

ECOSYSTEM APPROACH TO FISHERIES MANAGEMENT IN THE NORTHERN BENGUELA: THE NAMIBIAN EXPERIENCE

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The northern Benguela marine resources have a long history of questionable management primarily because of the limitations of management measures under the political framework of the time (either free access or under an international authority with no national representation, authority or enforcement power). Only after its Independence in 1990 could Namibia exercise its own national management policies, but by that time the most important commercial stocks were severely depleted. Since Independence, and despite strong management measures being implemented, the recovery of the stocks has not been as successful as expected. Some of the possible reasons are the effect of environmental variability on some stocks, ecosystem effects of fishing and unforeseen trophic interaction effects. To date, fisheries management has been based largely on a single-stock approach, but Namibia is committed to implement, in addition, an ecosystem approach to fisheries (EAF) management. The work leading to this implementation is described, in particular an ecosystem modelling study undertaken to summarize the current understanding of the northern Benguela ecosystem, to provide a basis for future work towards an EAF in Namibia. Model simulation results suggest that, given the present assumed trophic structure of the northern Benguela, altering the major fisheries would not result in recovery of the small pelagics to levels seen 40 years ago, suggesting that the original foodweb in the region may have been altered dramatically. Cape hake *Merluccius capensis* and *M. paradoxus* were negatively impacted when a large fishery on small horse mackerel *Trachurus trachurus capensis* was simulated. Model simulations illustrate the important finding that ecosystem effects of altered fishing scenarios are often not of a magnitude or direction that would be expected by considering predator-prey relationships in the absence of indirect trophic effects. Trophic effects may have large indirect consequences for some components of the system, for example seabirds.

Key words: ecosystem model, fisheries management, Namibia, northern Benguela

To date, management of Namibian fisheries has been based almost entirely on single-stock assessments. Only a few implemented management measures, such as the minimum depth limit for bottom trawling or size selection devices in codends, were motivated to limit the bycatch of juveniles or other non-targeted species. Trophic interactions, competition or predator-prey relationships are not taken into account in the management procedures. However, Namibia is committed to complement these methods by implementing an ecosystem approach to fisheries (EAF). This paper provides a background to the northern Benguela ecosystem and the history of the development and management of its main fisheries. It documents the progress made to date towards introducing an EAF in Namibia, with particular emphasis on the role of ecosystem modelling. The foundations that have already been laid for the future development of ecosystem-based management in Namibian fisheries are described, and attention is drawn to the need to further define and agree on the role and direction of future ecosystem research.

BIOLOGICAL AND HISTORICAL BACKGROUND

The northern Benguela system is a typical eastern boundary current upwelling system, characterized by high biological productivity. Its northern limit is the Angola–Benguela Front, a well-marked oceanographic feature that separates the tropical water of the Angola Current and the cool, nutrient-rich, upwelled water of the Benguela (Fig. 1). The front is situated off southern Angola, shifting seasonally between 14 and 17°S (Shannon 1985, Shannon and Nelson 1996). Off southern Namibia, the Lüderitz upwelling cell is a perennial feature, and is considered the most intense upwelling cell of the world's oceans (Bakun 1993, 1996). This region of very high wind stress, high turbulence and cold water seems to act as an environmental barrier to most fish stocks, and in effect divides the Benguela ecosystem into northern and southern components (Shannon 1985, Agenbag and Shannon 1988).

With the exception of some highly migratory large

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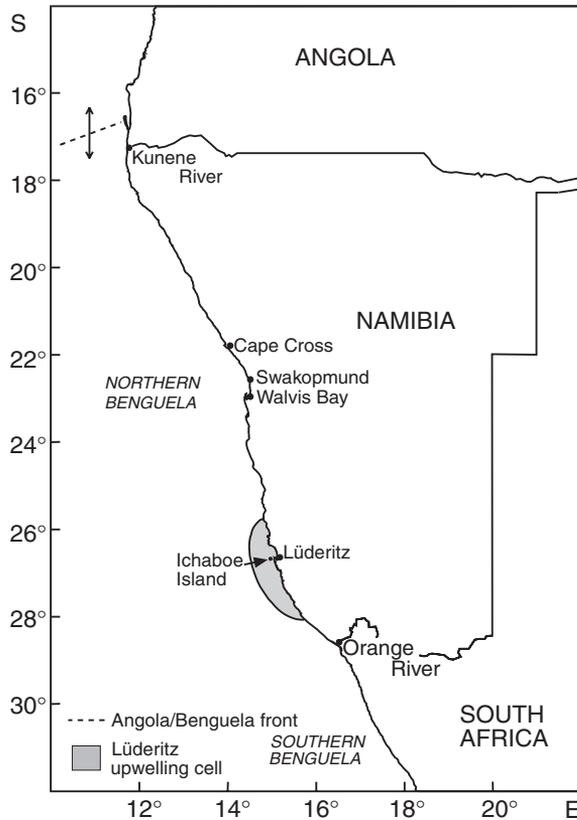


Fig. 1: Map of the northern Benguela showing places mentioned in the text

pelagic species such as tunas and swordfish, and possibly some deep-water demersal species, such as Cape hake *Merluccius paradoxus*, shared with the southern Benguela, most fish stocks of the northern Benguela are considered to be discrete, with their distribution limited by the Angola–Benguela Front to the north and the Lüderitz upwelling region to the south (Fig. 1). They can, therefore, be regarded as single management units. The composition of biological communities of the northern Benguela is comparable with that of the southern Benguela, the same species constituting the main commercial stocks (Bianchi *et al.* 1999). The northern Benguela system is subjected to large-scale environmental variability, which originates in the tropics and is similar to the *El Niño* phenomenon in the Pacific Ocean. These major environmental anomalies, termed Benguela *Niños*, have severe biological effects at all levels of the marine foodweb (Shannon *et al.* 1986, Gammelsrød *et al.* 1998, Roux 2003).

