Ecosystem Approaches to Fisheries in the Southern Benguela
Afr. J. mar. Sci. 26: 9–35
2004

AN ECOSYSTEM APPROACH TO FISHERIES IN THE SOUTHERN BENGUELA CONTEXT

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The 2001 Reykjavík Declaration on Responsible Fisheries in the Marine Ecosystem and the Plan of Implementation of the 2002 World Summit on Sustainable Development highlighted the need in fisheries to look beyond considering only the target species and to consider in fisheries management the impacts of fishing on the ecosystem as a whole, as well as the impacts of the ecosystem on fisheries. This paper examines the practical implications of progressing towards ecosystem approaches by reference, in particular, to the FAO technical guidelines on the topic. It goes on to examine the major fishery types in South Africa and the southern Benguela, and to consider the probable impacts of those fisheries on target species, bycatch species and the ecosystem, as well as the indirect impacts on other affected species. The review reveals that all fisheries have impacts beyond the target species and that an ecosystem approach is required in order to ensure the long-term sustainability of the living marine resources of the southern Benguela and the ecosystem as a whole. Finally, the likely obstacles to successful implementation of an ecosystem approach to fisheries in the southern Benguela are discussed.

Key words: ecosystem, fisheries management, impacts, interactions, objectives, southern Benguela

The concept of an ecosystem approach to fisheries management is not new, and it is likely that even the earliest human users of living marine resources had a reasonable understanding of the interrelationships and interdependence of the different components of the ecosystem from which they were extracting organisms. Certainly, the English fishers of the 14th century who complained to Edward III about the damage that would be caused by the small-mesh beam trawl, the *Wonderchoun*, were aware of the problems of bycatch and damage to the environment caused by fishing gear (Nicolson 1979, Caddy and Cochrane 2000).

Within the context of current attitudes and approaches to fisheries, there are some elements of an ecosystem approach contained in the United Nations Law of the Sea Convention of 1982. However, it could be argued that the real origins of an ecosystem approach to fisheries (EAF) can be found in Chapter 17 of Agenda 21 of the 1992 Rio Declaration on Environment and Development. The major phase of the development of the Code of Conduct for Responsible Fisheries by the Food and Agriculture Organization of the United Nations (FAO 1995) happened shortly after the Rio Declaration, and almost all the major features and requirements of EAF can be found within the Code, even though it does not explicitly refer to EAF. By the end of the 1990s, leading fishing nations such as Australia (e.g. Smith *et al.* 1999) and the USA (National Research Council 1999) were actively moving towards an ecosystem orientation in their fisheries management.

At an international level, the role and importance of EAF was recognized by the 47 countries participating in the Reykjavík Conference on Responsible Fisheries in the Marine Ecosystem, held in October 2001. At the end of that conference, all but two of those countries issued the Reykjavík Declaration on Responsible Fisheries in the Marine Ecosystem, which included a declaration "… that, in an effort to reinforce responsible and sustainable fisheries in the marine ecosystem, we will individually and collectively work on incorporating ecosystem considerations into that management…" (FAO 2001, p. 106). This Declaration was recognized and reinforced at the World Summit for Sustainable Development in Johannesburg in 2002. The Plan of Implementation of this Summit included the exhortation to "Encourage the application by 2010 of the ecosystem approach, noting the Reykjavík Declaration on Responsible Fisheries in the Marine Ecosystem and decision 5/6 of the Conference of Parties to the Convention on Biological Diversity" (Paragraph 29d; http://www.johannesburgsummit.org/html/documents/summit_docs/2309_planfinal.htm).

There is therefore international pressure on all fishing

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Received February 2004; accepted June 2004

Table I: The 12 principles of an ecosystem approach provided by the Convention on Biological Diversity (Decision V/6)

1:	The objectives of management of land, water and living resources are a matter of societal choice.
2:	Management should be decentralized to the lowest appropriate level.
3:	Ecosystem managers should consider the effects (actual or potential) of their activities on adjacent and other ecosystems.
4:	Recognizing potential gains from management, there is usually a need to understand and manage the ecosystem in an economic context. Any such ecosystem-management programme should: (a) reduce those market distortions that adversely affect biological diversity; (b) align incentives to promote biodiversity conservation and sustainable use; (c) internalize costs and benefits in the given ecosystem to the extent feasible.
5:	Conservation of ecosystem structure and functioning, in order to maintain ecosystem services, should be a priority target of the ecosystem approach.
6:	Ecosystems must be managed within the limits of their functioning.
7:	The ecosystem approach should be undertaken at the appropriate spatial and temporal scales.
8:	Recognizing the varying temporal scales and lag-effects that characterize ecosystem processes; objectives for ecosystem management should be set for the long term.
9:	Management must recognize that change is inevitable.
10:	The ecosystem approach should seek the appropriate balance between, and integration of, conservation and use of biological diversity.
11:	The ecosystem approach should consider all forms of relevant information, including scientific and indigenous and local knowledge, innovations and practices.
12:	The ecosystem approach should involve all relevant sectors of society and scientific disciplines.

nations to begin to implement an ecosystem approach in their domestic fisheries and in any international fisheries in which they participate. As with the Code of Conduct, implementation is likely to be slow, and many countries, agencies and individuals are still grappling with interpreting just what is intended by the term EAF. This paper attempts to address that question. In doing so it refers particularly to the FAO Guidelines on EAF (FAO 2003), but also goes beyond these to look at other interpretations and issues. Finally, it explores some of the needs and implications of implementing EAF in the South African fisheries of the southern Benguela ecosystem.

EAF: CLEARING THE MIST

EAF can mean different things to different people. Lackey (1999) quoted several interpretations, ranging from "The move to ecosystem management concepts is an evolutionary approach that has been underway for decades…" to "Ecosystem management defines a paradigm that weaves biophysical and social threads into a tapestry of beauty, health and sustainability". This diversity of views is well encapsulated in another quote from Lackey's paper: "I promise you that I can justify anything you want to do by saying it is ecosystem management".

The latter quotation may have been reflecting the reality, but does little to help the confused or bemused manager or policy-maker as he or she wrestles with the problem of just what they are being asked to do now. The ideal of an ecosystem approach is summarized by Chapter 17 of Agenda 21: "The marine environment – including the oceans and all seas and adjacent coastal areas – forms an integrated whole that is an essential component of the global life-support system and a positive asset that presents opportunities for sustainable development. International law … sets forth rights and obligations of States and provides the international basis upon which to pursue the protection and sustainable development of the marine and coastal environment and its resources." A number of attempts have been made to translate this ideal into a practical and feasible approach, including those of the National Research Council of the United States of America (1999), the Convention on Biological

Table II: Principles of an ecosystem approach to fisheries (FAO 2003)

1:	Natural resources should not be allowed to decrease below their level of maximum productivity.
2:	Fisheries should be managed to minimize their impact on the ecosystem.
3:	Ecological relationships between harvested, dependent and associated species should be maintained.
4:	Management measures should be compatible across the entire distribution of the resource (across jurisdictions and management plans).
5:	Because the knowledge on ecosystems is incomplete, the precautionary approach should be taken.
6:	Governance should ensure both human and ecosystem well-being, and equity.

Diversity (Decision V/6 of the Conference of the Parties, 2000), and the World Wide Fund for Nature (Ward *et al.* 2002).

Building on the work done by these and other groups, the FAO developed an interpretation consisting of a rationale and a definition, which should be read and interpreted together, and further expanded on the human component (FAO 2003):

Rationale — "The purpose of an ecosystem approach to fisheries is to plan, develop and manage fisheries in a manner that addresses the multiplicity of societal needs and desires, without jeopardizing the options for future generations to benefit from the full range of goods and services provided by marine ecosystems."

Definition — "An Ecosystem Approach to Fisheries strives to balance diverse societal objectives, by taking account of the knowledge and uncertainties about biotic, abiotic and human components of ecosystems and their interactions and applying an integrated approach to fisheries within ecologically meaningful boundaries."

The Convention on Biological Diversity provided a set of 12 principles for an ecosystem approach (Table I), whereas the FAO Guidelines provide a shorter list of principles focused on an ecosystem approach to fisheries management (Table II). Overall, the FAO Guidelines attempt to take a practical and pragmatic approach to implementing EAF, one that emphasizes evolution rather than revolution. This evolutionary approach may be seen by some interest groups to be lacking the necessary sense of urgency and imperative, but the difficulty of changing attitudes and behaviour in fisheries needs to be recognized. This inertia is usually generated by social and economic dependence on fisheries, which cannot simply be ignored. An evolutionary approach may therefore be both more realistic and more likely to gain support of the interest groups dependent on marine ecosystems for their livelihood, which will be the groups most directly affected by such change.

THE FAO RECOMMENDATIONS ON IMPLEMENTING EAF: OVERVIEW OF THE STRUCTURE OF THE GUIDELINES

The FAO Guidelines on EAF (FAO 2003) were produced as a supplement to the earlier Technical Guidelines for Responsible Fisheries: Fisheries Management (FAO 1997). The EAF Guidelines focus strongly on process, emphasizing the steps and tasks involved in an ecosystem approach to fisheries management. The details of issues such as the management objectives, the management measures used to achieve those objectives, the capacity of the management agency, and the nature of the fisheries will vary considerably from case to case. It is therefore more useful to present a generic process for arriving at a case-specific solution, rather than to attempt to provide generic solutions that must inevitably lack detail.

Table III presents the major headings and subheadings of the Guidelines as an illustration of the broader considerations requiring attention when implementing an EAF. Some sections of the Guidelines are discussed in more detail below, but this paper does not attempt to describe or summarize them as a whole.

Making EAF operational

The definitions developed by the Convention on Biological Diversity and FAO for an ecosystem approach provide a broad overview of what is required and in-

Table III: The basic structure of the FAO Guidelines on the ecosystem approach to fisheries (FAO 2003)

1. Introduction
   - 1.1 The need for and benefits of an ecosystem approach to fisheries (EAF)
   - 1.2 What is an EAF?
   - 1.3 Making EAF operational
   - 1.4 Moving towards EAF management
2. Ecosystem approach to fisheries data and information requirements and use
   - 2.1 Policy formulation
   - 2.2 Developing management plans
   - 2.3 Monitoring, implementing and performance reviews
   - 2.4 Uncertainty and the role of research
3. Management measures and approaches
   - 3.1 Introduction
   - 3.2 Options to manage fishing
     - Technical measures
     - Input (effort) and output (catch) control
     - Ecosystem manipulation
     - Rights-based management approaches
   - 3.3 Creating incentives for EAF
   - 3.4 Assessing costs and benefits of EAF
   - 3.5 Other considerations
4. Management processes
   - 4.1 Developing an EAF management plan
   - 4.2 Legal and institutional aspects of EAF
   - 4.3 Effective monitoring, control and surveillance
5. Research for an improved EAF
   - 5.1 Ecosystems and fishery impact assessments
   - 5.2 Socio-economic considerations
   - 5.3 Assessment of management measures
   - 5.4 Assessment and improving the management process
   - 5.5 Monitoring and assessments
6. Threats to implementing EAF

