The knowledge base for fisheries management in developing countries - alternative approaches and methods

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1 The context of fisheries research knowledge - how do we identify the relevance of methods ?

Fisheries management has as any other institution a normative, a cognitive and a regulatory aspect which are closely interlinked.

The subject for this paper is the cognitive basis for fisheries management in developing countries and specifically that part of the cognitive basis which relates to the functioning of the resource system. We will discuss various options from a technical point of view. However, the technical options are endless and only a small subset will be relevant in any specific situation dependent on the management context rather than the technicalities.

The cognitive base is closely associated with the normative base for management insofar as any technical or informal evaluation of the state of stocks and management options is based on explicit or implicit management objectives and will relate to a specific set of 'managers' who is to take note of the evaluation and implement management - wether this is a central government, a formal co-management committee or communities implementing informal access rules which may even not be understood as fisheries management by the communities in the first place. The character and relevance of biological knowledge for management is therefore constituted by the objectives for management and the identity of the 'managers'. And the explicit or - in most cases - implicit objectives for management are largely defined by the identity of the managers.

Efforts to develop fisheries management in developing countries has predominantly been based on what could be termed the 'modern fisheries management model', that is a model based on a concept of rationality in which mandated research within specialised institutions produce formalised knowledge which is then used as a basis for management decisions and implementation by a centralised bureaucracy in interaction with representative democratic institutions. The management objectives in this model are in many cases not explicit, but the long term sustainability of the resource base has been the overriding objective whenever objectives are stated. The implementation of management within this rationality is entirely linked to an assumption of predictability, that is an understanding that specific and predictable targets can be achieved by implementing specific regulatory measures such as catch or effort quotas or technical measures. This normative and regulatory context means that the production of biological knowledge of the stock dynamics and predictions of the response of stocks to fishing has been the dominating form of mandated science within this model. Development efforts based on the modern fisheries management model have thus emphasised the development of research institutions which can produce this kind of knowledge.

This has been done to the extent that the research institution model - including the associated relevance criteria for knowledge - has been promoted as an end in its own right, as something which is considered an essential component of any fisheries management system irrespective of normative or regulatory context. The result has been failures and frustrations when it is realised that research institutions, which have received massive support to capacity building, even when they perform as expected, still fail to produce knowledge which is used or is usable in the local management context. It is even worse when support to this specific type of knowledge production creates a snowball process where the research

development becomes the nucleus for the development of a complete management model based on the modern model in a situation where this model is entirely inappropriate.

These experiences have demonstrated that the modern fisheries management model and its associated knowledge production setup is at best only relevant in those cases in developing countries where industrialised fisheries under national control are operated by western type business organisations and where a government structure with the political and technical capacity to develop and implement governance systems based on modern rationality is in place. This is not the typical situation in developing countries and the cases of even moderately successful export of the modern fisheries management model are therefore few. The best example in a Norwegian (and even global) context is the industrialised fisheries of Namibia. Another candidate could be the industrialised shallow water shrimp fisheries of Mozambique, but this case has also demonstrated some serious shortcomings of the model and could just as well be used as an example to demonstrate the need to develop a knowledge model which is more adapted to the specific local context.

It is therefore not possible to identify appropriate methods for fisheries assessments for developing countries separate from the local institutional setup of management. This also means that generalisations only can be made across similar management models.

In order to discuss and generalise about the appropriateness of methods for fisheries assessments in developing countries some assumptions must be made about the normative and regulatory context. The assumption is that the typical pattern of the fisheries development context is that

- The management institutions are fragile - both for research and management implementation, the institutions for formal fisheries decisions are fragile or non-existing.

- The Fisheries are mainly coastal, non-industrialised, with widely spread coastal population

- The resource base is a composite of many stocks which are harvested simultaneously

These assumptions point to management models which must be based on extensive user participation to be effective. Quantitative predictability based on institutionalised research is not a feasible option both for cost reasons (as a result of the combination of fragile institutions and the complexity of the fisheries and the resource base) and for reasons of acceptability among participants.

A minimum set of criteria for fisheries assessment methodologies in a development context is that the knowledge base for fisheries must be valid for all stake holders and it must be feasible to produce this knowledge on a sustained basis within the economic means of the society.

This set of criteria will exclude most methodologies presently applied in the North Atlantic or elsewhere in relation to industrialised fisheries in industrialised countries.

There is a need to identify and develop methodologies which rely on observations which can be made at low cost and which reflect resource system features which can be recognised and accepted by fishermen and researchers alike. This points to using indicators rather than complex assessments as produced by full models of processes. The problem is then to develop an appropriate conceptual framework for such indicators and to identify and develop indicator candidates. This work has hardly begun.

2 Fisheries sustainability indicators

2.1 The development of the concept of indicators

The concept of indicators in relation to fisheries sustainability has taken place within two different agendas.

One agenda is concerned about establishing indicators which can be used to govern policies in the international domain, in relation to sustainable development and in relation to market regulations. This development is promoted by international organisations and NGO's and centres around the Indicators of Sustainable Development (ISD) initiative of the UN Commission on Sustainable Development (CSD) (CSD 2001), which is a body assigned to follow up on the UNCED agenda 21. OECD has likewise developed an indicator framework for environmental performance reviews (OECD 1993). This agenda has been developed in relation to environmental sustainability in general but is also reflected in fisheries. An account of this development in relation to fisheries has been presented by Dahl (2000) and Garcia and Staples (2000).

