability are to be correctly understood. The diversity of the imperfections of knowledge also means that it is not possible to use a single method or methodology in the analysis. Instead combinations of qualitative and quantitative approaches have to be used.

Management advice can deal with some of the problems associated with risks, uncertainties, ignorance and indeterminacy. It can estimate risks, identify and reduce uncertainties, develop contingency plans and adaptive processes to deal with ignorance and also provide background material for a resolution of indeterminacies. To meet these demands the management advice must address several disciplines and flexibly perceive the interactions between different management tasks. If those providing advice choose too narrow a focus they will find themselves victims of self-imposed ignorance. The connections between disciplines are inherent in all attempts to specify sustainable development.

Sustainable development comprises ecological, economic and cultural sustainability. Management advice that improves the understanding of the fish resources and their dynamics is necessary for an instrumental approach to ecological sustainability. This study has shown that progress towards sustainability cannot universally be based on the idea of synoptic planning. Synoptic planning would amount to treating the management of fisheries as if it were a control theory problem. Only special cases, such as short term managerial decisions concerning well defined stocks with few exploiters or fish pond management, can come close to the synoptic ideal. Normally the large number of stakeholders and the potential for conflict in different dimensions of knowledge, interests and world views force the management process towards social incrementalism. The large spectrum of world views and values defies synoptic control. Communicative rationality has to be accepted as a potential driving force, especially in the preparation of strategic managerial decisions. This is essential as willingness to take strategic decisions appears to be a precondition for sustainability. Otherwise the deadlock of status quo is likely to remain.

When management advice becomes part of a process that recognises communicative rationality, the decision support cannot be prescriptive in the sense of unambiguous presentations of single optima. Management will involve negotiations that, at best, take into account the constraints imposed by the biological system. This analysis and its supporting studies have examined some approaches and tools for dealing with imperfections in knowledge of the ecological systems which fisheries exploit. I have shown that risk analyses on their own are unlikely to bring much relief except when all other sources of imperfections of knowledge have been greatly reduced. If other imperfections of knowledge are neglected, a risk analysis may create a false sense of security and address minor issues that cannot lead the exploitation towards sustainability.

Quality criteria for sustainability-oriented management advice can be derived by examining combinations of planning theory, ecological theory and an analysis of conflicts. The criteria ensure that the advice meets instrumental demands concerning management measures, but they also take into account the communicative rationality that is needed to deal with imperfections of knowledge, which otherwise may foster conflicts. As our knowledge of ecological systems is far from perfect, ecological issues have to be dealt with at both an instrumental and a communicative level. Many of the frustrations experienced by the producers of fisheries management advice may be due to too strong an emphasis on the instrumental level.

My study has focused on issues that arise in dealing with imperfect knowledge in fisheries management aiming at sustainability. It has identified the connections between management as a planning activity and the key ecological concepts of exploited fish resources. Its emphasis has been on a conceptual clarification of relevant issues, using the supporting studies as cases to illustrate different points in a vast field. Additional studies are needed to develop and apply methods that meet the criteria of proper decision support in fisheries management, at the level of strategic decisions in particular. Universal solutions are unlikely as soon as non-instrumental rationality is accepted. The management of and management advice for major fish stocks in the Baltic Sea are, and have to be, different from those concerning coastal artisanal fisheries or lake fisheries dominated by recreational fishers, even though sustainability is a common objective. The challenge of methodological development is flexibility and sensitivity to context.

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Sammandrag

Den centrala frågeställningen i detta arbete gäller fiskerivetenskapens roll vid förvaltning av fiskresurser och styrning av fisket. En utgångspunkt är att både internationella och nationella målsättningar låter förstå att förvaltningen av fiskresurser strävar mot en hållbar utveckling. I verkligheten är många fiskerier långt från detta ideal. En del forskare hävdar att hållbar utveckling omöjligt kan uppnås eftersom okunskapen om fiskresurserna är stor samtidigt som ekonomiska intressen driver fram en hård exploatering av resurser. Bristen på kunskap betyder att förvaltningen inte hinner reglera utnyttjandet innan det är för sent. Vissa pessimister hävdar t.o.m. att de biologiska modeller som förvaltningen stöder sig på antingen saknar grund eller bygger på direkt felaktiga antaganden. Mer optimistiska forskare anser att den metodologiska utveckling som skett inom fiskerivetenskapen kan producera bättre råd för förvaltningen. Framför allt har forskningen utvecklat metodik som ger förvaltning och intressegrupper en möjlighet att bättre än hittills analysera risker vid förvaltningsbeslut t.ex. under förhandlingar om fiskekvoter.

I detta arbete har jag närmat mig argumenten kring förvaltningsproblemens natur genom att analysera var bristen på kunskap om fiskresurser och fiskerier uppstår. Bristen på kunskap kan gälla förändringar i fiskbestånden, t.ex. fiskynglens överlevnad, eller konsekvenser av förvaltningsbeslut för själva fisket. Nya begränsningar i fisket kan exempelvis ha överraskande effekter. För att kunna diskutera frågor kring kunskapsbristen i detalj har jag använt en kategorisering av bristen på kunskap. I detta arbete beskriver begreppet *risk* situationer då de möjliga konsekvenserna eller händelserna är kända samtidigt som deras sannolikhet kan bestämmas. Då *osäkerhet* råder är de möjliga händelserna eller konsekvenserna kända, men deras sannolikhet okänd. *Okunskap* betecknar situationer då man inte ens känner till de möjliga händelserna eller konsekvenserna. *Obestämdhet* förekommer då flera alternativa händelseförlopp är möjliga medan avgörandet mellan dem bestäms av framtida händelser.