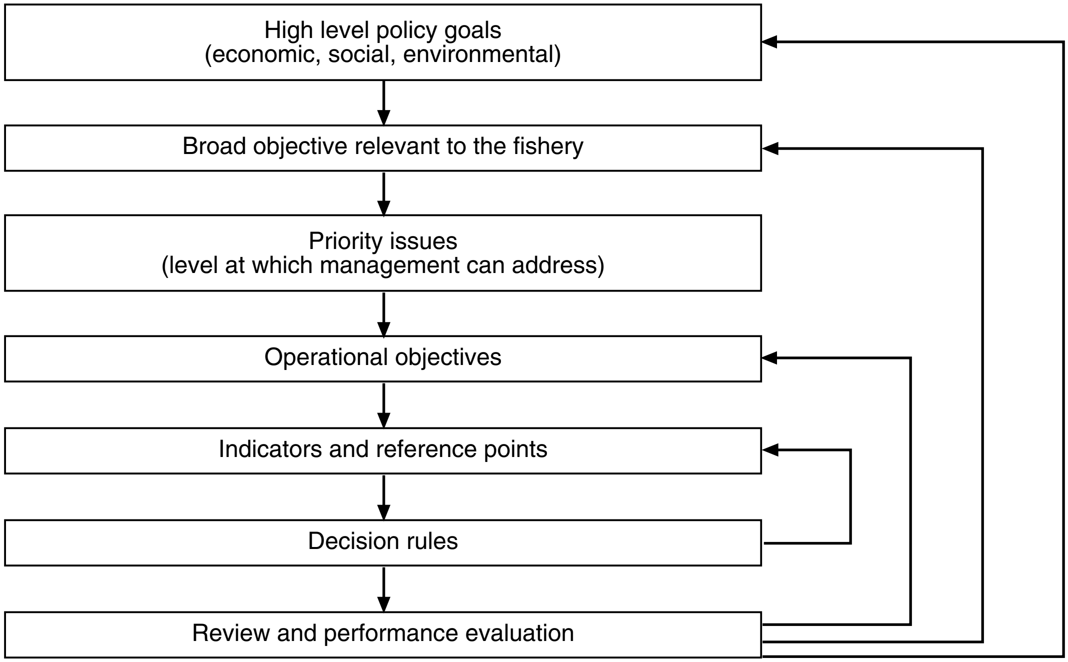

Fig. 1: From policy goals to action (after FAO 2003)

tended by such an approach. In practice, many countries have already included EAF in their national policies, even if under a different name. For example, the objectives and principles in Chapter 1 (Section 2) of the South African Marine Living Resources Act No. 18 of 1998 (Anon. 1998) include the following:

- the need to utilize marine living resources to achieve economic growth, human resource development, capacity building within fisheries and mariculture branches, employment creation *and a sound ecological balance*…;
- the need to protect the ecosystem as a whole, including species which are not targeted for exploitation;
- the need to protect marine biodiversity; etc.

South Africa's draft National Environmental Management: Biodiversity Bill also has bearing on ecosystem issues, referring in its prelude to "provide within the framework of the National Environment Act, 1998, for the management and conservation of South Africa's biodiversity; the protection of species and ecosystems that warrant protection; the sustainable use of indigenous biological resources….". In Chapter 4, it provides for "the protection of ecosystems….to ensure the maintenance of their ecological integrity".

These statements form a part of the broad policy goals for fisheries in South Africa and could be interpreted as encompassing the principles of EAF. However, they are still open to diverse interpretations and contain many potential conflicts. They need to be translated into case-specific operational objectives before they can be used to determine management actions. This step requires comprehensive consultation and probably negotiation within and among individual fisheries and other users of fish resources to identify and agree upon consistent operational objectives across the fisheries sectors.

In advising how to develop operational objectives from policy goals, the EAF Guidelines draw on the model used for national reporting on the implementation of ecologically sustainable development for capture fisheries in Australia (Fletcher *et al.* 2002). That model has been successfully used in a number of Australian federal fisheries to involve the major stakeholders in identifying and prioritizing the key issues in each fishery, where the "fishery" is clearly defined by the management agency. The process is simple and, working from a "generic component tree" that covers the range of issues relevant to fisheries in general, the set of stakeholders is required to work down from the high level policy goals, elaborating more and more detailed issues under each goal, until they reach a level of specificity that the management agency can address directly (Fig. 1). For each specific issue, it should be possible to identify an appropriate operational objective related to it, and then to define indicators and reference points reflecting the operational objective. Emphasis on EAF will require the inclusion of a wider range of ecosystem policy goals in the final operational objectives and therefore also require involving the full range of stakeholders in setting those objectives.

Management measures and approaches

There are no new and magical management instruments and measures available to facilitate implementation of EAF. Fisheries management agencies have to rely to a large extent on the same old and tested techniques that have both succeeded and failed over the decades of target-resource orientated management. The basic options remain largely as described in the 1997 Guidelines and elaborated in some depth by the various authors in Cochrane (2002). The carrying out of changes in management measures in implementation of EAF is likely to lead to some conflict with stakeholders, and this needs to be considered and allowed for in the process of developing EAF for specific fisheries. EAF will require that the application of these measures considers and allows for the broader goals of the approach, and this will require wider thinking, greater synergy between measures and, in all probability, more conservative or precautionary application of fishing methods than has usually been the case in the past. For example, Bjordahl (2002) lists the following issues that need to be considered when regulating fishing gear under EAF: selectivity for target species; bycatch; discards; by-mortality; ghost fishing; habitat effects; catch quality; energy efficiency; and pollution.

Technical measures

A primary issue in considering regulation of fishing gear is the need for greater attention to be given to the selectivity characteristics of the gear than has generally been the case in the past. This will include consideration of impacts on the abundance, size structure and genetic composition of the target species, but also the impact of the gear on non-target species. Attention to some of these aspects has been growing in recent years, with progress in the implementation of turtle-excluder devices, bycatch reduction devices, and other modifications to gear and fishing practices to reduce bycatch of, for example, turtles, seabirds and dolphins. These approaches have tended to focus on ameliorating impacts on species of commercial concern and charismatic, often depleted, species and populations. EAF will require maintenance of these efforts and expanding them to include other species, perhaps of little direct interest to humans, but which are components of the ecosystem.

A growing concern in recent years is the impact of fishing gear on ecosystem habitats, especially the bottom habitat. The seminal study by Sainsbury *et al.* (1997), demonstrating the far-reaching impacts of habitat modification by trawling the Australian northwest shelf, is one of the most rigorous and comprehensive studies of this impact, but there have been many other valuable studies (e.g. Hall 1999). An example of such an issue in South Africa was the damage to deep-sea corals and other benthic fauna caused by fishing for panga *Pterogymnus laniarius* with heavy trawl gear. This led to the trawl fishery being discontinued and investigations being launched into trap-fishing. In general, dragged demersal gears can be a particular problem, and a general response to fishing disturbance of benthic communities is for erect and sessile epifauna to be lost, smaller and faster-growing organisms to become more dominant and for there to be a general decline in species diversity. These effects are more severe in habitats of fixed but vulnerable structure, such as boulder, pebble and coral habitats. There is still debate on the impact in areas with mobile sediments or soft, sandy bottoms (Hall 1999, Bjordahl 2002).

Area and time controls have long been used in fisheries for purposes such as reducing mortality on species during vulnerable life stages, reducing bycatch and protecting critical habitats. In South Africa, for example, bays along the South Coast have been closed to trawling, and a one-month closed season has been introduced for chokka squid *Loligo vulgaris reynaudii* fishing at peak spawning (Augustyn *et al.* 1992). In recent years, largely within the movement towards EAF, there has been greater emphasis on marine protected areas (MPAs). The conservation benefits of MPAs are generally well understood and appreciated, although determining the actual characteristics of an MPA or set of MPAs to achieve particular objectives in specific localities is as complex and demanding of information and analysis as any other aspect of fisheries management. The benefits of MPAs for fisheries management are considerably less well studied and less clear (Hall 2002), although there is widespread consensus that they can have an important role to play, especially in providing some buffering against uncertainty and error in the application of other management measures. Reserves are most suitable for resident species, and are unlikely to offer much benefit for overexploited coastal migrants (Bohnsack 1993, Clark 1996). In general, MPAs are unlikely to be effective in isolation and will be most valuable when used as one measure in an integrated management plan (National Research Council 1999).

South Africa has designated 21 MPAs, covering most coastal habitats and 19% of the coastline, but very little of the continental shelf beyond the 100-m isobath. The first of these came into effect in 1964. New MPAs were added over the years, and four large MPAs were created in 2004. By default, all South African MPAs are "no-take", although certain fishing activities may be allowed in them under permit. In practice, this is achieved by zoning. Whereas some MPAs are entirely no-take, almost half are zoned to accommodate some types of fishing.

South Africa's policy with regard to protected areas is to ensure that all habitat types (e.g. rocky shores, coral reefs) and biogeographic zones (e.g. warm-temperate zone) are adequately represented in protected areas (Department of Environmental Affairs and Tourism 2000). In addition, approximately 20% of marine habitat should be included in MPAs (Attwood and Harris 2003), a target that was accepted at the Fifth World Parks Congress in 2003. The basis for that percentage is not firmly rooted in science, but represents a general target, well above that which most countries have been able to achieve. There is some justification on fishery grounds that 20–30% of the fish stock should be protected, but the recommendation is obviously not applicable to every situation. Various factors, including fishing intensity and history of exploitation, could modify the target either way.

Input and output controls

Input (effort) and output (catch) controls are the mainstay of modern fisheries management. Input controls typically address the amount of effort that can be applied in a particular fishery. Ideally they should also address the capacity of the fleet as well, because latent capacity can provide a powerful incentive to allow surplus effort into a fishery in order to meet short-term social and economic goals, even when the resource cannot sustain it. Within the context of EAF, control of fishing effort will frequently result in some control of fishing mortality across all species caught, an advantage in multispecies fisheries such as the South African traditional linefishery. Under EAF, appropriate effort levels would have to be established to take into account the different vulnerabilities and productivities of the range of species caught by each gear. The standard problems of effort control, such as "effort creep" brought about by ongoing technological improvements in fishing methods, and standardization of effort, will still hinder effective use of effort control under EAF.

Catch controls have been widely applied in commercial fisheries and are the basic management measure for most of South Africa's more important commercial species. An advantage of catch controls is that, if properly determined and enforced, they directly affect the fishing mortality on the target species (Pope 2002). Beverton (1994) suggested that they have other advantages, including that they avoid the problems of standardizing effort measurement and monitoring effort creep. In an ecosystem context, they can be effective if complemented with bycatch controls, but can give rise to substantial problems of discarding and high-grading (FAO 2003). They also require effective, often expensive, monitoring of catches and landings.

Ecosystem manipulation

EAF puts a greater emphasis on the interactions between the fishery, the target resources and the rest of the ecosystem. It recognizes that changes in any of these can affect the productivity of the target resources and that direct manipulation of the ecosystem can play a role in restoring or otherwise increasing the productivity of resources. Examples of such manipulations are preventing habitat degradation, restoring already degraded habitat or providing additional habitat, restocking and stock enhancement, culling predators on or competitors with species of fishery or conservation interest, and intentional introductions to boost existing production.

Each of these approaches may be useful, especially as a means to mitigate damage or loss incurred in the past. However, there is still only limited experience in successful ecosystem manipulation, and frequently therefore, there is risk of unexpected consequences. For example, there is pressure from some fisher groups and some managers in South Africa to cull seals on a large scale to enhance the survival of commercially important fish species, but the ecological consequences of that would be difficult to predict. Studies and a workshop showed that there was not enough information available on these interactions to enable clear predictions to be made (Punt and Butterworth 1995). A further example of ecosystem manipulation providing a possible solution to a fishery problem can be found in the lobster–urchin–abalone interactions described later in this paper. Cautious experimentation with careful monitoring of results may provide clarity in this and similar cases.