Another agenda is rooted in the scientific community, where an increasing number of those participating in the ongoing production of stock assessments and fisheries management advisory services realise that they have encountered the complexity wall: The combination of extensive overcapacity in fishing fleets and quota based management systems has in practice led to a requirement for the biologists to produce predictions of consequences of various fisheries management options which are far more precise than what is technically possible within existing economic resources for research. The widening of the political scope of sustainability of fisheries requires biological fisheries advice to address issues far beyond the traditional single stock domain including the hatching success of sea birds, the abundance of benthic invertebrates or the genetic integrity of anadromous species complexes. A response to the complexity wall has been explorations into an identification of proxies to the standard reference points of stock assessments and indicators which are assumed to capture the effects of fisheries pressures on the ecosystem. The investigation of indicators has especially been pertinent in relation to the wider ecosystem effects of fisheries, which has emerged much more recently as a research area compared to classical fisheries biology and where an approach to develop full fledged functional models as an extension of classical approaches seems impossible from the outset. Reviews of this work were presented at the ICES/SCOR Symposium on Ecosystem Effects of Fishing (ICES Journal of Marine Science 57 (3): 465-791. 2000).

There has been some convergence between the two agendas and the international policy agenda should in principle be an extension of and build on the research agenda. The gap is however still very wide as indicated by the fact that the MSY concept, which now largely has been abandoned as a relevant and measurable reference point among fisheries biologists, is the only fisheries related indicator on the CSD list of indicator candidates (Commission on Sustainable Development 2001). The US National Marine Fisheries Service has probably presented the most ambitious attempt to date to integrate scientific state of the art into a management framework in relation to ecosystem issues and the use of indicators. This was presented in a report to Congress 1999 (National Marine Fisheries Service 1999).

The importance of acceptance by stake holders or even participation is alluded to within most of the literature dealing with environmental or fisheries indicators in relation to management. This is however generally stated as an important issue without further consideration of the implications. Acceptance is dealt with as if it was a trivial add-on without implications for other parts of the management setup or the relevance of indicators. Acceptance is however not a trivial issue as should be well known to any

fisheries scientist who has been confronted with fishermen disagreeing strongly with their latest stock assessments - and in more cases than not such disagreements are not just a pretext to resist quota reductions but are based on a genuine sense by fishermen that they have observed a different reality than what is reflected in the assessment. Such disagreements on the actual state of the stock and the fisheries can be explained by many factors at play simultaneously including technical - that observations are made on different scales and with different sampling pattern - and social and cultural differences between scientists and fishermen and the institutions they operate in.

Studies of acceptance of the knowledge base for fisheries management are now forthcoming (see for instance Neis and Felt (eds) 2000). A study of the decision making process in the Northeastern US fisheries management demonstrated that the requirements for legal accountability may lead to a situation where assessments which were seriously questioned by the fishing industry and about which the participating biologists had serious doubts were preferred over alternatives because they possessed the ir rigour and tractability required by law (Wilson pers com).

These studies may illuminate a basic problem in using wide stake holder acceptance as a criterion for the validity of indicators to be used in fisheries management in a modern context: the rationality of modern fisheries management requires the cognitive basis for management decisions to be firmly rooted in what is considered to be scientific objectivity as the first priority. This requirement will in most cases result in assessments which technically are beyond grasp or which may even be counterintuitive to anybody beyond the small community of professional fisheries biologists. This problem is of course not specific to fisheries but is just another variant of the basic democratic problem of the alienation of the citizen vis a vis the cognitive basis for most decisions relating to our interaction with our physical and biological environment.

The need to standardise and to meet legal requirements will be very dependent on the specific management setting. In a development context these requirements are presently mainly present in relation to industrialised fisheries. However, the internationalisation of trade with fisheries products and the associated requirements for accountability and tractability will put increasing pressure on some coastal semi industrial and even artisanal fisheries. The resistance of some developing countries regarding proposals for green labelling is rooted in the fear that formal stock assessments rooted in mainstream science with all that goes with it in terms of costs is an illustration of this problem.

The issue of local acceptance has in a fisheries development context been equated to a need to revert to or revitalise what has been termed traditional management systems based in indigenous ecological knowledge (Johannes). The problems involved in the concept of traditional management and revitalisation of such systems are not the subject of this paper, but indigenous ecological knowledge is an important component in the identification of indicators which are locally meaningful (McGoodwin et al 2000).

2.2 The scope of fisheries sustainability indicators

Indicators are commonly defined and classified according to OECD’s Pressure-State-Response framework of environmental indicators (OECD 1993). This approach implies some causality between the pressures on the ecosystem exerted by society through loading or removal, the state of the ecosystem and the societal responses to the changes in the state - the OECD framework does not consider the response of the ecosystem to pressure as separate from the state.

This distinction between indicators may have some intuitive appeal, but it does only make sense in within the context of underlying assumptions of causalities. These assumptions may not be made explicit in the specific implementation in which case lists of indicators and monitoring programmes may be pointless in the end. It is in the case of fisheries management in the Northen Atlantic only within the last decade that a somehow consistent set of PSR indicators has been developed with an identification of pressure indicators with reference points (fishing mortality with limit and precautionary reference points), closely associated state reference points (spawning stock biomass with limit and precautionary reference points) and response indicators (action relative to harvest control rules which are based on the pressure and state indicators).