All fiskeriförvaltning strävar till att styra utnyttjandet av ekologiska processer. Kunskapen om de ekologiska processerna i fiskbestånd och fisksamhällen är inte perfekt och därmed står förvaltningen inför risk, osäkerhet, okunskap eller obestämdhet då beslut skall fattas. Denna studie visar emellertid att perspektivet på fiskerier och fiskeriförvaltning blir alltför snävt om man enbart beaktar ekologiska frågeställningar. Förvaltningen av fiskresurser är en ekonomisk och socio-politisk verksamhet som kan betraktas som en form av samhällsplanering. Studien lyfter fram två uppfattningar om planering: den ena betraktar planeringen som en

serie gradvisa ingrepp i utvecklingen medan den andra utgår från klara helhetslösningar med stark betoning på övergripande mål. Den stegvisa planeringen betonar kommunikation mellan alla inblande parter, möjligheter till revision av målsättningen och en diskussion om både mål och medel. En diskussion om de medel som skall användas för att uppnå fastslagna mål är irrelevant då planeringen utgår från helhetslösningar. Skillnaden i synsätt kan återföras på tolkningen av planeringens rationalitet. Helhetsplanering bygger på en instrumentell målinriktad rationalitet som utgår från att beräkningar kan ge de riktiga svaren på alla relevanta frågor. Den stegvisa planeringen betonar kommunikationens roll: planering handlar om att välja, men valet är en överenskommelse, inte resultatet av en uträkning.

I min studie har jag lyft fram kopplingen mellan planeringssynen och bristen på ekologisk kunskap. Denna synvinkel lyfter fram nya typer av brist på kunskap, men också nya källor till information för fiskeriförvaltningen. Bristen på kunskap kan gälla t.ex. socio-ekonomiska förhållanden eller orsakssammanhang, men också frågor om själva förvaltningens målsättning. Det allmänna målet att uppnå en hållbar utveckling är innehållslöst utan en detaljerad analys av vad begreppet innebär för fiskresurserna, för dem som utnyttjar bestånden och för andra intresserade. Ifall fiskeriförvaltningen förnekar detta och ifall fiskeriforskningen låter bli att befatta sig med dessa aspekter av förvaltningsproblematiken kan beslutsfattare drivas in i situationer av okunskap, vid vilka beslut fattas på otillräckliga grunder. Det faktum att förvaltningen av fiskeresurser ofta råkar in i låsta situationer tyder på att fiskeriförvaltningen sett sin uppgift alltför snävt. Det leder till att samma argument upprepas från år till år, trots att alla parter finner situationen ohållbar. Följden är att många viktiga fiskresurser överexploateras konstant. I värsta fall sker en förbättring först då resursen kollapsar, t.ex. därför att fiskbeståndets förökning misslyckats under en följd av år. I en sådan situationen blir alla parter tvungna att tänka om och då kan helt nya förvaltningssystem införas.

Detta arbete ger en ny syn på fiskerivetenskapens roll och möjlighet att bidra till en hållbar utveckling av fiskerier genom att samtidigt analysera fiskeriförvaltning som planering och tillgången på kunskap om ekologiska processer och andra relevanta faktorer. Slutsatsen är att den frustration forskningen ofta känt inför överexploateringen av fiskresurser delvis beror på föreställningen att fiskeriförvaltning alltid bör förverkligas som helhetsplanering på basen av (begränsad) biologisk kunskap. Det är lätt att visa att okunskapen om många fiskbestånd och deras exploatering är så kompakt att inte ens de mest sofistikerade riskanalyser förmår ge fiskeriförvaltningen en tillräcklig grund för entydiga beslut som skulle garantera ett hållbart fiske. Det finns emellertid också exempel på fiskerier som är så enkla och välundersökta att en expertstyrd förvaltning kan fatta beslut utgående från en avvägning mellan olika risker. För att skapa förutsättningar för produktion av relevant fiskerivetenskaplig information bör man för varje fiskeri reda ut vilket slag av brist på kunskap som dominerar samt hur denna brist påverkar de beslut som skall och kan fattas. Ur förvaltningens synvinkel kan detaljkunskap om t.ex. fiskens tillväxt vara meningslös ifall hela fiskets struktur håller på att förändras.

Det är svårt att kategorisera alla olika typer av fiskerier för att entydigt bestämma vilken typ av fiskerivetenskapligt stöd förvaltningen behöver. En sådan kategorisering skulle i själva verket stå i konflikt med en av de centrala observationerna i detta arbete: helhetsplanering utgående från fastspikade målsättningar är möjlig endast i begränsad omfattning i fiskeriförvaltningen. Istället är det möjligt att ställa upp vissa kriterier som bör uppfyllas av det vetenskapliga stödet för förvaltningsbeslut. Kriterierna skall beakta möjligheterna att genomföra målinriktade beräkningar av händelser och konsekvenser och behovet av diskussioner mellan alla inblandade parter om alternativa åtgärder och beslut. Därigenom bör det vetenskapliga stödet för fiskeriförvaltning vara:

- övergripande (beaktar både ekologiska och socio-ekonomiska sidor av utnyttjandet av fiskresurser),
- fokuserat (skapar inte skeninformation genom att upprepa samma resultat i olika form),
- logiskt uppbyggt (slutsatser kan härledas från premisser, samma grundantaganden används genomgående),
- noggrant (data insamlas och behandlas i enlighet med utvärderad och accepterad metodik),
- praktiskt tillämpbart (de alternativ som framförs förankras i verkligheten),
- förståeligt (de som utnyttjar informationen skall ha en reell möjlighet att se på vilka grunder olika alternativ har jämförts), samt

– neutralt (alternativ skall analyseras och presenteras utan dold eller öppen politisk eller ideologisk vinkling för att kunna stöda en diskussion som också utgår från värderingar).