Rights-based management approaches

In recent decades, the long-standing tradition of open access to marine resources has been discredited, and rapid progress has been made in limiting access to fishery resources, so avoiding the classic problem of the "tragedy of the commons" (Hardin 1968). A system of access rights appropriate for the ecology, social and economic context of a fishery is an important precondition for obtaining the optimal benefits from that fishery. There is a range of options for use rights (Charles 2002), including customary marine tenure and territorial use rights in fishing, limited entry (an input control), and quota allocations (an output control).

The South African Department of Environmental Affairs and Tourism has gone to great lengths and expense to implement a system of access rights across the fisheries sector of the country and, through the same means, to bring about transformation in the fishing industry, while setting appropriate levels of fishing effort. These efforts are beginning to show signs of success (Anon. 2002, Kleinschmidt *et al.* 2003). Imple-mentation of EAF could require changes in allocations to address, for example, interactions between two fisheries. These changes would need to be handled very sensitively given the recent restructuring of the fishing industry.

Creating incentives for EAF

Two institutional features of traditional fisheries management have been widely considered to have been important in the common failure of fisheries around the world: the prevalence of open access fisheries; and an almost ubiquitous reliance on centralized and top-down approaches to management (e.g. Pearse 1994, Symes 1996, Cochrane 2000). Separately and in combination, these two characteristics have failed to provide positive incentives to fishers to utilize resources sustainably, and have rather had the opposite effect. The problems have been recognized in target-resource orientated fisheries, and considerable progress has been made in implementing suitable systems of property or user-rights for fisheries and in fostering more participatory approaches to management (e.g. Charles 2002, Pinkerton 2002). In South Africa, the authorities have strongly supported such approaches with respect to traditional subsistence fisheries by means of a targeted and partially donor-funded programme, after efforts to introduce a management system initially met with much resistance in the KwaZulu-Natal and Eastern Cape provinces. In the new commercial fisheries management system, Management Working Groups are being introduced to better foster stakeholder inputs, scientific working groups, including an EAF working group, being one of these stakeholder inputs.

In addition to the incentives achieved through acceptable systems of user rights and co-management, the short-term impacts, on fishers in particular, of implementing EAF will frequently require the consideration and use of other incentives that can play a role in encouraging stakeholders to accept and adhere to the requirements of the approach. Some possible areas for creating better incentives are (FAO 2003):

- improvements to the institutional framework (including better management, research and compliance);
- developing collective values for sustainable use through education, training and dissemination of information;
- implementation of non-market incentives such as through taxes or subsidies;
- the creation of market incentives such as eco-labelling and the implementation of tradable property or access rights that provide an incentive to the owner to ensure that the value of the right does not fall through overexploitation.

All these incentives will require considerable attention to make progress in sustainable use of marine ecosystems. If their cooperation is to be obtained in implementation of EAF, consumptive users of marine ecosystems will need to be persuaded that the resulting longer-term benefits for them and society will justify any short-term costs they have to bear. Alternatively, viable and acceptable alternative or supplementary sources of income and livelihood will have to be available for them if reductions in effort are required. The users will also have to be guaranteed long-term access to the resource and hence the opportunity to utilize the longer-term benefits.

Principle 4 of the Convention on Biological Diversity on the ecosystem approach also provides useful guidance on the need for appropriate incentives. The need to "understand and manage the ecosystem in an economic context" requires that any market distortions that negatively impact on biological diversity should be reduced; that incentives to promote biodiversity conservation and sustainable use should be aligned; and that, as far as possible, the costs and benefits associated with use of any specific ecosystem should be internalized. The latter point reflects the problem, also raised by Balmford *et al.* (2002), that frequently the members of society who benefit from conservation, for example through improved aesthetic

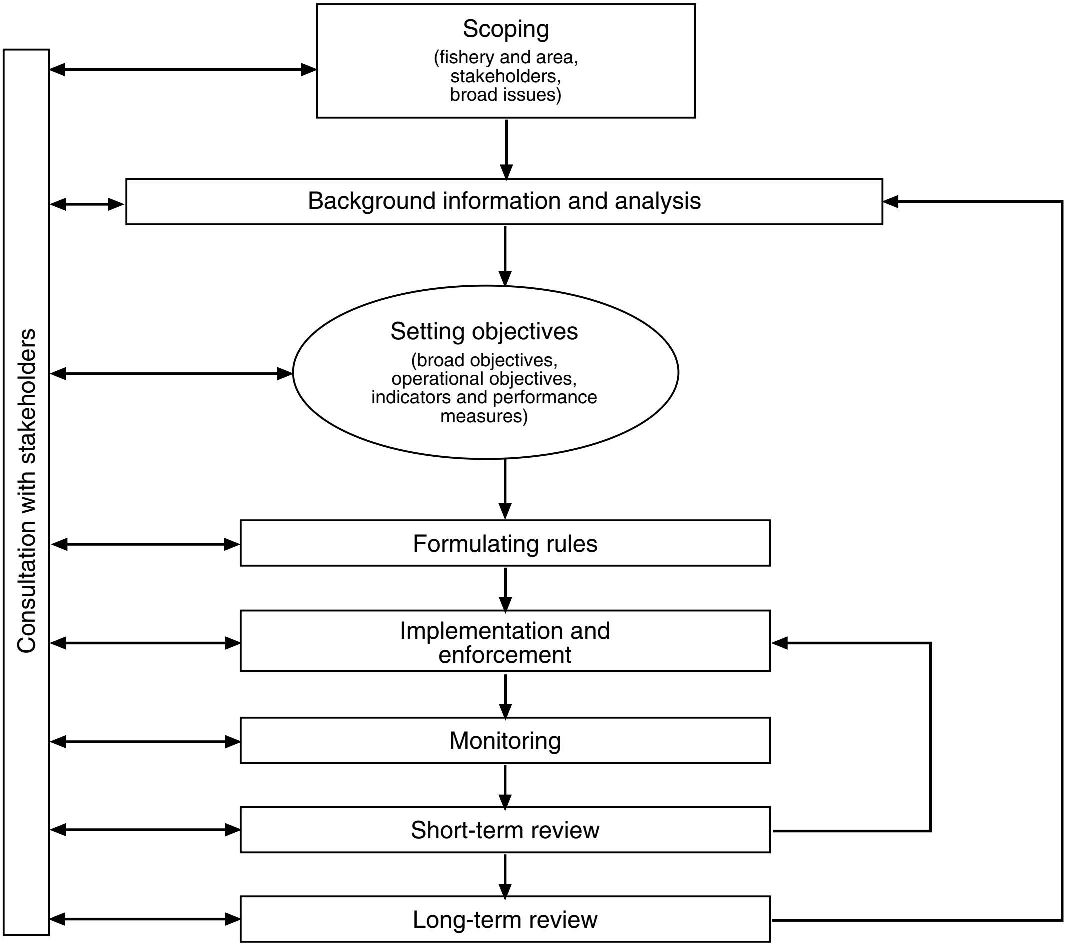

Fig. 2: Developing a management plan (after FAO 2003)

and recreational properties, do not pay the costs associated with that conservation and, at the same time, those who are responsible for some environmental costs, such as pollution, are not required to pay for them.

At this point in the debate, while the conservation benefits of EAF are clear, there is little information available on the social and economic impacts of implementing the approach in already established fisheries. It can easily be anticipated that there will be costs associated with changing gear and fishing practices, reducing effort, reducing catch quotas to reduce bycatch and with other possible requirements for tightening of input and output controls, but what will be the quantifiable benefits, and to whom? For real progress in EAF, this question needs to be addressed by appropriate ecological, economic and social research.

Developing a management plan

The FAO Guidelines emphasize the importance of the cycle of planning, implementation and review (Fig. 2). The most important difference between EAF and conventional fisheries management is that the former will involve a broader range of stakeholders and therefore a broader range of objectives, inevitably leading to a greater number of potential conflicts. The planning phase for establishing a management plan for EAF includes two main tasks: consideration of and agreement on a set of feasible and compatible operational objectives, based on relevant high-level policy goals; and the formulation of management strategies, consisting of a set of management measures, to achieve the operational objectives. Successful development and genuine acceptance of operational objectives will usually be a major task in fisheries, and conflicts between stakeholders may be difficult to resolve, as South African fishery stakeholders know well from the attempts to implement the 1998 Marine Living Resources Act. However, also learned through hard experience, without the development of and agreement on reconciled operational objectives, fisheries management, whether traditional or under EAF, will fail.

Once the operational objectives have been reconciled, it is necessary to develop management rules or strategies that will achieve those objectives. In practice, the two steps tend to be iterative, because the operational objectives may well need to be refined as the performance of different management strategies is considered. The recommended approach for developing strategies is based on that used in operational management procedures (OMPs), an approach already well established in several South African fisheries (Cochrane *et al.* 1998, De Oliveira *et al.* 1998, Geromont *et al.* 1999). A major challenge will be to apply the same process and approach to fisheries and ecosystems where considerably fewer data and less information exist. Finally, models are only as good as the information they contain and the validity of the assumptions on which they are based. Some scientific capacity will need to be used to address key research questions identified as central to implementing, or enhancing, an ecosystem approach.

EAF IN THE SOUTHERN BENGUELA

Research and science

South African marine science is internationally respected, not least for the high quality, integrated studies and programmes on ecosystem and multispecies dynamics and interactions. Much of the ecosystem-orientated research on the southern Benguela took place under the umbrella of the Benguela Ecology Programme (Moloney *et al.* 2004). The programme was established in 1982 with the overall objective "to provide scientific information on the structure and functioning of the constituent ecosystems, to complement the knowledge which is required for the management of the renewable natural resources of the Benguela Current region" (Shannon *et al.* 1988). During its first five-year phase, it involved more than 150 scientists from seven different research organizations. Many of the results of that phase were published in the volume "The Benguela and Comparable Ecosystems" (Payne *et al.* 1987a). The focus of the volume was clearly on multispecies and ecosystem dynamics, and it includes sections on hydrodynamic influences on biological populations, biological interactions in variable systems, whole system ecology and harvesting in complex systems.

The second five-year phase moved away, to some extent, from the focus on the renewable natural resources, and the key objective of that phase was to study the dynamic processes controlling the abundance of standing stocks of key species in the Benguela system (Rothschild and Wooster 1992). That phase was also marked by the publication of a volume reflecting many of the results of the previous five years, "Benguela Trophic Functioning" (Payne *et al.* 1992). Two key sessions in that volume, from the perspective of EAF, were production and energy flows in ecosystems, and influences of predation and competition on fisheries, including multispecies models.

The third phase ran from 1992 to 1995. A poorer general economic climate resulted in a decrease in the funding available for the programme, but it still involved more than 50 scientists covering at least five different professional organizations. The phase was marked by a shift back to a more applied focus, and its primary objective was to make a measurable contribution to the optimal utilization and management of Benguela living resources, with emphasis on the short-lived, commercially important pelagic species, particularly anchovy *Engraulis encrasicolus* (previously known as *E. capensis*), and sardine *Sardinops sagax*, and chokka squid (Cochrane and Krohn 1994). Many of the scientific results of this phase are described in the volume "Benguela Dynamics: Impacts of Variability on Shelf-sea Environments and their Living Resources (Pillar *et al.* 1998).