The increasing emphasis on the need to consider fisheries sustainability in relation to not only the stock but the entire ecosystem has necessitated a corresponding extension of the scope of indicators from the single stock being the unit and yield sustainability being the main concern to the ecosystem being the unit and the maintenance of system integrity being the main management concern.

A number of single stock fisheries indicators based on analytical assessments have been identified and estimation methodologies and reference points have been developed. This development has at the same time brought serious problems with these approaches to the surface, which are still not resolved. The work to identify indicators, methodologies and reference points on ecosystem level has in comparison hardly begun. Indicators on ecosystem level can presently not be based on a full understanding and monitoring of the underlying processes. It is realised that ecosystem indicators must be meta indicators which summarise the outcome of many and complex underlying processes which may not be understood in detail.

The National Marine Fisheries Service study of ecosystem based fisheries management identified a number of principles, goals and policies to guide such ecosystem based fisheries management (see text box).

BASIC ECOSYSTEM PRINCIPLES, GOALS AND POLICIES (NMFS 1999)

Based on the Panel's experience and review of the fisheries ecosystem literature, we suggest that the following Principles, Goals and Policies embody key elements for ecosystem-based management of fisheries.

Principles

- The ability to predict ecosystem behavior is limited.
- Ecosystems have real thresholds and limits which, when exceeded, can effect major system restructuring.
- Once thresholds and limits have been exceeded, changes can be irreversible.
- Diversity is important to ecosystem functioning.
- Multiple scales interact within and among ecosystems.
- Components of ecosystems are linked.
- Ecosystem boundaries are open.
- Ecosystems change with time.

Goals

- Maintain ecosystem health and sustainability.

Policies

- Change the burden of proof.
- Apply the precautionary approach.
- Purchase "insurance" against unforeseen, adverse ecosystem impacts.
- Learn from management experiences.
- Make local incentives compatible with global goals.
- Promote participation, fairness and equity in policy and management.

The NMFS document presents a comprehensive discussion of the issues involved in the inclusion of ecosystem aspects in fisheries management, but it is at the same time clear that there still is along way to go before operational indicators have been identified. The principles refer to thresholds and limits, but this is used in a generic sense without reference to specific indicators and such indicators are not identified.

The question of scale of indicators is not trivial, especially not in relation to acceptability. The indicators should reflect the scale of the management unit, but stake holders' observations and understanding of the resource system may reflect a very different - more local - scale.

2.3 Criteria for indicator validity

A range of sets of criteria for indicators have been defined including the OECD (1993) basis for environmental performance reviews and - in relation to fisheries - criteria used by Australian authorities (Ward 2000).

The OECD criteria for environmental performance review (OECD 1993) would apply equally to

The OECD critera for indicators for environmental performance reviews (OECD 1993)

Policy relevance and utility for users

An environmental indicator should:

• provide a **representative picture** of environmental conditions, pressures on the environment or society's responses;

• be simple, **easy to interpret** and able to show **trends over time**;

•be **responsive to changes** in the environment and related human activities;

• provide a basis for **international comparisons**;

• be either **national in scope** or applicable to regional environmental issues of national significance;

• have a **threshold or reference value** against which to compare it so that users are able to assess the significance of the values associated with it.

Analytical soundness

An environmental indicator should:

• be theoretically **well founded** in technical and scientific terms;

• be based on international standards and **international consensus** about its validity;

• lend itself to being linked to economic models, forecasting and information systems.

Measurability

The data required to support the indicator should be:

• readily available or made available **at a reasonable cost/benefit** ratio;

• adequately **documented** and of **known quality**;

• **updated** at regular intervals in accordance with reliable procedures.

*These criteria describe the "ideal" indicator and not all of them will be met in practice.

fisheries and emphasises policy relevance and utility for users, analytical soundness and measurability (see text box).

Ward (2000) quote a set of criteria for sustainability indicators which relate more specifically to fisheries and have been used by the Department of Environment, Sport and Territories, Australia in relation to marine ecosystem management. They are generally a rewording of the OECD criteria but with the addition that indicators should 'where possible and appropriate, facilitate community involvement'. The combination of two reservations in this wording may indicate the priority level of this requirement.

In summary, fisheries sustainability indicators should be :

Observable

- within economic resources for research on a sustained basis

- by stake holders, either directly or by transparency in the observation process

Understandable

- they should have research based substance - reflect analytical soundness

- they should reflect features in accordance with stake holders' understanding of the resource system

Acceptable

- by fishers

- by public at large

Related to management

- they should have associated reference values (limits, targets, precautionary etc)

- they should respond to management measures

The identification of indicators meeting these criteria and development of corresponding estimation methods and reference points is still in its infancy and there are thus very few indicators available which can be implemented on short term in management. A list of some of the candidates which have been presented in the literature in recent years is presented below, but the development of the validity and methodology for most of these indicators into operationality is still a research task which lies ahead of us.