Dessa kriterier kan anpassas flexibelt till olika situationer och kan användas trots att fiskets variationsrikedom utesluter en entydig tolkning av begreppet hållbar utveckling. Forskningen kan emellertid skapa förutsättningar för en strävan mot hållbar utveckling på ett praktiskt plan genom att stöda de diskussioner som behövs. En förutsättning är att forskningen lyckas ge förvaltningen och andra intressegrupper en uppfattning om hur olika ekologiska och socio-ekonomiska förhållanden hänger ihop, samtidigt som den öppet redovisar för bristerna i vår kunskap om fiskerier.

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Appendix I

The ACFM reports used as a source of data.

ICES 1979. Reports of the ICES Advisory Committee on Fishery Management, 1978. *ICES Coop. Res. Rep.* 85, 157 pp.

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ICES 1992. Reports of the ICES Advisory Committee on Fishery Management 1991. *ICES Coop. Res. Rep.* 179, part 1, 368 pp., part 2, 72 pp.

ICES 1993. Reports of the ICES Advisory Committee on Fishery Management 1992. *ICES Coop. Res. Rep.* 193, part 1, 389 pp., part 2, 79 pp.

ICES 1994. Reports of the ICES Advisory Committee on Fishery Management 1993. *ICES Coop. Res. Rep.* 196, part 1, 374 pp., part 2, 102 pp.

ICES 1995. Reports of the ICES Advisory Committee on Fishery Management, 1994. *ICES Coop. Res. Rep.* 210, part 1, 312 pp., part 2, 222 pp.

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Appendix 2

Data used to analyse association between recommendations for exploitation (RECCAT), recruitment variability (RECVAR, RECRCLA), mean catch (MEANCA), unit price (UNITP, PRICCLA) and value of catch (VALCLA). Data on recommendations from ICES ACFM reports 1990-1996, recruitment variability from Myers et al. (1995a) and price information from FAO (1995b). Stock codes (abbreviated to eight characters) as in appendix 3.

STOCK	YEAR	RECCAT	RECVAR	MEANCA	UNITP	RECRCLA	VALCLA	PRICCLA
anglervi	1992	2	0.37	16.90	2.660	4	3	3
anglervi	1993	1	0.37	16.90	2.660	4	3	3
anglervi	1994	1	0.37	16.90	2.660	4	3	3
anglervi	1995	1	0.37	16.90	2.660	4	3	3
capba	1990	1	0.64	1052.00	0.074	4	3	1
capba	1992	1	0.64	1052.00	0.074	4	3	1
capba	1993	3	0.64	1052.00	0.074	4	3	1
capba	1994	3	0.64	1052.00	0.074	4	3	1
capba	1995	3	0.64	1052.00	0.074	4	3	1
capelvxi	1990	1	0.20	605.00	0.074	2	3	1
capelvxi	1992	1	0.20	605.00	0.074	2	3	1
capelvxi	1993	1	0.20	605.00	0.074	2	3	1
capelvxi	1994	1	0.20	605.00	0.074	2	3	1
capelvxi	1995	1	0.20	605.00	0.074	2	3	1
cod2224	1990	3	0.33	36.70	0.990	3	3	2
cod2224	1992	3	0.33	36.70	0.990	3	3	2
cod2224	1993	3	0.33	36.70	0.990	3	3	2
cod2224	1994	1	0.33	36.70	0.990	3	3	2
cod2224	1995	3	0.33	36.70	0.990	3	3	2
cod2532	1990	3	0.31	214.00	0.990	3	4	2
cod2532	1992	3	0.31	214.00	0.990	3	4	2
cod2532	1993	3	0.31	214.00	0.990	3	4	2
cod2532	1994	3	0.31	214.00	0.990	3	4	2
cod2532	1995	3	0.31	214.00	0.990	3	4	2
codfarvb	1990	3	0.25	24.70	0.990	2	3	2
codfarvb	1992	3	0.25	24.70	0.990	2	3	2
codfarvb	1993	3	0.25	24.70	0.990	2	3	2
codfarvb	1994	3	0.25	24.70	0.990	2	3	2
codfarvb	1995	3	0.25	24.70	0.990	2	3	2
codgreen	1990	1	.	110.00	0.990	.	4	2
codgreen	1994	3	.	110.00	0.990	.	4	2
codgreen	1995	3	.	110.00	0.990	.	4	2
codicelv	1990	1	.	.	.	.	.	.
codicelv	1992	3	0.16	393.00	0.990	1	4	2
codicelv	1993	3	0.16	393.00	0.990	1	4	2
codicelv	1994	3	0.16	393.00	0.990	1	4	2
codicelv	1995	3	0.16	393.00	0.990	1	4	2
codiiia	1990	2	0.26	19.60	0.990	2	3	2
codiiia	1992	3	0.26	19.60	0.990	2	3	2
codiiia	1993	3	0.26	19.60	0.990	2	3	2
codiv	1990	3	0.26	196.00	0.990	2	4	2
codiv	1991	3	0.26	196.00	0.990	2	4	2
codiv	1992	3	0.26	196.00	0.990	2	4	2
codiv	1993	3	0.26	196.00	0.990	2	4	2
codiv	1994	3	0.26	196.00	0.990	2	4	2