There have been many other studies relevant to EAF undertaken in South Africa, and such work continues in the country through a number of projects and programmes. Of particular note are the regional programmes, the Benguela Environment Fisheries Interaction and Training (BENEFIT) programme, which is a regional partnership between Namibia, Angola and South Africa focused on fisheries and the marine resources of the Benguela ecosystem off south-western Africa, and the Benguela Current Large Marine Ecosystem (BCLME) programme. The latter is also a multinational programme involving Angola, Namibia and South Africa, which aims to provide support in the management of the living marine resources of the Benguela Current large marine ecosystem in an integrated and sustainable manner, and to protect the marine environment. It would not be feasible to review or even list in this paper all the research projects and studies that have been undertaken and have contributed to knowledge relevant to the implementation of EAF in the southern Benguela ecosystem. However, a few key examples include the assessments of the impact of Cape fur seals *Arctocephalus pusillus pusillus* on Cape hake *Merluccius capensis* and *M. paradoxus* (e.g. Punt and Leslie 1995), work on MPAs (Attwood *et al*. 1997a, b, Beaumont 1997, Hockey and Branch 1997), studies relating seabird survival and population size to availability of food (e.g. Crawford 1998, 1999, 2003, 2004) and research into the development of subsistence fisheries in South Africa (Branch 2002).

The issues

With the exception of serious problems in specific fisheries, such as those for abalone, *Haliotis midae*, and linefish, South Africa's fisheries in the southern Benguela system have generally been well-managed in recent years (Payne and Bannister 2003). The management process has typically included good scientific advice and input, good, although not always complete (e.g. Cochrane *et al.* 1998), participation by stakeholders in decision-making, reasonably effective control, and generally adequate to good levels of compliance. As a result, the target resources in the best-managed fisheries, such as Cape hake, anchovy and sardine are in a healthy condition, whereas West Coast rock lobster *Jasus lalandii* is recovering after a period of poor growth in the second-half of the 1990s (Cockcroft and Payne 1999). However, in all cases, management of fisheries in South Africa has followed the conventional approach of focusing on target resources and assuming that they are, totally or at least to a very large degree, independent of the rest of the ecosystem. Given this successful management, it is pertinent and necessary to consider where, if at all, there is a need for a broader, ecosystem approach to these fisheries, and what benefits would arise from the approach.

This question was put to the participants at a workshop on ecosystem modelling approaches to fisheries management in the southern Benguela, held in Cape Town in December 2002 (Shannon *et al*. 2004). The participants, predominantly scientists, agreed that EAF would be useful for Benguela fisheries and that it could contribute to rebuilding depleted stocks, would encourage the consideration of wider fisheries effects such as bycatch, and would make better use of existing knowledge of the structure and dynamics of the southern African marine ecosystems, so reducing the risks of management decisions that would lead to major ecological, economic or social crises in the fisheries. Overall, it was considered that implementing an EAF would increase the likelihood of obtaining the optimal benefits from the fishery sector in a sustainable manner.

The workshop also listed several issues as of particular importance within an EAF regime in the southern Benguela. These included:

- the possibility of degradation of the benthic habitat through fishing and mining;
- environmental influences on fish abundance and production;
- operational interactions, including bycatch, discarding and conflict between different gear users (e.g. longlines and trawls);
- the need to consider spatial issues, such as the movements and distribution of resources in relation to those of fishing effort;
- foodweb interactions, such as dependence of seabirds on pelagic forage species, taking account of cannibalism in hake, and the importance of currently underutilized resources as prey and the likely impact of increasing fishing mortality on such species.

In the following section, each of the major fisheries is considered in terms of current status, ecosystem interactions and possibilities for ecosystem approaches to their management.

The pelagic fishery

This fishery in the southern Benguela (Fig. 3) uses purse-seines and targets sardine and anchovy in particular, but also catches substantial quantities of round herring *Etrumeus whiteheadii* and Cape horse mackerel *Trachurus trachurus capensis*.

Bycatch of horse mackerel and sardine during anchovy fishing are important problems in the pelagic fishery, but attempts are being made to reduce these and to determine optimal approaches that recognize these interactions. Total Allowable Catches (*TAC*s) for both anchovy and sardine are set according to management procedures that make allowance for the bycatch of juvenile sardine in the anchovy fishery (De Oliveira *et al.* 1998, De Oliveira, 2003). The approach has encountered problems, particularly in identifying compromise solutions that would be accepted by both the sardine and anchovy fishing groups, but good progress has been made in implementing a management strategy for sustainable use (Cochrane *et al.* 1998, De Oliveira *et al.* 1998, De Oliveira 2003, Cunningham and Butterworth 2004).

Juvenile horse mackerel are also caught as bycatch in the pelagic fishery, with potentially negative im-

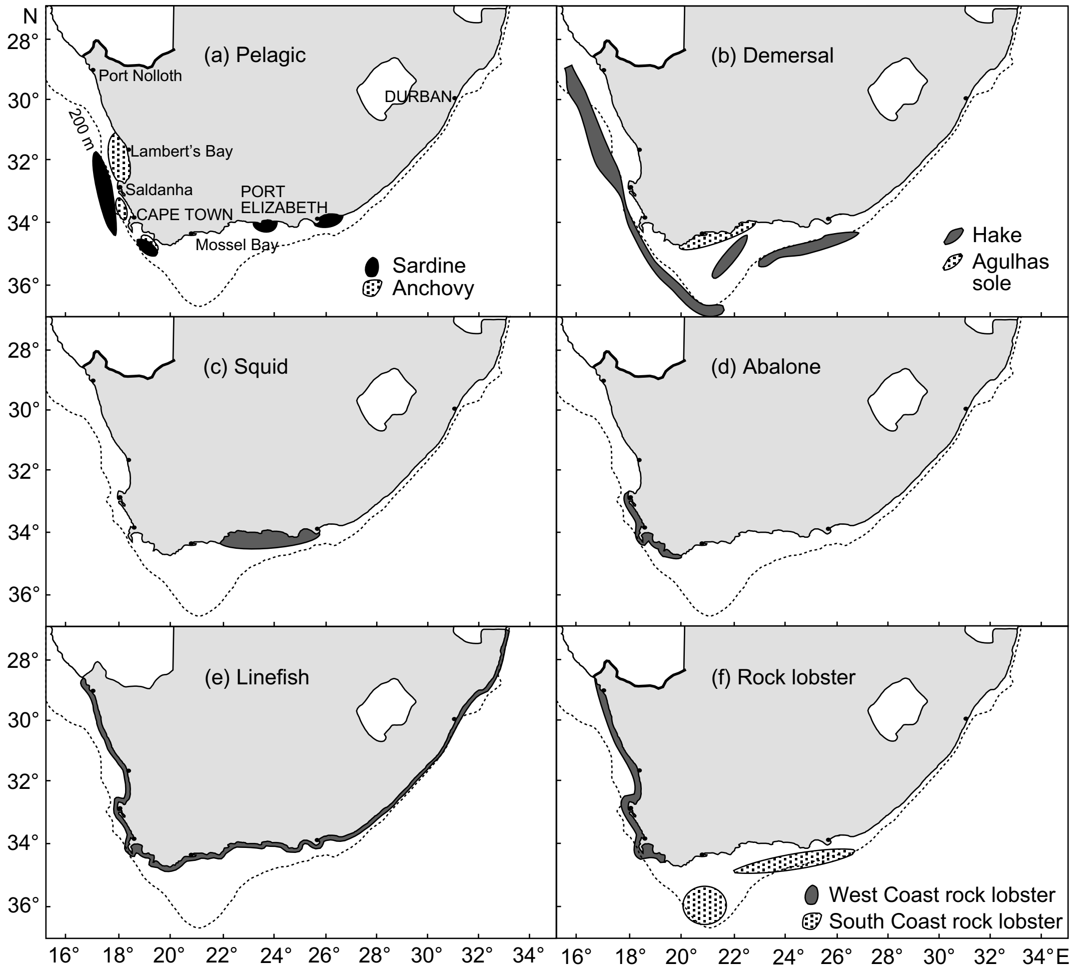

Fig. 3: The fishing grounds for the major fisheries and resources in the southern Benguela

pacts on the fishery for adult horse mackerel. The fishery for horse mackerel is managed primarily by a catch limit on adults, but this has rarely been fully subscribed in recent years, and the species is considered to be under-utilized. However, since 1997, efforts have been made to encourage better use of the resource and, in 2001, fishing rights for horse mackerel in South Africa were allocated to 18 companies as a part of a plan to develop the fishery (Anon. 2002). This was accompanied by setting a maximum permitted catch level for adults, and also a precautionary upper catch limit of 5 000 tons of juveniles in the pelagic fishery (Leslie 2001). The precautionary limit was increased to 7 500 tons in 2002, during which year the pelagic fishery was closed for three weeks when this limit was exceeded. The strategy has not yet been in operation long enough to evaluate its performance, but it is consistent with the concept of an EAF.

A technical interaction is identified when one fishery using a particular technology impacts another fishery that usually uses a different technology but exploits the same resources as target or bycatch. While the strongest technical interactions are being addressed within the pelagic fishery, the management strategies for the pelagic fishery do not take into account the biological interactions among the species. In the case of anchovy and sardine, competition as well as cannibalism and predation on eggs (Valdés-Szeinfeld and Cochrane

1992) are considered to be important, although there is doubt about the extent of actual competition between the two species (Shackleton 1987, Louw *et al.* 1998, Van der Lingen 2002). Some measure of competition is also likely with juvenile horse mackerel and with round herring. These biological interactions, particularly competition for food, could lead to the existence of an upper limit, or carrying capacity, for the combined populations of the small pelagic species in the southern Benguela, rather than independent carrying capacities for each. In the northern Benguela, the impact of competition between the currently dominant horse mackerel population and other pelagic species has been identified as a priority concern (Roux and Shannon 2004).

While an upper limit on the biomass of small pelagic fish should in theory exist, trends in population abundance over the last two decades have indicated how variable this limit can be. Biomass surveys undertaken by Marine and Coastal Management (MCM) since the mid-1990s recorded a peak of nearly 3 million tons of anchovy, sardine and round herring combined in 1991 and 1992. This biomass fell to <2 million tons in 1994, driven largely by a collapse in anchovy biomass, but subsequently increased to nearly 7.5 million tons in 2001 as a result of increases in both the anchovy and sardine biomasses. Such variability and uncertainty will severely hinder any attempt to manage the fishery for small pelagic fish on the basis of a theoretical upper limit.

The small pelagic fish are important prey for a large number of predators. Cury *et al.* (2000) suggested that small pelagic fish in upwelling ecosystems exert both top-down control on their invertebrate prey and bottom-up control on their predators, a trophic role they described as "wasp-waist". The predators on small pelagics in the southern Benguela include Cape hake, snoek *Thyrsites atun* and other linefish, tuna, a number of shark species, seabirds and Cape fur seals. The bottom-up control postulated by Cury *et al.* (2000) implies that any impacts by a fishery on one or more of the small pelagic species, for example a fishing-induced change in abundance, will impact other dependent and affected species in the system (e.g. Crawford 1998). For example, Wolfaardt *et al.* (2001) reported an increase of 14% in the number of breeding pairs of African penguins *Spheniscus demersus* in South Africa between 2000 and 2001, despite the negative impact on the penguin population of a major oil spill on the west coast of South Africa in June 2000. The authors attributed this increase to the abundance of the two prey species.