3 Candidate methods and indicators

A range of candidates for indicators and reference points have been suggested and are in use. Those in use refer generally to single fish stocks and focuses on the sustainability of the resource base for the fisheries in this limited sense. Indicators referring to habitats, ecosystems or even integrating biological and societal issues are in the process of being identified and have only been used as a basis for actual management in few cases. This is a reflection of the recent history of wider considerations being included in fisheries management.

3.1 Stock indicators

3.1.1 Maximum sustainable yield

Concept

The concept of MSY is the classical sustainability indicator in fisheries biology. This fisheries concept has been used in the general sustainability discussion as a classical example of a concept which combines the need to harvest with sustainability considerations - and even offers and optimisation criterion (for instance by Bell and Morse 1999). MSY is the only fisheries related indicator in the list developed by the Commission for Sustainable Development (CSD 2001) as follow up to chapter 17 of agenda 2001, MSY has been used as a reference point in many fisheries development projects and is established as a legal concept in the US fisheries legislation.

MSY is generally used as a generic concept independently of the methodology to estimate it, based on the simple notion that the harvest from a renewable resource will increase with increasing exploitation pressure, reach a maximum and decrease thereafter. This is intuitively understandable and probably the basis for the success of the concept within fisheries and beyond.

MSY is used as a state indicator even though it must be used in combination with an abundance measure or a pressure indicator - fishing effort or fishing mortality - to make sense. The yield itself is no indicator of the state of the stock or the sustainability of the fisheries as a low yield may result from high and low fishing pressure alike. The variant proposed in the CSD list is either '(i) the ratio between maximum sustained yield (MSY) abundance and actual abundance ; or (ii) the deviation in stock of marine species from the MSY level'.

Scientific basis

MSY is founded in mainstream fisheries biology and several weaknesses of the approach have been identified. The MSY is the indicator in fisheries management which has the longest history and it has been subject to extensive discussion and critique. An evaluation of the utility of this concept may therefore seem excessively negative compared to other candidates which only recently have attracted attention.

The dynamics underlying the yield curve and thus the estimation of MSY has basically been modelled in two ways - as an extension of a logistic growth curve of individuals or biomass in a population (the surplus production models from Schaefer onwards) and on basis of an age structured population model following mainstream Baranov - Beverton and Holt fisheries biology. The two approaches are in principle equivalent but they are in practice very different because of their different data needs and especially because the assumptions underlying the estimation are very different.

The age structured approach can capture changes in fishing patterns and recruitment over time which circumvents some of the most serious shortcomings of the surplus production models. However, data requirements are much higher and new assumptions must be introduced in the estimation. The estimate of MSY and the corresponding fishing mortality is especially sensitive to the assumptions regarding natural mortality in an age structured model.

MSY has been developed as a single species concept, but it can - in the surplus production model variant - equally well be applied to a multistock resource base as long at the objective is to estimate the overall yield potentials of the stocks without consideration of the abundance or even extinction of single stocks within the aggregate.

The shortcomings of the MSY concept are many and have been demonstrated both on the theoretical and the practical level. One of the earliest applications was in the Peruvian anchovy fishery where it lead to overestimates of sustainable effort (Laws 1997).

The problem demonstrated in this case and in many other cases later is that the maximum of the yield curve is very poorly estimated when only data at the left and close to the maximum are available.

Another and even more serious problem with the MSY concept when used as single indicator is that it decouples exploitation and recruitment. The 'sustainability' term implies that the spawning stock remaining at MSY exploitation will be sufficient to maintain reproduction, but this is not explored explicitly as an integral part of the model.

Resource requirements - data and institutions

Estimation of MSY on basis of surplus production models requires catch and effort data from both low and high exploitation - in practice beyond MSY - with the same fishing fleet or with fleets which can be standardised to the same effort measure. On all levels of exploitation an equilibrium between effort and yield should have been achieved. Such data never exist and various proposals have been made to repair for shortcomings, especially in relation to the equilibrium requirement.

Estimation of MSY on basis of a full analytical assessment in an age structured model requires the same extensive time series of age disaggregated catch data and supporting effort data as discussed in relation to analytical assessments.

Acceptance

The MSY concept has - as a generic concept - been widely embraced by . The specific estimates and specific methodologies may however be subject to the same problems as analytical assessments (see below).

Management issues

The estimation of MSY and the corresponding effort level on basis of catch and effort data will generally require that effort well beyond the effort corresponding to MSY has been exerted over some time.

Fishing at MSY is generally considered excessive. This is however not a basic problem with the indicator as such, reference points can be defined at lower levels but relative to MSY.

3.1.2 Simple stock size measures

Concept

Direct observation of stock size is a powerful state indicator. However, such observations are not available from fisheries, but it may be relevant to use indirect observations of stock abundance such as catch per unit effort or even total landings. Abundance measurements or proxies of abundance are not in themselves sufficient to identify reference points or direction of management.

Scientific basis

CPUE is considered proportional to stock size if the catchability is constant over time and independent

of overall stock size (by definition). Total landings are under the same conditions an abundance indicator if effort can be considered constant. Changes in area of distribution of a stock will confound both indicators. Independence between catchability and stock size requires random spatial distribution of the fishery relative to the stock distribution which is the exception rather than the rule in any fishery except in relation to some bycatch species. Constant catchability over time is also the exception due to developments in fishing technology.

Acceptance

Changes in catch rates are generally accepted by fishermen as a measure of changes in abundance.