codiv	1995	3	0.26	196.00	0.990	2	4	2
codnearc	1990	1	0.33	677.00	0.990	3	4	2
codnearc	1992	1	0.33	677.00	0.990	3	4	2
codnearc	1993	1	0.33	677.00	0.990	3	4	2
codnearc	1994	1	0.33	677.00	0.990	3	4	2
codnearc	1995	1	0.33	677.00	0.990	3	4	2
codvia	1990	3	0.21	16.10	0.990	2	2	2
codvia	1991	3	0.21	16.10	0.990	2	2	2
codvia	1992	3	0.21	16.10	0.990	2	2	2
codvia	1993	3	0.21	16.10	0.990	2	2	2
codvia	1994	3	0.21	16.10	0.990	2	2	2
codvia	1995	3	0.21	16.10	0.990	2	2	2
codviia	1990	3	0.20	9.62	0.990	2	2	2
codviia	1991	3	0.20	9.62	0.990	2	2	2
codviia	1992	3	0.20	9.62	0.990	2	2	2
codviia	1993	3	0.20	9.62	0.990	2	2	2
codviia	1994	3	0.20	9.62	0.990	2	2	2
codviia	1995	3	0.20	9.62	0.990	2	2	2
codviid	1993	3	0.28	5.59	0.990	3	1	2
codviid	1994	3	0.28	5.59	0.990	3	1	2
codviid	1995	3	0.28	5.59	0.990	3	1	2
codviifv	1990	3	0.36	6.68	0.990	4	1	2
codviifv	1991	2	0.36	6.68	0.990	4	1	2
codviifv	1992	3	0.36	6.68	0.990	4	1	2
codviifv	1993	3	0.36	6.68	0.990	4	1	2
codviifv	1994	3	0.36	6.68	0.990	4	1	2
codviifv	1995	3	0.36	6.68	0.990	4	1	2
grhalsub	1990	3	0.08	28.10	2.660	1	3	3
grhalsub	1992	3	0.08	28.10	2.660	1	3	3
grhalsub	1993	2	0.08	28.10	2.660	1	3	3
grhalsub	1994	3	0.08	28.10	2.660	1	3	3
grhalsub	1995	3	0.08	28.10	2.660	1	3	3
grhalvxi	1990	2	0.09	32.80	2.660	1	4	3
grhalvxi	1992	1	0.09	32.80	2.660	1	4	3
grhalvxi	1993	1	0.09	32.80	2.660	1	4	3
grhalvxi	1994	3	0.09	32.80	2.660	1	4	3
grhalvxi	1995	3	0.09	32.80	2.660	1	4	3
hadfarvb	1990	1	0.50	16.20	0.990	4	2	2
hadfarvb	1991	1	0.50	16.20	0.990	4	2	2
hadfarvb	1992	2	0.50	16.20	0.990	4	2	2
hadfarvb	1993	3	0.50	16.20	0.990	4	2	2
hadfarvb	1994	3	0.50	16.20	0.990	4	2	2
hadfarvb	1995	1	0.50	16.20	0.990	4	2	2
hadiv	1990	3	0.49	300.00	0.990	4	4	2
hadiv	1991	3	0.49	300.00	0.990	4	4	2
hadiv	1992	3	0.49	300.00	0.990	4	4	2
hadiv	1993	3	0.49	300.00	0.990	4	4	2
hadiv	1994	3	0.49	300.00	0.990	4	4	2
hadiv	1995	2	0.49	300.00	0.990	4	4	2
hadnearc	1990	3	0.59	122.00	0.990	4	4	2
hadnearc	1992	1	0.59	122.00	0.990	4	4	2
hadnearc	1993	1	0.59	122.00	0.990	4	4	2
hadnearc	1994	1	0.59	122.00	0.990	4	4	2
hadnearc	1995	1	0.59	122.00	0.990	4	4	2
hadvia	1990	3	0.47	34.20	0.990	4	3	2
hadvia	1991	3	0.47	34.20	0.990	4	3	2
hadvia	1992	3	0.47	34.20	0.990	4	3	2