The importance of the small pelagic fish as prey also raises the question of whether or not a reduction in fishing mortality on them could lead to increased catches of hake and some important linefish species, as well as improvements in the status of species of conservation interest such as seabirds and Bryde's whale *Balaenoptera edenii*. The latter could have positive economic implications through ecotourism. However, the high levels of uncertainty in estimates of biological interactions and ecosystem responses to management are obstacles to developing an effective ecosystem strategy, particularly because such a strategy would almost certainly have marked short-term negative social and economic impacts and could also have unexpected negative ecological impacts. Rigorous ecological modelling may provide some measure of information on the ecological ramifications and feasibility of such strategies, but the limitations of existing ecological understanding and models to provide reliable scientific advice to managers and policy-makers must always be considered when developing strategies based, particularly, on uncertain biological interactions.

Demersal fishery

This fishery has traditionally used bottom trawls, but recently demersal longline and handline fisheries have developed. All these gear types are likely to have some detrimental ecosystem effects. Bottom trawls can cause substratum damage (Sainsbury *et al.* 1997, Hall 1999) and linefisheries can be associated with, for example, seabird mortality and ghost-fishing (Alexander *et al.* 1997, Ryan and Boix-Hinzen 1998).

Catches in the demersal trawl fishery of the southern Benguela fall into three main groups: the major target species Agulhas sole *Austroglossus pectoralis* and Cape hake (the shallow-water *M. capensis* and the deep-water *M. paradoxus*); commercially important bycatch species such as kingklip *Genypterus capensis*, monkfish *Lophius vomerinus*, and snoek; and non-commercial bycatch species such as grenadiers *Coelorinchus symorhynchus, C. braueri* and *Malacocephalus laevis*. The trawl fleet can be divided into two main categories: the inshore trawl fishery, based on the South-East Coast, which targets mainly shallow-water hake and Agulhas sole; and the deep-sea hake fishery that targets mainly deep-water hake, which make up as much as 90% of their catches (Fig. 3).

The handline fishery operates along the South-East Coast, and the longline fishery is divided roughly evenly between the South-East and West coasts. The hand- and longline fisheries target mainly shallow-water hake on the South-East Coast, and both hake species on the West Coast.

As with the pelagic fishery, there are interactions that warrant immediate attention to determine their extent and impact, and to identify means to address any urgent problems. Probably the most important of these relate to discarding and bycatch. During a pilot observer programme conducted between 1995 and 2000, the number of species recorded in commercial catches was 56 (sole-directed trawls), 63 (South-East Coast hake-directed trawls) and 74 on the West Coast (Walmsley *et al.* in prep.). Those authors estimated that, in 1997, 9 000–10 000 tons of fish were discarded on the South-East Coast and 17 000–25 000 tons on the West Coast, compared with total landings of some 160 000 tons.

Several linefish species, for example silver kob *Argyrosomus inodorus*, carpenter *Argyrozona argyrozona* and white stumpnose *Rhabdosargus globiceps*, are taken as bycatch in the hake/sole-directed inshore trawling fleet. As a result, it has often been alleged by linefishers that inshore trawlers are responsible for declines in catches of linefish resources. Information on the number of fish and the size distribution of linefish species both landed and discarded by the inshore trawling fleet was collected between 1994 and 2000 as part of an observer programme, and are being analysed.

The demersal longline fishery has the potential to take an appreciable bycatch of kingklip. An experimental kingklip-directed longline fishery (1986–1989) was closed because of overexploitation, and the South Coast component of the kingklip resource is still considered to be depleted (Mori and Butterworth 2002). Consequently, the kingklip longline bycatch is limited to 15% of the hake catch per landing. In addition, the mortality rate of white-chinned petrels *Procellaria aequinoctialis* during the initial period of the longline fishery was well above sustainable levels (Barnes *et al.* 1997). Another serious concern with the longline fishery is unrecorded mortality through lost catches, both through theft by seals and through fish breaking off the lines. The latter is particularly serious, because fish can break-off and surface as many as 2 miles away from the vessel in moderately rough seas (R.W. Leslie, MCM, pers. comm.), where they are not visible to the fishers. It is therefore difficult to convince fishers of the real magnitude of the unrecorded mortality.

Conflicts between different gear users, such as between the trawl and longline fisheries, which target hake and kinglip, also requires a broader approach to management than would be necessary in a single fishery isolated from interactions with others.

Biological interactions take place between the target species of the demersal fishery, with target species of other fisheries and also with other species of lesser or no commercial value. The diet of the two Cape hake species includes shallow-water and deep-water hakes, sardine, anchovy, round herring, chub mackerel *Scomber japonicus*, chokka squid and many other fish, crustaceans and molluscs (Payne *et al.* 1987b, Punt *et al.* 1992, Pillar and Wilkinson 1995). Punt and Leslie (1995) described cannibalism of small hake by larger hake in both hake species. This cannibalism influences the estimated response of the hake populations to a possible cull of seals (Punt and Butterworth 1995), as discussed later. Such interactions could adversely affect the performance of current single-species management strategies, a possibility that should be considered using the best science available.

Squid

The South African squid fishery is based on chokka squid, which inhabit the shelf region mainly between Cape Town and East London (Fig. 3). The fishery uses hand jigs on natural spawning concentrations inshore, as well as on artificial aggregations, created using powerful lights, offshore. The fishery does not take bycatch or damage the substratum**,** except for some damage done to squid eggs by dragged anchors (Sauer 1995). There is also the possibility of some disturbance to squid behaviour caused by the lights. A closed season has been instituted in the fishery to protect spawners at a vulnerable stage, and to act as a brake on effort (Augustyn *et al.* 1992).

The primary considerations in the squid fishery should therefore be an evaluation of the biological interactions between squid and the ecosystem, and their implications for maximizing sustainable yields of the fisheries affected by those interactions. Chokka squid feed on some commercial species, including anchovy and Cape hake, but the total consumption and its potential impact on the populations of prey species have not yet been quantified. A preliminary evaluation was done by Lipiński (1992), who concluded that chokka is the only cephalopod species impacting as a predator on commercial species of fish on the southern African shelf, but that its impact upon stocks of anchovy and other pelagics is likely to be relatively small. Adult chokka are opportunistic predators and will easily switch to abundant and energetically suitable prey. However, chokka are likely to be important as prey for other marine organisms. Prior to the impacts of fisheries on the stocks, juvenile chokka, which live in the epipelagic zone of inshore waters, were the prey of many linefish, chondrichthyan and mammal species, many of which have been substantially reduced in abundance by fisheries. It is possible that, with the reductions in many natural predators of chokka, the abundance of squid may have increased. These foodweb considerations may be important for the management of chokka resources but, because of the short life cycle, fast turnover rate and time, opportunistic behaviour as a predator, and imperfect knowledge about foodweb links, quantification of these interactions will be difficult.

Of more immediate concern for sustainable utilization of chokka squid are environmental, socio-economic and effort-management considerations. For example, recent construction of the Coega Port near Port Elizabeth is likely to have negative impacts on traditional squid spawning grounds near there. Quantification of this impact is needed. The socio-economic significance of the various tools of effort regulation is also a matter for concern. In particular, it is intended to investigate in more detail all aspects of the link between the length of the closed season and *ad hoc* increases in fishing effort.

Abalone

The commercial fishery for abalone takes place in the Western Cape between Cape Columbine and Quoin Point (Fig. 3), although the population extends farther north and east of this region (Griffiths and Branch 1999). Fishing is from small boats, mostly in water shallower than 10 m. Divers use hookah breathing apparatus and a blunt instrument to collect abalone, so there is negligible bycatch or damage to the rocky substratum. Biological interactions may be more important. It has been estimated that abalone at an unexploited site consumed up to 23% of algal production, although much of this is in the form of detached algal fragments (Barkai and Griffiths 1988).

One biological interaction that may need to be addressed in future is that between abalone and West Coast rock lobster. In the early 1990s, lobsters moved in significant numbers into a part of the distributional range of abalone (Tarr *et al.* 1996). Lobsters feed on sea urchins *Parechinus angulosus*, and the invaders soon caused a substantial reduction in numbers of this prey. Because sea urchins provide important shelter for juvenile abalone, and also promote environmental conditions that favour survival of abalone recruits (Day and Branch 2000), it is probable that their reduction in abundance in the affected areas exposed the young abalone to predation by the lobsters and other predators, with negative impacts on the reproductive success of the abalone (Mayfield and Branch 2000, Tarr and MacKenzie 2002). Diving surveys in the affected areas (Cape Hangklip to Hermanus) have confirmed the virtual absence of juvenile abalone (MCM unpublished data), and it appears that there has been sustained recruitment failure since the early 1990s as a direct result of the "rock lobster effect". Model projections now indicate that, even in the absence of poaching, which is rife (Hauck and Sweijd 1999), this region cannot sustain further abalone fishing. Given that approximately 35% of the commercial fishery yield has emanated from this region, the losses attributable to ecosystem changes will be substantial. A further effect of this change was to nullify the management benefits (for the abalone fishery) of a marine protected area that lay within the "rock lobster area". This has required that a new abalone harvest refugium be adopted farther east, outside the rock lobster area.

Whereas the causes of the movement of rock lobsters into the region are unknown, it appears that it may have resulted in an alternative stable state, where rock lobsters are now the dominant inshore predators. The rock lobsters have modified the previous ecological balance of the inshore kelp *Ecklonia maxima* forests, where there are now few small invertebrate prey species (R. J. Q. Tarr, MCM, unpublished data).

Crustaceans

Earliest records of human exploitation of the West Coast rock lobster, based on remains found in Khoi-San caves, date back to the early Holocene, some 10 000 years ago. Commercial exploitation commenced in the late 19th century and expanded during the early 20th century. The current South African crustacean fisheries can be divided into three major sectors: a multispecies crustacean trawl fishery targeting prawns, langoustines, crabs and deep-water rock lobsters *Paninurus delagoae* off KwaZulu-Natal; an inshore (5–100 m) trap and hoopnet fishery for West Coast rock lobster extending from the Namibian border southwards, around Cape Point to Danger Point just south-east of Cape Town (Fig. 3); and a deep-water (50–200 m) trap-fishery for South Coast rock lobster *Palinurus gilchristi* on the Agulhas Bank and eastwards to East London. Only the latter two fisheries are considered here, because they are within the southern Benguela ecosystem (Fig. 3).

WEST COAST ROCK LOBSTER

This fishery was initially based on the use of hand-hauled baited hoopnets, but baited traps now account for some 75% of the total catch. Both hoopnets and traps are highly species-specific, with minimal bycatch of fish, octopus and other invertebrates. Management measures that ensure this are: (i) only traps and hoopnets covered with legislated size mesh are permitted; and (ii) deck grid-sorters, which allow the immediate return of undersize lobster and by default any small invertebrate bycatch, are compulsory aboard all trap vessels. Rock lobster traps are large and heavy, and some damage to the benthos, largely during deployment and retrieval, is inevitable. However, the damage to the benthic community is localized in area and limited in extent; recovery in the subtidal areas of the West Coast is considered to be rapid (A. C. Cockcroft, MCM, pers. obs.). Management on an areal and zonal basis, which ensures the distribution of fishing effort over the lobster fishing grounds, reduces the risk of severe localized impact of traps. Designated closed areas and rock lobster reserves provide further protection for both rock lobster and associated benthic communities. When viewed in the light of the extensive faunal mortalities caused by naturally occurring phenomena such as low-oxygen events, black tides and toxic algal blooms (Cockcroft *et al*. 1999), the localized disturbance by traps is small.