Resource requirements - data and institutions

Collection of basic and representative catch and effort data is the first requirement of any fisheries statistical system. Data collected for basic statistical purposes may however not be adequate to track changes in CPUE - either because of insufficient disaggregation or definition of catch categories/species or because of insufficient recording of vessel or gear specifics.

Management issues

CPUE data are - as any other indicator - only useful for management if they can be related to reference points which make it possible to ascertain the relative position of the present state. Such reference points must be established by other means. This is trivial, but statements using decline of catch rates as a direct indication of overfishing are still commonplace

CPUE data may be useful to monitor the changes in the resource system over time.

3.1.3 Exerted effort

The effort applied in a fishery is the simplest pressure indicator. Standardisation between fleets and over time is the main problem if effort is to be used as an indicator of overall trends in fishing pressure. See the discussion above regarding CPUE based indicators.

3.1.4 Size composition

It is one of the basic conclusions of fisheries biology that the average size of fish in the catches and in the stock will decline when fishing mortality increases. The size distribution in the landings may therefore be used as a state indicator. The use of size is subject to the same qualifications as the use of CPUE as discussed above including the problem that this indicator in itself is insufficient to establish reference points and thus direction for management. Length frequency sampling as a considerable add-on to catch and effort sampling and the added utility may not justify the costs if the only use of the data is to monitor size changes. Relative size information of more conspicuous species may be available from indigenous knowledge.

3.1.5 F and Biomass based indicators

Concept

These are the indicators used in analytical assessments. The term 'indicator' is usually not used within the classical stock assessment approach, but fishing mortality and spawning stock biomass are basically pressure and state indicators respectively - although indicators which rely on estimates within a model of the underlying processes.

The literature on indicators and reference points in fisheries management is nearly exclusively build on these concepts and the identification and use of limit and target (or precautionary approach) reference points formulated on basis of fishing mortality and spawning stock biomass is the underlying concept in the FAO Code of Conduct for Responsible Fisheries and the fisheries policy of most North Atlantic countries and commissions including Norway and the EU.

Scientific basis

These indicators are firmly rooted in mainstream fisheries biology in the Baranov - Beverton and Holt tradition.

The main current issues in relation to application relate to estimation methodology and uncertainty. The estimation methods available are various variants of fitting population dynamics models to a catch-at-age matrix - either through back-calculating (VPA) techniques supplemented with some 'tuning' procedure using CPUE data or through more rigid statistical procedures. The development of methods has in recent years increasingly focussed on the estimation of the uncertainties associated with the parameter estimates. This is based on a realisation among scientists and managers alike that the failure to understand the uncertainties has lead to serious management failures and collapse of fisheries (Cochrane 1999). There is an increasing awareness of the uncertainties arising from the model structure itself although the evaluation of these is less straightforward (Anon 1999, McAllister et al 1999).

Application of these methodologies in a tropical development context have generally not been successful. The immediate obstacle - that it may be necessary to use length as a proxy for age due to the difficulties in ageing tropical fish - is not the major problem. The main problems in implementation have been the multispecies character of the fisheries and the insufficiency of the data bases available relative to the requirements of analytical assessments.

Resource requirements - data and institutions

The basic requirement for data are time series of catch-at age data and CPUE data from a subset of commercial fleets or from surveys. Regular, representative and reliable collection of such data is costly and requires considerable institutional capacity -both in terms of logistics and analytic capacity and in terms of institutional stability over time. Useful catch data may be impossible to get even when institutional capacity is present due to extensive discarding and mis- and nonreporting. The production of consistent and representative time series of survey data are beyond economic reach of most developing countries.

Survey data are a requirement under the following conditions :

- for recruitment estimation : the fisheries is regulated by quotas based on catch predictions from the stock assessment, it is highly dependent on recruiting year classes and the abundance of these

yearclasses is very variable and can not be estimated from data from the commercial fishery or any other means than recruitment surveys

- as source of CPUE data: there is no subset of commercial CPUE data which reflect stock abundance

Survey data have also been used as the sole basis for assessments in cases where commercial catch data were non-existing or highly unreliable (Cook 1997) or to get initial estimates of resources before fisheries are developed as a guidance to development.

Acceptance

Although F and SSB indicators may have some intuitive appeal, the methods applied in analysis originate in a science discourse which is alien to fishermen and to most other stake holders (see for instance several contributions in Neis and Felt 2000).

Management issues

The F and SSB indicators originating from analytical assessments are generally used to produce catch predictions in the context of quota regulated fisheries with extensive over capacity. The requirements for precision in these cases are high and more often than not beyond the technical capabilities of the methods applied. Attempts have been made to alleviate the problem by introducing uncertainty estimates and using so called pa (precautionary approach) reference points which should incorporate a safety margin accounting for uncertainty (ICES 2000). It has however been difficult to capture model uncertainty in these procedures (ICES 1999) and the use of even pa reference points may lead to overexploitation. The problem in these situations is however not the assessment methodology per se but that the excessive capacity drives the management system beyond the capabilities of the knowledge base.

F and SSB indicators from analytical assessments may also be used as a guidance in other management regimes based primarily on effort control and technical measures. The stock trend estimates (the F and SSB history) from analytical assessments are generally considered more robust than the catch predictions. An adaptive management system based on effort control and technical measures can utilise stock trend estimates from analytical assessments to identify reference points and the direction of measures to be implemented.