hadvia	1993	3	0.47	34.20	0.990	4	3	2
hadvia	1994	3	0.47	34.20	0.990	4	3	2
hadvia	1995	3	0.47	34.20	0.990	4	3	2
hadvib	1990	1	0.48	6.30	0.990	4	1	2
hadvib	1991	1	0.48	6.30	0.990	4	1	2
hadvib	1992	3	0.48	6.30	0.990	4	1	2
hadvib	1994	1	0.48	6.30	0.990	4	1	2
hadvib	1995	1	0.48	6.30	0.990	4	1	2
hakeiiia	1990	1	0.10	57.30	0.990	1	3	2
hakeiiia	1992	2	0.10	57.30	0.990	1	3	2
hakeiiia	1993	3	0.10	57.30	0.990	1	3	2
hakeiiia	1994	3	0.10	57.30	0.990	1	3	2
hakeiiia	1995	3	0.10	57.30	0.990	1	3	2
hakeviii	1990	2	0.42	15.30	0.990	4	2	2
hakeviii	1992	3	0.42	15.30	0.990	4	2	2
hakeviii	1993	3	0.42	15.30	0.990	4	2	2
hakeviii	1994	3	0.42	15.30	0.990	4	2	2
hakeviii	1995	3	0.42	15.30	0.990	4	2	2
her2529	1990	1	0.16	281.00	0.230	1	3	1
her2529	1991	1	0.16	281.00	0.230	1	3	1
her2529	1992	1	0.16	281.00	0.230	1	3	1
her2529	1993	1	0.16	281.00	0.230	1	3	1
her2529	1994	1	0.16	281.00	0.230	1	3	1
her2529	1995	1	0.16	281.00	0.230	1	3	1
her30	1990	1	0.29	25.90	0.230	3	1	1
her30	1991	1	0.29	25.90	0.230	3	1	1
her30	1992	1	0.29	25.90	0.230	3	1	1
her30	1993	1	0.29	25.90	0.230	3	1	1
her30	1994	1	0.29	25.90	0.230	3	1	1
her30	1995	1	0.29	25.90	0.230	3	1	1
her31	1990	1	0.28	7.50	0.230	3	1	1
her31	1991	1	0.28	7.50	0.230	3	1	1
her31	1992	1	0.28	7.50	0.230	3	1	1
her31	1993	1	0.28	7.50	0.230	3	1	1
her31	1994	1	0.28	7.50	0.230	3	1	1
her31	1995	1	0.28	7.50	0.230	3	1	1
hercelvi	1990	3	.	21.10	4.224	.	3	3
hercelvi	1993	1	.	21.10	4.224	.	3	3
hercelvi	1994	1	.	21.10	4.224	.	3	3
hercelvi	1995	3	.	21.10	4.224	.	3	3
hericelv	1990	1	0.40	49.60	0.230	4	2	1
hericelv	1991	1	0.40	49.60	0.230	4	2	1
hericelv	1992	1	0.40	49.60	0.230	4	2	1
hericelv	1993	1	0.40	49.60	0.230	4	2	1
hericelv	1994	1	0.40	49.60	0.230	4	2	1
hericelv	1995	1	0.40	49.60	0.230	4	2	1
heriviii	1990	2	0.38	565.00	0.230	4	4	1
heriviii	1991	1	0.38	565.00	0.230	4	4	1
heriviii	1992	2	0.38	565.00	0.230	4	4	1
heriviii	1993	1	0.38	565.00	0.230	4	4	1
heriviii	1994	2	0.38	565.00	0.230	4	4	1
heriviii	1995	3	0.38	565.00	0.230	4	4	1
hernospr	1990	3	0.82	571.00	0.230	4	4	1
hernospr	1991	3	0.82	571.00	0.230	4	4	1
hernospr	1992	1	0.82	571.00	0.230	4	4	1
hernospr	1993	1	0.82	571.00	0.230	4	4	1
hernospr	1994	1	0.82	571.00	0.230	4	4	1

hemospr	1995	1	0.82	571.00	0.230	4	4	1
herviaC	1990	1	.	5.50	0.230	.	1	1
herviaC	1991	1	.	5.50	0.230	.	1	1
herviaC	1992	3	.	5.50	0.230	.	1	1
herviaC	1994	3	.	5.50	0.230	.	1	1
hervian	1990	1	0.30	56.20	0.230	3	2	1
hervian	1991	1	0.30	56.20	0.230	3	2	1
hervian	1992	1	0.30	56.20	0.230	3	2	1
hervian	1994	1	0.30	56.20	0.230	3	2	1
mackevii	1995	1	.	53.60	0.300	.	2	1
mackewes	1990	3	0.29	680.00	0.300	3	4	1
mackewes	1993	1	0.29	680.00	0.300	3	4	1
mackewes	1994	3	0.29	680.00	0.300	3	4	1
mackewes	1995	3	0.29	680.00	0.300	3	4	1
megrviib	1992	1	0.12	18.70	0.990	1	2	2
megrviib	1993	1	0.12	18.70	0.990	1	2	2
megrviib	1994	1	0.12	18.70	0.990	1	2	2
megrviib	1995	1	0.12	18.70	0.990	1	2	2
megrviic	1990	1	.	.	.	.	.	.
megrviic	1992	1	0.45	1.80	0.990	4	1	2
megrviic	1993	1	0.45	1.80	0.990	4	1	2
megrviic	1994	2	0.45	1.80	0.990	4	1	2
norpouti	1993	1	0.31	297.00	0.074	3	3	1
norpouti	1994	1	0.31	297.00	0.074	3	3	1
norpouti	1995	1	0.31	297.00	0.074	3	3	1
plaiceiv	1990	1	0.18	122.00	2.660	1	4	3
plaiceiv	1991	2	0.18	122.00	2.660	1	4	3
plaiceiv	1992	1	0.18	122.00	2.660	1	4	3
plaiceiv	1993	1	0.18	122.00	2.660	1	4	3
plaiceiv	1994	3	0.18	122.00	2.660	1	4	3
plaiceiv	1995	3	0.18	122.00	2.660	1	4	3
plaicevi	1990	3	0.15	3.90	2.660	1	2	3
plaicevi	1991	2	0.15	3.90	2.660	1	2	3
plaicevi	1992	2	0.15	3.90	2.660	1	2	3
plaicevi	1993	2	0.15	3.90	2.660	1	2	3
plaicevi	1994	2	0.15	3.90	2.660	1	2	3
plaicevi	1995	1	0.15	3.90	2.660	1	2	3
plaicevi	1990	2	0.15	6.50	2.660	1	2	3
plaicevi	1991	3	0.15	6.50	2.660	1	2	3
plaicevi	1992	1	0.15	6.50	2.660	1	2	3
plaicevi	1993	1	0.15	6.50	2.660	1	2	3
plaicevi	1994	1	0.15	6.50	2.660	1	2	3
plaicevi	1995	1	0.15	6.50	2.660	1	2	3
plaicevi	1992	2	0.22	1.53	2.660	2	1	3
plaicevi	1993	1	0.22	1.53	2.660	2	1	3
plaicevi	1994	2	0.22	1.53	2.660	2	1	3
plaicevi	1995	3	0.22	1.53	2.660	2	1	3
plaicevi	1990	1	0.23	1.42	2.660	2	1	3
plaicevi	1991	1	0.23	1.42	2.660	2	1	3
plaicevi	1992	1	0.23	1.42	2.660	2	1	3
plaicevi	1993	1	0.23	1.42	2.660	2	1	3
plaicevi	1994	1	0.23	1.42	2.660	2	1	3
plaicevi	1995	2	0.23	1.42	2.660	2	1	3
saitfarv	1990	3	0.21	33.50	0.990	2	3	2
saitfarv	1992	3	0.21	33.50	0.990	2	3	2
saitfarv	1993	3	0.21	33.50	0.990	2	3	2
saitfarv	1994	3	0.21	33.50	0.990	2	3	2