J. lalandii is a keystone species in the nearshore rocky subtidal areas of the West Coast, and it shapes community structure via predation on a wide range of benthic organisms (Mayfield *et al*. 2000). Relatively abundant rock lobsters can lead to a reduction in density, or even elimination, of black mussel *Choromytilus meriodonalis*, the preferred prey of the species, and alter the size structure of populations of ribbed mussels *Aulacomya ater*, reducing the proportion of selected size-classes (Griffiths and Seiderer 1980, Griffiths and Branch 1999). Comparative and experimental studies have demonstrated that their role as predators can reshape benthic communities, resulting in large reductions in taxa such as black mussels, urchins, whelks and barnacles and in the dominance of algae (Barkai and Branch 1988a, b). There is evidence of two stable states in the community structure of benthic, coastal ecosystems in the Benguela, one dominated by *J. lalandii* and the other in which the species is rare (Barkai and Branch 1988a, Griffiths and Branch 1999).

Recent assessments estimate the exploitable biomass (lobsters >75 mm carapace length [*CL*]) of the species to be around 10% of its pristine value, with spawning biomass (females >65 mm *CL*) estimated at around 21% (Johnston 1998). The large reduction in biomass is likely to have had a substantial impact on the ecosystem. However, the influences of any ecosystem effects of fishing on community structure are complicated and obscured by the impacts of poorly understood large-scale environmental events that have taken place over the past decade or so. These have probably resulted in the reduction of rock lobster somatic growth rates (Pollock *et al*. 1997), increased occurrence and severity of low-oxygen induced rock lobster walkouts (Cockcroft 2001), a shift in rock lobster abundance to the more southern fishing areas, and the movement of rock lobsters into areas that did not have high densities a decade or so ago with concurrent lobster/urchin/abalone interactions (Tarr *et al*. 1996). Despite the current lack of clear indicators of environmental conditions and community structure indicators that could be used directly in management, a move to an ecosystem approach is reflected in the resource rebuilding strategy that has been incorporated into the OMP used to set the *TAC* for the species (Johnston and Butterworth 2003). The OMP uses both fisheries-dependent and -independent data as well somatic growth rates, which reflect a composite picture of environmental conditions experienced by lobsters.

SOUTH COAST ROCK LOBSTER

This fishery started in 1974, and recent assessments estimate its biomass to be between 20 and 30% of pristine (Johnston and Butterworth 2001). The fishery is assessed annually, based on commercial catch rates, and a combined *TAC* and total allowable effort (*TAE*) management strategy is used. *P. gilchristi* does not occur in dense concentrations like *J. lalandii*, and the fishing gear used is therefore longlines, each fitted with 100–200 lightweight plastic traps. The ecological effect of reducing the abundance of *P. gilchristi* by means of fishing has not yet been investigated. A gear-related effect is increased predation on *P. gilchristi* by octopus *Octopus magnificus* (J. C. Groeneveld, MCM, pers. obs.). Octopus, a bycatch in the fishery, opportunistically follow rock lobsters into traps, where they present an easy prey. Ghost-fishing is not considered a major problem, because traps that are retrieved after long periods at sea are invariably empty. Seasonal recruitment of *P. gilchristi* juveniles to the western Agulhas Bank is followed by a long-distance migration of up to 800 km eastwards (Groeneveld and Branch 2002). Targeting these migrants results in very high catch rates, but may affect adult recruitment eastwards to Port Elizabeth. Low numbers of slipper lobster *Scyllarides elisabethae* appear in traps in the eastern part of the fishery.

Small invertebrate fisheries

The next most important fisheries for invertebrates in South Africa, after those for rock lobsters and abalone, are the fisheries for mussels. Recreational and subsistence fisheries for mussels occur along the coast of South Africa, but the combination of high incidence of red tide poisoning and low human population density along the west coast of the country, where mussels are most abundant, have meant that there is no commercial fishery on wild stocks (Griffiths and Branch 1999). There is, however, a well-developed aquaculture industry based on the introduced Mediterranean mussel *Mytilus galloprovincialis* in Saldanha Bay (Fig. 3). The only other commercial fishery for coastal invertebrates is that for Cape rock oysters *Striostrea margaritacea*. An experimental fishery for the common octopus *Octopus vulgaris* has also recently been initiated.

THE OYSTER FISHERY

A commercial fishery for the Cape rock oyster along the south coast of South Africa commenced in the late 19th century (Stander 1991). The fishery is small, with about 27 participants holding limited commercial rights and one holding a full commercial right, employing a total of 76 oyster-pickers.

Oysters are dislodged from rocks in the surf-zone during spring low tides by means of a pointed steel bar (crowbar). Although effective, this method of harvesting can result in the incidental mortality of juvenile oysters and non-target intertidal organisms. Oysters are believed to be a keystone species in reef and estuarine ecosystems (Hargis and Haven 1988, Mann *et al*. 1991, Ulanowicz and Tuttle 1992). The complex relationships between black mussels and the Mediterranean mussel, red bait *Pyura stolonifera* and oysters are poorly understood, and the effects of oyster removal on the South Coast reef community structure have not been assessed.

Management measures designed to mitigate the impacts of oyster harvesting include a limit on effort (number of harvesters) and annual catch (60 000 oysters per limited commercial right-holder). Gear restrictions include a minimum length limit on the oyster pick (protects oysters living in crevices or cracks and under rocks) and a ban on the use of diving fins and artificial breathing apparatus, which protects subtidal oyster beds. Closed areas in the vicinity (Tsitsikamma, Goukamma and De Hoop Nature Reserves) provide further protection. Substantial research is required to assist in developing understanding of the interactions between benthic community structure, environmental changes and oyster harvesting.

THE OCTOPUS EXPERIMENTAL FISHERY

The common octopus is considered to be an underexploited resource in South Africa. At present it is only harvested in the intertidal zone by recreational and subsistence fishers, largely for bait, and to a lesser extent for human consumption. In other parts of the world, it is considered to be a delicacy and can fetch a high price on the international market. A five-year experiment to determine the biological and economic feasibility of harvesting octopus for human consumption commenced in 2003. During the experiment, octopus will be harvested in eight areas off the South African coast using unbaited pots attached to longlines deployed on shelly or coarse sandy bottoms at depths of 10–50 m. Pots soak for a period of about one week at a time.

A precautionary approach towards management of the octopus experimental fishery will be adopted by restricting the number of participants. The gear and harvesting technique prescribed is passive and non-destructive, and it is considered that no bycatch will result (Oosthuizen 2003). Additional measures will include prescribing pot type, size and limiting the number of pots. Furthermore, no fishing will be allowed in marine reserves or closed areas. No catch limits, size restrictions or closed seasons will be imposed during the initial stages, but may be introduced as the experiment progresses. The diet of *O. vulgaris* in South African waters varies according to locality, and ranges from mussels, abalone and crustaceans to fish (Oosthuizen 2003). With available knowledge, the indirect impacts on the ecosystem of a fishery on common octopus would be difficult to estimate. The experimental fishery should supply invaluable information on the biology, ecology and population dynamics of this species, which could form the basis of an ecosystem approach to the management of this potential fishery.

Linefish

Established in the early 1800s, commercial fishing with handlines or rod and reel is arguably one of the oldest fishing industries along the 2 500 km stretch of the southern Benguela coastline between the Orange and Kei rivers (Griffiths 2000). About 40 species have been targeted, of which 20 are considered to be economically important. Traditional linefish consist of demersal and pelagic shelf species, which, based on movement patterns, can be divided into the following categories (Griffiths 2000):

(1) coastal migrants that undertake a seasonal migration to spawn in KwaZulu-Natal, e.g. seventyfour *Polysteganus undulosus* (Ahrens 1964) and geelbek *Atractoscion aequidens* (Griffiths and Hecht 1995);
(2) resident reef-associated fish, including sea bream *Chrysoblephus* spp. with high site fidelity (Griffiths and Wilke 2002), and others such as silver kob with large home ranges (Griffiths 1997); and
(3) pelagic nomads such as snoek and yellowtail *Seriola lalandi,* which, although shelf-dwelling

species, cover large areas in short periods and are less predictably located (Nepgen 1979, Wilke and Griffiths 1999).

With the introduction of tuna pole and large-pelagic longline fisheries in the mid 1980s and late 1990s respectively, a fourth group was added, the oceanic migrants, including tunas, swordfish and pelagic sharks.

THE TRADITIONAL LINEFISHERY

Stock assessments and trends in catch per unit effort (*cpue*) reveal that, with the exception of shelf nomads and oceanic migrants, the other linefish species targeted in the southern Benguela were heavily overexploited during the last century; particularly the warm/temperate populations east of Cape Point (Griffiths 2000). Stock assessments also reveal that several estuarine and surf-zone species, including representatives of the Sparidae (Bennett 1993), Dichistiidae (Bennett 1988) and Sciaenidae (Griffiths 1997), have been depleted by South African recreational shore-anglers. Factors contributing to the demise of South African linefish stocks include unchecked commercial effort, inappropriate recreational regulations, and biological factors such as predictable locality (coastal migrants and resident reef fish), longevity (12–25 years) and late maturity (7–55% of maximum age). Plans to rebuild South African linefish stocks include drastic reductions in commercial effort and the introduction of more stringent bag limits for recreational anglers.

Although there has been no empirical research on the ecosystem effects of linefishing in South Africa, most of the depleted species are important predators of both benthic and pelagic organisms (teleosts and large invertebrates). Their removal is therefore likely to have precipitated substantial changes in the species diversity and the trophic pathways of pelagic and demersal foodwebs of shallow (<100 m) sub-tidal ecosystems (e.g. Hall 1999, Griffiths 2000). Evidence of such changes has been obtained from surveys in the Southern Cape, which revealed that predatory fish were more abundant in a marine reserve (Buxton and Smale 1989), and that there were also differences in the familial composition of cryptic fish on exploited and unexploited reefs (Burger 1990). The exploitation of predators of large invertebrates, including large sparids, was recently shown to change the community structure and reduce both primary and secondary production in a temperate-reef ecosystem off New Zealand (Babcock *et al.* 1999). Marine protected areas, in association with other management measures, have an important role to play in conserving the shallow, subtidal ecosystems, as has been recognized by MCM.

As previously mentioned, sardine and anchovy are important food species of pelagic and several reef-associated linefish. As a result of the seasonal spawning migrations of these two prey species and their importance as food to linefish species, linefish fulfil an important role in biologically pumping energy and carbon from the highly productive West Coast upwelling system onto the eastern seaboard. This process could allow warm/temperate reef ecosystems to support larger shoals of piscivores than the reefs could otherwise sustain. It is therefore not coincidental that the most important reef-associated species have been those that feed on clupeoids, i.e. carpenter, silver kob, geelbek *Atractoscion aequidens* and seventyfour. The extent to which these linefish may have fertilized reefs (nitrogen excretion and faecal pellets) is unclear, but it is possible that their demise has concomitantly reduced the links of reef ecosystems with the pelagic foodweb.