3.2 Habitat indicators

3.2.1 Habitat extent and quality

Concept

The area of habitat types and the species composition within habitats are used as state indicators which are assumed to respond to a variety of external pressures beyond fisheries.

Scientific basis

Habitat extent and the integrity of communities in habitats are core concerns in conservation biology. An example are the Australian criteria for marine ecosystems (Ward 2000) which include a range of indicators relating to area of various habitat types and species presence within these habitats. Theses indicators are linked to fisheries through the importance of the specific habitats to the productivity of fish stocks and through the impact of fisheries on the habitats. The basis and methodologies were discussed

by the ICES Working Group on Marine Habitat Mapping (Anon 1999).

Resource requirements - data and institutions

These indicators require monitoring of habitat areas and sampling of those species which are included in the indicators. Absolute measurements of these indicators in the larger marine environment is a costly undertaking and is not something which can be expected to be done on a regular basis with, say, yearly intervals, see Anon (1999). However, used as relative measures and in a local context in the coastal zone or in some freshwater habitats, this kind of indicators may lend themselves to monitoring in close cooperation with the local communities or with stake holders. The relative changes in extent of local habitats and changes in abundance of conspicuous species within habitats may be directly observable by local resource users and may even be available on a historical basis as a part of the indigenous knowledge.

Acceptance

The possibilities for direct observation and the intuitive ease with which these indicators are connected to fisheries may be a good foundation for acceptance. This must however be demonstrated in practice.

Management issues

Changes in habitat extent and quality can be used as an important indicator to evaluate the impact of other external factors on fisheries, especially in coastal and freshwater environments. The loss of mangrove, reef and floodplain areas and integrity will be important factors in management decisions.

The impact of fisheries on habitats can be managed through gear restrictions and closed areas. The recent debate on marine protected areas has extended the objective of closed area management from protection of specific sensitive habitats to a more general measure to reduce overall pressure on the marine ecosystem and to create a buffer in the system by excluding a pool of both targeted and other species from exploitation altogether.

3.2.2 Essential fish habitat

Concept

Essential fish habitats are habitats which are essential to a part of the life cycle of a specific species. This may be suitable substrates for spawning or juvenile feeding areas. The extent and quality of such essential habitats may be used as a state indicator. The essential fish habitat concept has legal status in US fisheries management.

Scientific basis

There are many cases demonstrating that fish stocks depend on survival in critical life stages and thus are dependent on the availability of suitable habitats (in the wide sense where habitat is understood as the presence in the appropriate time-space window of a specific combination of physico-chemical and biological conditions) during these stages. The surge in recruitment research in the late 1970s-1980s was largely triggered by expectations that recruitment variability could be explained if such critical stages and habitats were identified. Progress has been made in the scientific domain, but the US experience, where essential fish habitats have status as a legal concept, has been that it has been difficult to establish identifications of essential habitats which are operational in relation to management. Such identification

may however be more accessible in some coastal and freshwater environments in developing countries where the rich literature on mangrove, coral reef and flood plain habitats indicate clear linkages.

Resource requirements - data and institutions

The identification of essential fish habitats implies tracking the life cycles of individual fish stocks and identifying stages which are critical in relation to habitat availability. Mapping and monitoring of identified essential habitats may be even more complex if habitats have been identified on basis of parameters which are more temporal and evasive than stationary substrate characteristics. However, relative measures of essential habitat extent using generalised information on habitat linkages in those cases where such linkages have been demonstrated may be obtained at low cost and with participation of resource users.

Acceptance and management issues - se remarks regarding habitat extent and quality above.

3.3 Ecosystem indicators

Ecosystem indicators of fishing impact have not yet been developed into operationality although there have been a range of suggestions regarding candidates. NMFS (1999) notes the difficulties in defining a healthy ecocystem and recommends that 'unhealthy' states to be avoided are identified along with health endicators. As examples of ecosystem helth indidators or indicators of states to be avoided related to fishing, NMFS (1999) mention : 'prevent the extinction of any ecosystem component, to maintain a specific, high mean trophic level in the ecosystem or to maintain benthic biomass within the range of natural variability'. Murawski (2000) considered ecosystem overfishing indicators to include important stocks or species assemblages falling below minimum biologically acceptable limits, diversity of communities or populations declining significantly, increasing year-to-year variation in catches or populations, increased sensitivity of the ecosystem to perturbations arising from non-biological factors, lower overall benefits to society and mortalities resulting from fisheries impairing the long-term viability of non-resource species. Rice (2000) provided a review of candidates of community metrics which may be used to evaluate fishery impacts including diversity indices, ordination methods, size spectra, k-dominance curves and trophic properties. He studied those metrics for which most work has been done to develop a theoretical basis and quantification methods but concluded that there is yet no metric that can be endorsed without reservation. He also concluded that it is hard to apply ecosystem metrics with a known level of rigour and to interpret the results with a high degree of scientific objectivity.

It should be clear that some work still has to be done before such indicators can be monitored and used in practical management. Some of the candidates are briefly discussed below.