saitfarv	1995	1	0.21	33.50	0.990	2	3	2
saithvi	1995	1	0.12	23.10	0.990	1	3	2
saiticel	1990	2	0.21	76.40	0.990	2	3	2
saiticel	1992	1	0.21	76.40	0.990	2	3	2
saiticel	1993	1	0.21	76.40	0.990	2	3	2
saiticel	1994	1	0.21	76.40	0.990	2	3	2
saiticel	1995	1	0.21	76.40	0.990	2	3	2
saitivii	1995	1	0.24	169.00	0.990	2	4	2
saitnear	1990	1	0.20	161.00	0.990	2	4	2
saitnear	1992	1	0.20	161.00	0.990	2	4	2
saitnear	1993	1	0.20	161.00	0.990	2	4	2
saitnear	1994	1	0.20	161.00	0.990	2	4	2
saitnear	1995	1	0.20	161.00	0.990	2	4	2
salmbal	1992	3	.	3.00	2.900	.	1	3
salmbal	1993	3	.	3.00	2.900	.	1	3
salmbal	1994	3	.	3.00	2.900	.	1	3
salmbal	1995	3	.	3.00	2.900	.	1	3
salmgof	1990	2	.	0.20	2.900	.	1	3
salmgof	1991	2	.	0.20	2.900	.	1	3
salmgof	1992	3	.	0.20	2.900	.	1	3
salmgof	1993	3	.	0.20	2.900	.	1	3
salmgof	1994	3	.	0.20	2.900	.	1	3
salmgof	1995	3	.	0.20	2.900	.	1	3
sandeeli	1990	1	0.30	717.00	0.074	3	3	1
sandeeli	1993	1	0.30	717.00	0.074	3	3	1
sandeeli	1994	1	0.30	717.00	0.074	3	3	1
sandeeli	1995	1	0.30	717.00	0.074	3	3	1
sandeels	1990	1	0.34	15.10	0.074	3	1	1
sandeels	1991	1	0.34	15.10	0.074	3	1	1
sandeels	1992	3	0.34	15.10	0.074	3	1	1
sandeels	1993	1	0.34	15.10	0.074	3	1	1
sandeels	1994	1	0.34	15.10	0.074	3	1	1
sardviii	1990	1	0.19	162.00	0.230	1	3	1
sardviii	1995	3	0.19	162.00	0.230	1	3	1
sebmarvv	1995	3	0.48	76.60	2.240	4	4	2
sebmenne	1990	2	0.28	59.90	2.240	3	4	2
sebmenne	1992	1	0.28	59.90	2.240	3	4	2
sebmenne	1993	1	0.28	59.90	2.240	3	4	2
sebmenne	1994	3	0.28	59.90	2.240	3	4	2
sebmenne	1995	3	0.28	59.90	2.240	3	4	2
sebmenva	1995	2	.	46.70	2.240	.	4	2
soleiv	1990	3	0.35	22.70	2.660	3	3	3
soleiv	1991	2	0.35	22.70	2.660	3	3	3
soleiv	1992	1	0.35	22.70	2.660	3	3	3
soleiv	1993	1	0.35	22.70	2.660	3	3	3
soleiv	1994	1	0.35	22.70	2.660	3	3	3
soleiv	1995	3	0.35	22.70	2.660	3	3	3
soleviia	1990	2	0.23	1.52	2.660	2	1	3
soleviia	1991	1	0.23	1.52	2.660	2	1	3
soleviia	1992	2	0.23	1.52	2.660	2	1	3
soleviia	1993	1	0.23	1.52	2.660	2	1	3
soleviia	1994	3	0.23	1.52	2.660	2	1	3
soleviia	1995	2	0.23	1.52	2.660	2	1	3
soleviid	1990	1	0.30	4.10	2.660	3	2	3
soleviid	1991	3	0.30	4.10	2.660	3	2	3
soleviid	1992	3	0.30	4.10	2.660	3	2	3
soleviid	1993	2	0.30	4.10	2.660	3	2	3