Owing to the dependence of many traditional linefish species (including pelagic nomads) on exploited forage fish, it may be important to ensure that there is sufficient biomass of these prey not only to support current populations, but also to allow overexploited linefish stocks to recover. This may not simply be a case of sufficient biomass, but rather sufficient biomass in inshore areas of the East and West coasts where linefish predators are abundant. Ecosystem modelling could have an important role to play in evaluating the ecosystem effects of overfishing linefish predators as well as the impacts on predators of exploiting prey populations, in particular the clupeoids. Such models would need to account for distribution patterns of predators and prey through spatial components.

SHARKS AND TUNA

The southern African chondrichthyan fauna (sharks, skates, rays and chimaeras) is highly diverse, with approximately 210 species along the 6 400 km coastline (Compagno 1999), consisting of the tropical Indian Ocean fauna on the northern East Coast to the cool water Atlantic fauna of the Benguela. Those on the west coast of southern Africa includes 55 species of cartilaginous fish (Compagno *et al.* 1991), which are taken mainly by trawl. The fauna found deeper than 1 000 m is poorly known.

Despite this richness, chondrichthyans are among the most neglected fisheries groups in southern Africa, in part because there is no traditional high value fishery for them. They are taken as a bycatch in most fisheries. Although some species, such as soupfin shark *Galeorhinus galeus*, have been targeted for decades and there is an increasing interest in other line-caught species such as smoothhounds *Mustelus mustelus* and offshore pelagic sharks such as mako *Isurus oxyrinchus*, the existing databases on catch, effort and discards are inadequate for management purposes (Smale 1997, Japp 1999). In most fisheries databases, sharks and rays are aggregated, preventing detailed investigation of catch trends.

Annual total recorded landings (certainly an underestimate) of chondrichthyans from South African fisheries (trawl, set net, line catches by the KwaZulu-Natal Sharks Board) in the early 1990s were around 2 000 tons (Smale 1997, Japp 1999). This value excludes discards and catches made by illegal nets, such as those on the Cape west coast where targeting of smoothhounds may net up to 20 tons per month during summer from the Saldanha/Langebaan region alone (Hutchings and Lamberth 2002).

Shark mortalities and bycatch of harmless animals from the Sharks Board bather protection set net fishery in KwaZulu-Natal have probably decreased, because live sharks are now released whenever possible (Cliff and Dudley 1992). Furthermore, there has been a decrease from 44 km of nets along the 326 km of coastline in the late 1980s, to 41 km in 1997 (Dudley and Gribble 1999) and the number of nets continues to decrease (G. Cliff, KwaZulu-Natal Sharks Board, pers. comm.). In addition, the removal of bather protection nets from the water during the winter "sardine run" since 1975 (Dudley and Gribble 1999) has markedly decreased catches of shark predators following the sardine schools, particularly bronze whalers *Carcharhinus brachyurus*. This species is caught throughout the year in the Eastern and Western Cape by linefishing, trawling and purse-seining, where they feed predominantly on sardine and squid (Smale 1991). They enter waters of KwaZulu-Natal mainly in the winter "sardine run" (Cliff and Dudley 1992).

There are currently no quotas on commercial catches of chondrichthyans in South Africa, although recreational anglers have a daily bag limit of 10, but may kill or release any number. However, four species may not be exploited commercially, (the ragged-tooth shark *Carcharias taurus*, gully shark *Triakis megalopterus* and two species of catshark *Poroderma africanum* and *P. pantherinum*). The white shark *Carcharodon carcharias* is totally protected from exploitation off South Africa and Namibia. The effectiveness of this and other fisheries legislation is uncertain because of inadequate levels of compliance, monitoring and policing (Japp 1999).

Chondrichthyans commonly exhibit late maturity, relatively slow growth and low reproductive capacity, which makes them particularly vulnerable to overexploitation (Holden 1974, Compagno 1990), whether by directed fisheries or as bycatch. Increasing concern has been expressed on fishery impacts on deep-water species, because life history styles are likely to be similar and stock sizes small, given the lower productivity of deep waters.

In response to the FAO International Plan of Action (IPOA) for the Conservation and Management of Sharks, South Africa has developed a national plan of action. However, it is acknowledged that a lack of human resources for shark research and assessment is hindering implementation of this plan (Tilney 2003). Concern for the inadequacies of management of chondrichthyans has recently been taken up by various non-governmental organizations with an interest in the conservation of sharks (Fowler 1999). This political pressure may contribute towards a more holistic approach to management.

The earliest South African commercial tuna longline catches were recorded from the 1960s (Talbot and Penrith 1968). The vessels used similar gear to that of the Japanese vessels operating in the region, and caught mainly southern bluefin tuna *Thunnus maccoyii*, followed by albacore *T. alalunga* in later years. This fishery was short-lived as increasing numbers of fishers turned their attention to other more lucrative fisheries, and by 1970 longline permits were largely retained only to target demersal sharks (Nepgen 1970). South African fishers became reinterested in tuna longlining after a joint-venture permit issued in 1995 confirmed that tuna and swordfish *Xiphias gladius* could be exploited profitably in South African waters. In response to this interest, South African authorities issued 30 experimental permits in 1997 to target tuna (Penney and Griffiths 1998). The average size of South African longline vessels is small (~30 m), with fishing operations largely limited to the EEZ. Swordfish constitute the bulk of the catch (>1 000 tons landed in 2002). Nominal *cpue* for swordfish has remained stable at about 0.5 kg hook^{-1} over the past three years, a marked decline from 3.4 kg hook^{-1} in 1997 (Kroese 2000). Other species targeted include bigeye *T. obesus* and yellowfin *T. albacares* tuna. Important bycatch species include albacore, blue sharks *Prionace glauca* and mako sharks. Of growing concern is the incidental catches of seabirds and turtles. The fishery is still in the experimental phase until commercial fishing rights are allocated in 2005.

The South African tuna pole fishery developed in 1980 in response to a large run of yellowfin tuna in 1979 (Molteno and Riley 1986). The fishery consists of approximately 150 vessels, which operate along the west coast of South Africa, targeting small to mediumsized albacore during summer and autumn when the species is concentrated nearshore. Nominal *cpue* fluctuates considerably, and largely depends on large-scale environmental conditions, which in turn influences fish distribution. Other species landed by poling vessels include yellowfin and bigeye tuna. Along the east coast of South Africa, small yellowfin tuna are targeted by trolling.

Because large pelagic species are highly migratory, domestic catches and catch rates have to be viewed in a regional context. To date, billfish, tuna and pelagic shark species in South African waters are considered to be from both Atlantic and Indian Ocean stocks – "separated" at 20°E. Large pelagic stocks to the west, most relevant to this paper, are managed by the International Commission for the Conservation of Atlantic Tunas (ICCAT), and to the east by the Indian Ocean Tuna Commission (IOTC). The exception to this is southern bluefin, which are managed by the Commission for the Conservation of Southern Bluefin Tuna (CCSBT).

Tunas consume mainly small pelagic fish and to a lesser extent squid and crustaceans (Nepgen 1970). Swordfish have a more varied diet, which includes squid, pelagic and deep-water demersal fish, and invertebrates (Ward and Elscot 2000). Blue and mako sharks are more opportunistic feeders and consume small tunas, billfish, sharks, pelagic fish, squid and turtles (Compagno and Smale 1995).

There is international concern regarding the high fishing mortality of blue and mako sharks. Sharks are particularly susceptible to overfishing because they are slow-growing, mature late and have a low fecundity. Furthermore, if these apex predators are overexploited, it could cause a proliferation of smaller shark species (e.g. dusky sharks *Carcharhinus obscurus*). In turn, a high biomass of small sharks could reduce the numbers of squid and small pelagic fish, which tuna and swordfish consume. This ecological interaction is particularly important when considering the female swordfish of the South Atlantic and Indian Ocean stocks, which utilize South African waters as feeding grounds.

BYCATCH OF SEABIRDS

Pelagic longlining, by virtue of the size of international fleets and the methods employed, is responsible for the deaths of thousands of albatrosses and petrels each year (Petersen *et al*. 2003). This problem led to the development of and agreement on an IPOA for Reducing Incidental Catch of Seabirds in Longline Fisheries by FAO-member countries. During the mid-1990s, steps were taken in the hake longline fishery to reduce incidental bycatch of seabirds and these led to a significant drop in such catches (Ryan and Boix-Hinzen 1998). Subsequent studies have shown good compliance with the regulations, but also that some bycatch still occurs, with white-chinned petrels being the species most commonly caught (Osborne and Mullins 2001). The seabird bycatch of the South African domestic pelagic longlining fleet averages 0.34 birds per thousand hooks, which, although much lower than the 2.64 birds per thousand hooks recorded for distant-water vessels fishing off South Africa, is still unacceptably high (Petersen *et al*. 2003). Of particular concern is the fact that 70–80% of the seabirds caught during pelagic longlining within the South African EEZ are albatrosses. Observer coverage indicates that mandatory bird-scaring lines were employed in less than 30% of longline sets made by South African vessels (Petersen *et al.* 2003). Clearly, progress is being made in addressing this problem, but additional efforts are still required.

Seals

The "fishery" for the skins, meat and oil of Cape fur seals *Arctocephalus pusillus pusillus* is one of the oldest in the southern Benguela, having existed for more than four centuries (Skead 1980). Seal harvesting commenced with the arrival of the early sailing ships before 1600 and, although stopped by the South African Government in 1990, still continues in Namibia. The early exploitation of seals was completely uncontrolled. This resulted in adults and young of both sexes being killed at all times of the year, including during the breeding season. The impact of this exploitation became evident through the extinction of at least 23 island breeding colonies (Shaughnessy 1984) and the marked decline in the population of seals in Cape waters. This prompted the Government of the Cape Colony to grant the first legal protection for seals in 1893 (Shaughnessy 1984). Since then the population has recovered to its current level of about 1.5–2 million animals in the combined southern and northern Benguela Current (Butterworth *et al.* 1995).

Cape fur seals prey on many of the species targeted by the commercial fisheries, such as the two species of hake, sardine, anchovy, horse mackerel, snoek and squid (Rand 1959, David 1987, Lipiński and David 1990, Punt *et al*. 1995). This has led to complaints from sections of the industry that seals reduce stocks of commercial fish, steal their catch, damage their nets and interfere with fishing operations. Wickens *et al.* (1992) reported that seals interact operationally with most fisheries in South Africa through consumption of catches, damage to fishing gear or interference with fishing operations. They estimated that the most significant interactions, in financial terms, included consumption of catches in the demersal longline and the handline fisheries, and damage to gear and disturbance in the trawl fishery. Consumption of catches and operational disturbances by seals in the purse-seine fishery were also recorded. Those authors reported that the estimated financial losses to the fisheries by these interactions were small, in the region of 0.3% of the wholesale value of the landings. They also noted that seals suffered some mortality from the interactions.

Calls for reducing the number of seals have been resisted by the South African Government under advice from its marine scientists, owing to the uncertainties in the argument resulting from the complex and dynamic nature of the marine foodweb. One example of this complexity involves the cannibalistic interactions of the two species of Cape hakes and predation by seals. A mathematical model of these interactions revealed that the net result of culling seals was not intuitively obvious. In some scenarios, reducing predation by seals on hake actually resulted in more inter-hake cannibalism and hence fewer hake overall (Punt and Butterworth 1995). The model was a minimal realistic model that included the two species of hake, the Cape fur seal and a model group to represent other predatory fish.