3.3.1 Species composition 1 - sensitive species as indicator of overall fisheries pressure

The abundance (or even extinction) of sensitive species may be used as an indicator of the overall fishing pressure on the system, especially when such species are mainly taken as incidental by-catch. The example which has been best documented is probably the reduction or extinction of stocks of sharks and rays worldwide including species for which there have never been any directed fisheries (Stevens et al 2000).

The practical utility of such information in the short term is limited due to the large variability of short time series of catch data on species with low abundance. Data on by-catches are generally not available and must be sampled specifically from the fisheries or by surveys. Historical information may be available

from indigenous knowledge.

3.3.2 Species composition 2 - trophic composition

The average trophic level based on Ecopath modelling has been proposed as an indicator of the fishing pressure on the marine ecosystem (Pauly et al 1998). It is specifically suggested that an observation of decreasing average trophic level in catches can be explained by fisheries initially having targeted apical predators. Predation pressure on lower levels have been released as a result and lower trophic levels have then subsequently been targeted by fisheries as the apical predators became scarce. This explanation has however been questioned (Caddy and Garibaldi 2000).

The estimation of average trophic level in a specific ecosystem according to this approach requires the development of an Ecopath model of the system.

The trophic level changes can only be observed over long time periods. The trophic composition of catches is therefore not expected to be useful as an indicator to guide management on short or medium term.

3.3.3 Species composition 3 - gene pool/ biodioversity

Biodiversity is a general term used to cover several aspects of biological systems including species richness and evenness (Hill 1973) and intra-species genetic variability (Ryman et al 1995). Results from studies linking various biodiversity measures to fisheries have so far not shown clear, general trends. A global comparison of demersal fish communities revealed no general indication that fishing had changed the community in the direction of lower diversity (Bianchi et al 2000). Apparent increases in diversity in some of the cases in this study might be influenced by improved survey methodology.

Biodiversity measures are entirely dependent on the sampling methodology applied including gear charcateristics, sampling strategy and identification procedures. Two measurements are basically incomparable if they have been made with different methods. This makes cross-system comparisons difficult and time series from the same system are generally subject to changes in methodology over time.

A more operational approximation may be an interpretation of the changes in individual species.

3.3.4 Size composition/size spectra

Size spectra seem so far to be the most promising candidate among those proposed as indicators for ecosystem effects of fishing.

The relation between changes in size spectra of fish populations and fisheries is well established both theoretically (Gislasson and Lassen1997, Gislason and Rice 1998) and empirically (Gislasson and Rice 1996, Bianchi et al 2000). The cross system study of Bianchi et al (2000) concluded that the slope of size spectra are related to fishing intensity but also that the specific interpretation within an area is not straightforward. This study also concluded that size spectra can be used as a coarse indicator of exploitation in poorly known areas.

Size spectra have been constructed on basis of length compositions (age compositions converted into size) from catch data, survey data and population estimates from multispecies models. All data sources are subject to high annual variability due to variable recruitment. Conclusions regarding exploitation can only be made on basis of averages over several years. Size spectra do thus require size information to be available from the fishery or from surveys and covering several years. This may be impossible to obtain in many cases.

Size spectra share one problem with most other ecosystem indicators : the underlying theory and the estimation models belong entirely in the research domain and the results may not be easily acceptable by other stake holders.

3.4 Synthesising indicators

The sustainability of fisheries is a multidimensional issue and it is therefore necessary to balance information regarding many aspects of the fisheries in the management decision process. Finding and deciding on this balance is the very core of any multi-stakeholder decision process, which only can be played out in an interaction between people where arguments and interests are stated, debated and some consensus reached.

There has however been attempts to develop methods to synthesise these multiple dimensions into summary statements about the overall balance of the fisheries system. These statements may be presented in a graphical form where various indicators have been condensed into a few axes representing aggregated aspects of social and ecological sustainability. The Sustainable Development Reference System (SDRS) of FAO (Garcia and Staples 2000) suggest 'human well-being' to be represented by jobs and revenues and 'ecological well-being' to be represented by nurseries and spewning biomass. The Rapid Fisheries Appraisal (RAPFISH) approach (Pitcher and Preikshot 2001) uses a more complex aggregation of indicators along four axes - ecological, economic, social and technological. The benefit of these approaches is that they condense a very complex set of information into a single reasonably simple image. It is however essential for such didactic tools to be useful that all stake holders agree on the selection and scaling of the underlying indicators and on the methodology used for their aggregation. It is difficult to see how such condensation procedures *in silicio* can replace the exchange of viewpoints between real people.

4 The future

It does in summary seem harder than ever before to identify a relevant knowledge basis for fisheries management in developing countries; We have lost several illusions about the possibilities of transplanting mainstream stock assessment methods with minor modifications to accommodate the ageing problem and we have even become more sceptical towards mainstream methods in our own environment. New considerations have been added on top of standard yield maximisation and the costs of data gathering and assessments to address these considerations may have increased out of reach in the North Atlantic while even the costs of basic data collection schemes may be prohibitive in many developing countries. New and simple indicators which fulfil a basic set of reasonable objectives regarding scientific and social validity and management utility have not been identified yet or those which have been proposed are still far from maturation into operationality.

The only way forward is to continue and expand the research work which is already starting up to

identify, develop and operationalise fisheries sustainability indicators which fulfil requirements for observability, understandability, acceptance and management relevance.