soleviid	1994	1	0.30	4.10	2.660	3	2	3
soleviid	1995	1	0.30	4.10	2.660	3	2	3
soleviie	1990	3	0.20	0.90	2.660	2	1	3
soleviie	1992	3	0.20	0.90	2.660	2	1	3
soleviie	1993	1	0.20	0.90	2.660	2	1	3
soleviie	1994	1	0.20	0.90	2.660	2	1	3
soleviie	1995	2	0.20	0.90	2.660	2	1	3
soleviif	1990	1	0.25	1.18	2.660	2	1	3
soleviif	1991	1	0.25	1.18	2.660	2	1	3
soleviif	1992	1	0.25	1.18	2.660	2	1	3
soleviif	1993	1	0.25	1.18	2.660	2	1	3
soleviif	1994	1	0.25	1.18	2.660	2	1	3
soleviif	1995	2	0.25	1.18	2.660	2	1	3
soleviii	1992	1	0.07	5.25	2.660	1	2	3
soleviii	1993	1	0.07	5.25	2.660	1	2	3
soleviii	1994	1	0.07	5.25	2.660	1	2	3
soleviii	1995	1	0.07	5.25	2.660	1	2	3
soleviii	1990	1	.	.	.	.	.	.
soleviii	1991	1	0.07	5.25	2.660	1	2	3
sprat223	1990	1	0.36	119.00	0.074	4	1	1
sprat223	1991	1	0.36	119.00	0.074	4	1	1
sprat223	1992	1	0.36	119.00	0.074	4	1	1
sprat223	1993	1	0.36	119.00	0.074	4	1	1
sprat223	1994	1	0.36	119.00	0.074	4	1	1
sprat223	1995	1	0.36	119.00	0.074	4	1	1
whitiv	1990	3	0.23	150.00	0.990	2	4	2
whitiv	1991	3	0.23	150.00	0.990	2	4	2
whitiv	1992	3	0.23	150.00	0.990	2	4	2
whitiv	1993	3	0.23	150.00	0.990	2	4	2
whitiv	1994	3	0.23	150.00	0.990	2	4	2
whitiv	1995	2	0.23	150.00	0.990	2	4	2
whitiv	1995	2	0.25	6.50	0.990	2	1	2
whitvia	1990	3	0.29	13.90	0.990	3	2	2
whitvia	1991	3	0.29	13.90	0.990	3	2	2
whitvia	1992	3	0.29	13.90	0.990	3	2	2
whitvia	1993	3	0.29	13.90	0.990	3	2	2
whitvia	1994	3	0.29	13.90	0.990	3	2	2
whitvia	1995	3	0.29	13.90	0.990	3	2	2
whitviia	1990	3	0.13	11.10	0.990	1	2	2
whitviia	1992	3	0.13	11.10	0.990	1	2	2
whitviia	1993	1	0.13	11.10	0.990	1	2	2
whitviia	1994	1	0.13	11.10	0.990	1	2	2
whitviia	1995	1	0.13	11.10	0.990	1	2	2
whitviif	1990	1	0.27	9.64	0.990	2	2	2
whitviif	1991	1	0.27	9.64	0.990	2	2	2
whitviif	1992	2	0.27	9.64	0.990	2	2	2
whitviif	1993	3	0.27	9.64	0.990	2	2	2
whitviif	1994	3	0.27	9.64	0.990	2	2	2
whitviif	1995	3	0.27	9.64	0.990	2	2	2

Appendix 3

Stocks used in the analysis of recommendations and their codes in systematic order.

CLUPEIFORMES

		Code
Clupeidae		
Clupea harengus (Herring)		
	Baltic Sea, Subdivisions 25-29,32 and Gulf of Riga	her2529
	Bothnian Sea, Subdivision 30	her30
	Bothnian Bay, Subdivision 31	her31
	Celtic Sea and Division VIIj	hercelviij
	Icelandic summer spawning herring	hericelva
	North Sea herring	heriviiia
	Norwegian spring spawning herring	hernosprsp
	Clyde herring	herviaC
	Division VIa North	hervian
Sardinia pilchardus (Sardine)		sardviiicixa
Sprattus sprattus (Sprat)		
	Baltic Sea, Subdivisions 22-32	sprat2232

GADIFORMES

Gadidae		
Gadus morhua (Cod)		
	Baltic Sea, Subdivisions 22-24	cod2224
	Baltic Sea, Subdivisions 25-32	cod2532
	Faroe plateau, Subdivision Vb1	codfarvb1
	East Greenland cod	codgreen1
	Icelandic cod	codicelva
	Division IIIa	codiiia
	North Sea cod	codiv
	North-East Arctic cod	codneare
	Division VIa (West of Scotland)	codvia
	Irish Sea	codviia
	English Channel	codviid
	Celtic Sea	codviifviigviih
Melanogrammus aeglefinus (Haddock)		
	Faroe haddock, Division Vb	hadfarvb
	North Sea haddock	hadiv
	North-East Arctic haddock	hadnearc
	Division VIa (West of Scotland)	hadvia
	Division Vb (Rockall)	hadvib
Merlangius merlangus (Whiting)		
	North Sea whiting	whitiv
	Division VIa (West of Scotland)	whitvia
	Irish Sea	whitviia
	Celtic Sea	whitviifviigviih