Other ecosystem interactions of seals involve their predation on threatened populations of seabirds such as the African penguin, Cape gannet *Morus capensis* and Cape, bank and crowned cormorants *Phalacrocorax capensis, P. neglectus* and *P. coronatus* respectively. All these are Red Data Book species, because of large decreases in their populations (BirdLife International 2000, Barnes 2000). Seals not only kill adults and chicks, but also encroach on the breeding areas of penguins, gannets and cormorants on certain islands. Acknowledging the poor conservation status of these seabirds, the South African Government took the decision to control seals in specific cases where the individuals responsible for killing seabirds can be identified. To this end, 153 seals, which were seen to kill fledgling gannets, have so far been culled at Malgas Island off the Western Cape (David *et al.* 2003). Other methods employed to protect seabirds are through the prevention or control of seals invading the seabird breeding areas. This has been accomplished successfully at Mercury Island, off the coast of Namibia, through the deliberate and systematic disturbance of the seals, until they eventually vacated the island and moved to the nearby mainland (Crawford *et al.* 1989, 1994). A similar process of disturbance is currently being pursued at other seabird colonies.

THE NEED FOR EAF

The brief review above demonstrates that none of the fisheries in South Africa operate without some direct and indirect interactions with other fisheries, resources or components of the ecosystem. The interactions range from direct ones between, for example, the different sectors of the hake fishery or the anchovy and sardine fishery, to indirect interactions through the foodweb that would be difficult, if not impossible, to estimate with existing knowledge. Examples of the latter include the impacts of biological interactions between the different species in the linefishery, and the complex direct and indirect interactions between the hake fishery and other dependent and affected species.

South Africa has built a very good foundation of knowledge and understanding of ecosystem dynamics and some important inter-specific interactions. In addition, there have also been advances in implementing EAF, for example, the progress in the establishment of MPAs, progress towards implementation of the international plans of action on seabirds, and in the joint management of the anchovy and sardine fishery. Despite this, much remains to be done if South African fisheries management is to ensure that the fisheries of the southern Benguela are not leading to significant undesirable and potentially irreversible changes to the affected ecosystems. The demands in human and financial resources for full implementation of EAF in South Africa are large and almost certainly exceed the financial and human capacity currently available to meet them. Retaining sufficient scientific expertise in the region is already a major challenge and will be even more important in the implementation of an EAF. The FAO Guidelines recognize that an incremental approach will usually have to be followed, and state "EAF is neither inconsistent with nor a replacement for current fisheries management approaches…, and is likely to be adopted as an incremental extension of current fisheries management approaches" (FAO 2003, p. 7).

The first task for the management agency in consultation with the range of stakeholders in fisheries is to identify the primary problems, issues and needs related to EAF within the existing management strategies for the different fisheries and the other impacting activities. Thereafter, key, feasible objectives will need to be developed. These should emerge from and be consistent with the Marine Living Resources Act (Anon. 1998). Once these objectives have been agreed, it will be necessary to prioritize them, taking due account of their ecological and socio-economic importance and potential benefits, and the costs of implementing the necessary changes. This set of prioritized objectives could form the basis for the Ecosystem Sector Plans that will need to be developed to guide implementation. The review would also indicate which objectives could be addressed in the short-term, with available knowledge and resources, and which will require greater knowledge and capacity before they can be addressed. However, in the planning process, due account should also be given to the need for a precautionary approach, so as to minimize the risks, as far as is practical, of causing major damage to stocks and ecosystems. In accordance with the precautionary approach and the Code of Conduct, "the absence of scientific information should not be used as a reason for postponing or failing to take conservation and management measures" (Paragraph 7.5.1, FAO 1995, p. 12).

The Cape Town workshop made a start in identifying objectives and priorities, but it recognized that these are ultimately a societal choice and the final decisions will require participation and support from the key, legitimate stakeholders, guided by the Marine Living Resources Act and other relevant legislation. The group at the workshop also recommended a separate broad management plan (i.e. Ecosystem Sector Plan) that would list the bioregions/ecosystems within South Africa's borders, and include appropriate time frames (3–4 years) and reference points that should be informed by appropriate models (Shannon *et al.* 2004). The group noted that the Marine Living Resources Act and other relevant acts, international agreements and conventions would need to be taken into account in developing an Ecosystem Sector Plan. It was agreed that new fisheries (e.g. octopus, East Coast abalone, tuna longlining) must be subjected to an ecosystem approach and it was felt by the group that this is one way of initiating the process of implementing an EAF in South Africa.

THE SOCIAL AND ECONOMIC IMPLICATIONS OF EAF

This paper has intentionally focused on the ecological and operational interactions within the southern Benguela that result in the need for an ecosystem approach to fisheries in South Africa. Fisheries exist to meet social and economic needs and goals, and the implementation of any management action is likely to have impacts on the social and economic costs and benefits of the fishery. The implementation of EAF, even in an evolutionary manner, will almost definitely incur both costs and benefits. Moving towards reconciling conflicting objectives and practices in the fisheries of the southern Benguela is going to require addressing trade-offs between different users and user groups. Costs could be incurred through, for example, requirements for changes in gear, fishing methods or areas, or reductions in effort. In some cases, reductions in effort may be required from one user group in order to address negative impacts on another. This will effectively require policy decisions on allocations.

Commercial marine fisheries in South Africa account for <1% of the gross domestic product of South Africa, but nevertheless produce annual landings worth approximately R10 billion and employ almost 30 000 people (Cochrane and Payne 1998, Anon. 2002). Among the socio-economically most important fisheries within the country are those targeting primarily Cape hake, small pelagic species and West Coast rock lobster. In addition, there are significant subsistence fisheries in the country and Clark *et al.* (2002) identified about 147 fishery communities that included nearly 30 000 subsistence fishers and fisher households. These subsistence fishers operate predominantly on the East Coast, so do not use resources from the Benguela ecosystem. Nevertheless, there are subsistence fishers on the West Coast, exploiting mainly nearshore resources, including fish, lobster and abalone (Clark *et al.* 2002).

There are also other users of the living marine resources of the Benguela, including recreational fishers, tourism including eco-tourism, and conservation groups. In addition, other uses, such as the oil and gas industry, marine mining, and coastal-zone development also impact upon the ecosystem. The social and economic aspects of all of these activities will need to be considered alongside the ecological implications, and will have to be addressed if long-term solutions to the sustainable use of the Benguela ecosystem and resources are to be identified and implemented. This will require comprehensive consultation and participation of all legitimate stakeholders.

THREATS TO IMPLEMENTATION OF EAF

Throughout the world, stakeholders and governments are struggling to improve their management and utilization of living marine resources, so as to ensure that optimal benefits can be obtained from them in a sustainable manner. The implementation of EAF will make additional demands on already stretched political and social will, capacity and resources. Success in implementation is by no means guaranteed.

The FAO Guidelines recognize that one major problem is that, to date, there has been very little experience in implementing EAF in practice. There are therefore few case studies, successes or failures, from which lessons can be learned and which can be used as examples for others. There is also scepticism, or caution, about the immediate need for EAF, and its implications for management of the high priority target species. For example, in the 2001 report of the National Marine Fisheries Service of the United States of America, it was stated that: "While minor stocks may be important in an ecosystem context, they are not the primary target species of directed fisheries. Therefore, due to funding constraints and other management concerns, these stocks cannot be given the same level of priority that targeted fisheries must be given..." (NMFS 2002, p. 2).

Emphasis on selected species, rather than on the ecosystem as a whole, does not come only from concerns about socio-economically important species. Conservation concerns also tend to focus on particular species, especially those considered most endangered or at risk of extinction. For example, in terms of the legally binding regulations of the Convention for the Conservation of Endangered Species of Wild Fauna and Flora (CITES), the export of any specimen of a species included in Appendix II of CITES shall require the prior grant and presentation of an export permit. The permit must certify that the exported product has been obtained legally and that the export will not be detrimental to the species concerned (Article IV of the Convention text, http://www.cites.org/eng/disc/text.shtml#IV). The implications of this are that export of an Appendix II species will require all the monitoring, assessment and management commonly associated with species of high economic value.

The FAO Guidelines identify a mismatch between expectations and financial resources as another threat. Societal demand for improved approaches to utilization and management of living marine resources, which implicitly includes EAF, is justifiably high and is growing stronger. However, there is little indication of an increase in the availability of human and financial resources to facilitate its implementation. The ability of many mandated management agencies to meet the expectations of EAF with the human and financial resources currently available must be questionable. Unless this problem is addressed, the risk of failure is likely to be high.

Arguably, the greatest threat to successful implementation of EAF will be the problems associated with reconciling the conflicting demands of different stakeholders. FAO (2003) suggested that, in some cases, differences will not be reconcilable, and high-level intervention will be required to make the final decisions. Participants at the Cape Town workshop identified a number of probable conflicts in objectives that could jeopardise the successful implementation of EAF for the southern Benguela (Shannon *et al.* 2004). These included resistance by existing quota- or rights-holders to accept reduced quotas or access rights if these were required to address ecosystem issues, even if there are potential long-term benefits; conflicts between consumptive and non-consumptive users; conflicts between different sectors, such as demersal and pelagic, and small-scale, recreational and large-scale fishers; conflicts related to other users of marine ecosystems, such as mining and industrial users. These would all need to be confronted and reconciled or resolved by high level policy decisions if EAF is to be implemented.

The conflicts can only be resolved, if at all, through effective consultation and participation. Problems could arise through factors such as an unwillingness of stakeholders to participate in negotiations, sometimes in the belief that non-cooperation is the option most likely to achieve their own objectives. Inadequate systems of user rights, a lack of access to information, insufficient time and resources devoted to consultation, a lack of capacity or large differences in capacity that will hinder real negotiation, will all impede good consultation and participation, reducing the probability of achieving support for an EAF. FAO (2003) describes a number of other threats, and the sections of this paper on the different fisheries in South Africa refer to a number of actual and potential conflicts between different stakeholders and differing goals. These problem areas must be identified, examined and addressed, where appropriate, in the early stages of implementation of EAF.

CONCLUSIONS

The objectives and principles set out in Chapter 1 of the South African Marine Living Resources Act No. 18 of 1998 (Anon. 1998) already provide the incentive and the mandate for progressing towards an EAF in South Africa's marine fisheries. In addition, much groundwork has already been done in extensive and high-quality research into many aspects of the features, interactions and dynamics of key marine ecosystems in the country, especially the Benguela ecosystem, and in implementation of solid and frequently effective fishery management institutions and regimes. As described in this paper, some progress has also been made in implementation of EAF practices in some fisheries.

The paper has also shown that important problems and questions remain in and across all fishery sectors. Some of these issues are clearly substantive and relatively urgent if the risk of irreversible damage is to be sufficiently reduced and progress made in restoring the most heavily impacted components of the Benguela ecosystem and its sub-ecosystems to their full potential. This paper and the FAO Guidelines are intended to demonstrate the general process necessary to begin implementation of EAF. It is suggested that the process may be best initiated by conducting an extensive review of the primary interactions between fisheries and their target and affected species, as well as between fisheries and other users, and the ecosystem itself. The review should also consider the implications of these interactions for the long-term sustainability and productivity of the country's marine ecosystems and living marine resources. The results of the review should indicate where changes and enhancements of the existing single-species management strategies will be required. This paper is intended to provide a starting point for this process.

ACKNOWLEDGEMENTS

We wish to express our thanks to a number of individuals who have made valuable contributions to this paper. Particular mention should be made of George Branch, Rob Crawford, Rob Leslie, Carl van der Lingen and Colin Attwood.

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