5 References

Neis, B. and Felt, L (eds). 2000. Finding our Sea Legs. Linking Fishery People and Their Knowledge with Science and Management. ISER Books, Memorial University of Newfoundland. St John's, Newfoundland.

Anon. 1993. OECD core set of indicators for environment performance reviews. OECD. http://www.oecd.org/env/docs/gd93179.pdf

Anon. 1999. Ecosystem Based Fisheries Management. A Report to Congress. National Marine Fisheries Service. http://www.nmfs.noaa.gov/sfa/EPAPrpt.pdf

Anon. 1999. Report from the Study Group on Marine Habitat Mapping. ICES CM 1999/E:10. ICES.

Anon. 1999. Report of the Comprehensive Fishery Evaluation Group. ICES CM 1999/D:1. ICES.

Anon. 2000. Sustainability indicators in Marine Capture Fisheries: papers derived from a technical consultation organised by the Australian Department of Primary Industries in co-operation with FAO. Mar.Freshwater Res. 51.

Anon. 2001. Ecosystem Effects of Fishing. Proceedings of an ICES/SCOR Symposium. ICES J.Mar.Sci. 57[3]: 465-791.

Bell, Simon and Stephen Morse. 1999. Sustainability indicators. Measuring the immeasurable. Earthscan Publishers. London.

Bianchi, G., H. Gislasson, K. Graham, L. Hill, X. Jin, K. Koranteng, S. Manickchand-Heileman, I. Paya, K. Sainsbury, F. Sanchex, and K. Zwanenburg. 2000. Impact of fishing on size composition and diversity of demersal fish communities. ICES J.Mar.Sci. 57: 558-571.

Caddy, J. F and R Mahon. 1995. Reference Points for Fisheries Management. [347]. Rome, FAO. FAO Fisheries Technical Paper.

Caddy, J. F. and L. Garibaldi. 2000. Apparent changes in the trophic composition of world marine harvests: the perspective from the FAO capture database. Ocean & Coastal Management 43: 615-655.

Cochrane, K. L. 1999. Complexity in fisheries and limitations in the increasing complexity of fisheries management. ICES J.Mar.Sci. 56: 917-926.

Commission on Sustainable Development. 2001. Indicators of sustainable development. www.un.org/esa/sustdev/isd.htm

Cook, R. M. 1997. Stock trends in six North Sea stocks as revealed by an analysis of research vessel surveys. ICES J.Mar.Sci. 54: 924-933.

Dahl, Arthur Lyon. 2000. Using indicators to measure sustainability: recent methodological and conceptual developments. Mar.Freshwater Res. 51: 427-433.

Garcia, S. M, D. J Staples, and J. Chesson. 1999. The FAO guidelines for the development and use

of indicators for sustainable development of marine capture fisheries and an Australian example of their application. ICES CM 1999/P:05.

Garcia, S. M and D.J.Staples. 2000. Sustainability reference systems and indicators for responsible marine capture fisheries : a review of concepts and elements for a set of guidelines. Mar.Freshwater Res. 51: 385-426.

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

Gislasson, H. and J. C. Rice. 1998. Modelling the response of size and diversity spectra of fish assemblages to changes in exploitation. ICES J.Mar.Sci. 55: 362-370.

Hall, S. J. 1999. The ecosystem effects of fishing. Chapman and Hall.

Jennings, S. J. and M. J Kaiser. 1998. The effects of fishing on marine ecosystems. Advances in Marine Biology 34: 201-252.

Kaiser, M. J and S. J. deGroot. 2001. The effect of fishing on non-target spcies and habitats. Blackwell Science. Oxford.

Laws, E. A. 1997. El Nino and the Peruvian Anchovy Fishery. University Science Books. Sausalito, California.

McAllister, M. K., P. J. Starr, V. R. Restrepo, and G. P. Kirkwood. 1999. Formulating quantitative methods to evaluate fishery-management systems: what fishery processes should be modelled and what trade-offs should be made ? ICES J.Mar.Sci. 56: 900-916.

McGoodwin, J. R., B. Neis, and L. Felt 2001. Integrating Fishery People and Their Knowledge into Fisheries Sceince and Resource Management: Issues, Prospects and Problems. p.249-264 in: Neis, B. and Felt, L (eds.) 2001. Finding our Legs. ISER Books. St John's, Newfoundland.

Murawski, S. A. 2000. Definitions of overfishing from an ecosystem perspective. ICES J.Mar.Sci. 57: 649-658.

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

Pitcher, T. J. and D. Preikshot. 2001. RAPFISH: a rapid appraisal technique to evaluate the sustainability status of fisheries. Fisheries Research 49: 255-270.

Rice, J. C. and H. Gislasson. 1996. Patterns of change in the size spectra of numbers and diversity of the North Sea fish assemblage, as reflected in surveys and models. ICES J.Mar.Sci. 53: 1214-1225.

Rice, J. C. 2000. Evaluating fishery impacts using metrics of community structure. ICES J.Mar.Sci. 57: 682-688.

Stevens, J. D., R. Bonfil, N. K. Dulvy, and P. A. Walker. 2000. The effects of fishing on sharks, rays and cimaeras (chondrichtyans), and the implications for marine ecosystems. ICES J.Mar.Sci. 57: 476-494.