Pollachius virens (Saithe)	
Faroe saithe, Division Vb	saitfarvb
Division VI	saithvi
Icelandic saithe, Division Va	saiticelva
North Sea	saitiviiia
Nort-East Arctic saithe	saitnearc
Trisopterus esmarkii (Norway pout)	
North Sea	norpoutiv
Merluccius merluccius (Hake)	
Northern stock	hakeiiiaivviviiviiiabN
Southern stock	hakeviiicixa
LOPHIIFORMES	
Lophiidae	
Lophius piscatorius (Anglerfish/Monkfish)	anglerviib-kviia-b
PERCIFORMES	
Ammodytidae	
Ammodytes marinus (Sandeel)	
North Sea	sandeeliv
Shetland Islands	sandeelshet
Scombridae	
Scomber scombrus (Mackerel)	
Western mackerel	mackewes
Division VIIIc, IXa	mackeviiicixas
PLEURONECTIFORMES	
Pleuronectidae	
Pleuronectes platessa (Plaice)	
North Sea	plaiceiv
Irish Sea	plaiceviia
Division VIId	plaiceviid
Division VIIe	plaiceviie
Celtic Sea	plaiceviifviig
Reinhardtius hippoglossoides (Greenland halibut)	
Sub-areas I, II	grhalsub12
Sub-areas V, XIV	grhalvxiv
Scophthalmidae	
Lepidorhombus whiffiagonis (Megrim)	
Divisions VIIb,c,e-k,VIIIa,b	megrviib,c,e-kviiia,b
Divisions VIIIc,IXa	megrviiicixa
Soleidae	
Solea vulgaris (Sole)	
North Sea	soleiv
Irish Sea	soleviia
Division VIId	soleviid
Division VIIe	soleviie
Celtic Sea	soleviifviig
Bay of Biscay	soleviiia,b

SALMONIFORMES

Osmeridae

Mallotus villotus (Capelin)

Barents Sea	capba
Sub-areas V, XIV, Division IIa W of 5°W	capelvxiv

Salmonidae

Salmo salar (Salmon)

Baltic main basin and Gulf of Bothnia	salmbal
Gulf of Finland	salmgof

SCORPAENIFORMES

Scorpaenidae

Sebastes marinus (Redfish)

North-East Arctic	sebmarnear
Sub-areas V, XIV	sebmarvvixiv

*Sebastes m*entella (Redfish)

North-East Arctic	sebmennea
Sub-areas V, XIV	sebmenvaxiixiv

Appendix 4

Errata in the individual contributions and additional technical clarifications

Paper I

p. 124 column 2, Equation 8: N_t should be N_T
p. 125 column 1, line 1 N_t should be N_T
p. 125 column 1, line 3 F_t should be F_T
p. 125 column 1, line 5:
$N'_T = C_T(Z_T+M_T)/\{F_T[1-\exp(-Z_T)]\}$, should be:
$N'_T = C_T(Z_T+\Delta M_T)/\{F_T[1-\exp(-Z_T-\Delta M_T)]\}$
p. 125 column 1, line 9 from bottom:
will increase the relative error, should be:
will increase the absolute error
p. 125, column 2, Table 1 In all column headings M should be ΔM
p. 126, line 3:
...the relative error will increase. should be:
...the absolute value of the relative error will increase.
p. 126, column 1, line 7 from bottom N_t/N_t should be $\Delta N_t/N_t$
p. 127:
Additional information for Table 2: M=0.3, C has been scaled out by expressing $S(C_t)$ relative to the catch (see equation 17). This is possible as the catch is fixed (given). All subscripts can be replaced with T-1. Note the tendency to conserve Z_{T-1}.

F_{T-1}	F_{T-1}' (ΔM=+.2)	F_{T-1}' (ΔM=-.2)
0.1	0.084	0.121
0.2	0.169	0.240
0.3	0.257	0.357
0.4	0.346	0.472
0.5	0.437	0.586
0.6	0.529	0.698
0.7	0.622	0.809
0.8	0.717	0.919
0.9	0.812	1.028
1.0	0.908	1.137
1.1	1.005	1.244
1.2	1.103	1.351
1.3	1.202	1.458
1.4	1.301	1.563
1.5	1.400	1.669

Paper II

p. 91, column 2, 4 lines above heading "Results"
...an even distribution..., should be: ...a uniform distribution...
p. 96, error in alphabetical order of reference list: Finn et al. (1986) should follow Gulland (1983).

Paper III

p. 40, Equation (3) should be:

$$V(k,R(0)) = \sum_{j=0}^{k} \sigma^{j+1} H(j)$$

p. 40, line 13 k = 1,2..., should be: k=0,1,2,...
p. 44, Equation should be corrected as equation (3) to

$$V(\infty,R(0)) = \sum_{j=0}^{\infty} \sigma^{j+1} H(j)$$

Paper IV

Heading: Short Team TAC Advice ..., should be:
Short Term TAC Advice
p. 54, Fig. 1, figure caption: missing specification of axis: SSB = Spawning Stock Biomass
p. 55, Table 1,
table caption. BIO=Spawning Stock Biomass; add missing additional explanation: (=SSB in Figures).
Column heading R^2 (regression coefficient) should not be confused with R = recruitment in models used.
p. 55, Fig. 2,
specification of x-axis SBB should be SSB
Figure caption, line 2: stock biomass and recruitment...
should read: stock biomass...
p. 56, Fig. 3,
errors as for Fig. 2
p. 57, Fig. 4,
Figure caption, line 2: stock biomass and recruitment...
should be: stock biomass...
p. 58, Fig. 5,
errors as for Fig. 4
p. 60, line 1: In the approach outline above..., should be:
In the approach outlined above...

Paper V

p. 65, line 6: Management, 1980, should be Management, 1990

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Book chapters:

Tamminen P. & Starr M. 1990. A survey of forest soil properties related to soil acidification in Southern Finland. In: Kauppi P., Anttila P. & Kenttämies K. (eds.), *Acidification in Finland*, Springer, Berlin, pp. 237-251.

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