Early history of marine resource exploitation, 1750–1900

The hyper-arid and inhospitable Namib Desert extends along the entire coast of the northern Benguela. Consequently, the coast has always been sparsely populated, and there is no evidence that indigenous populations were ever engaged in artisanal fisheries. However, the few nomadic groups that roamed the coastline until the late 19th century used marine resources extensively, as the composition of their middens attests: shellfish, seabirds and marine mammals were the major food, penguin and fur seal pelts were used for clothing, and whale bones to build shelters (Avery 1985, Best and Ross 1996, Kinahan 1990).

The first large-scale commercial exploitation of marine resources in the northern Benguela started about three centuries ago, with the development of sealing and whaling activities by North American and European vessels (Best and Ross 1995, David 1995). The operations were unregulated and indiscriminate, and resulted in the rapid depletion of the targeted stocks. Catches of southern right whales *Eubalaena australis* were already declining by the beginning of the 19th century (Richards and du Pasquier 1989) and the local breeding population was possibly totally eradicated by the beginning of the 20th century (Best and Ross 1986, Roux *et al.* 2001). Similarly, the Cape fur seal *Arctocephalus pusillus pusillus* population was driven to very low levels by the end of the 19th century, and had been eradicated from at least 23 island colonies throughout the species' range (Shaughnessy 1984, David 1995). After the decline in fur seals, attention was shifted to the exploitation of guano from the seabird islands, which started with the Ichaboe Island guano rush (1843–1845) and has continued until present. Whalers, sealers and other mariners episodically exploited seabird colonies off Namibia throughout the 18th and 19th century for fresh meat and eggs (Anon. 1845, Eden 1846, Best and Shaughnessy 1979, Kinahan 1990). The additional exploitation of seabirds and disturbance associated with the early guano exploitation during the late 19th and early 20th centuries impacted negatively on seabird populations and particularly the African penguin, *Spheniscus demersus* (Roux *et al.* in press). By the end of the 19th century, only a few stocks had been exploited, but in the absence of regulatory management measures they were all severely depleted.

Pre-independence 20th century fisheries expansion

At the beginning of the 20th century, regulation of sealing began, as government enterprise, on all islands. Seal harvests at Cape Cross, which had been conducted

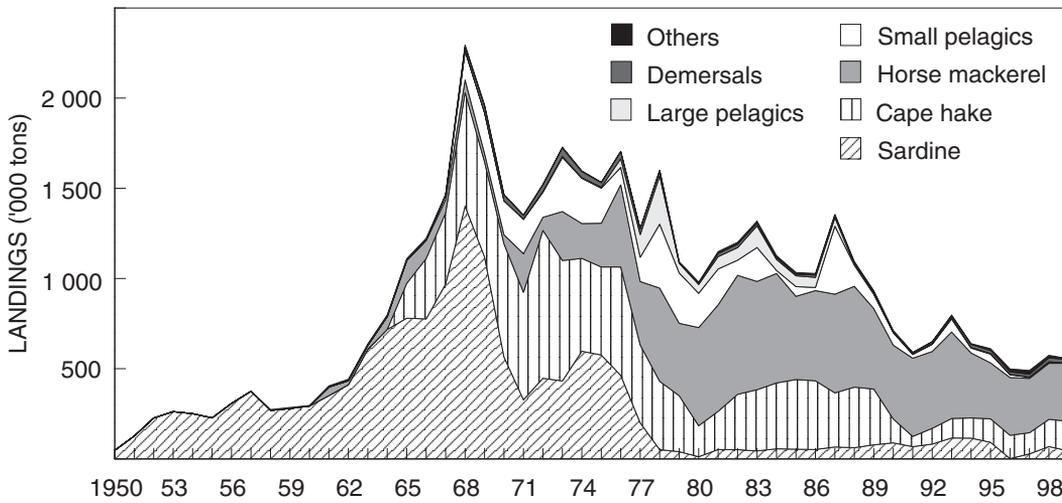


Fig. 2: Commercial landings from the northern Benguela during the second half of the 20th century (redrawn and modified from Boyer and Hampton 2001, and Ministry of Fisheries and Marine Resources, unpublished data). The species were grouped as follows: Cape hake, both *M. capensis* and *M. paradoxus*; horse mackerel: catches from midwater trawlers and purse-seiners; small pelagic: anchovy, round herring and other small pelagic species; large pelagics: tuna, snoek and chub mackerel *Scomber japonicus*; demersal: includes monkfish *Lophius* spp., kingklip *Genypterus capensis*, sole *Austroglossus* spp. and *Dentex*; others: includes fur seals, crustaceans (rock lobster and crab), and orange roughy *Hoplostethus atlanticus*

by private enterprise since before 1903, continued without catch limits until the mid 1970s (Best 1973, Shaughnessy 1982). As the seal population recovered from earlier unregulated overexploitation, the catches increased rapidly, particularly between 1950 and 1970 (Wickens *et al.* 1991). In the early 1970s, government involvement in sealing ended and, with the promulgation of the Sea Birds and Seals Protection Act in 1973, the industry has been run by private concessionaires under a quota system. In addition to those legally harvested, a considerable number of seals were also killed by fishing vessels at sea, particularly by purse-seiners targeting small pelagic fish, and linefish boats targeting snoek *Thyrstites atun* (Shaughnessy 1982, Wickens *et al.* 1991). In the early 1980s, the catches declined markedly as a result of a collapse of the world market for pelts, the main product of the sealing industry up until then.

The Cape rock lobster *Jasus lalandii* fishery, operating from Lüderitz, started in the 1930s and developed rapidly in the late 1940s, peaking in the early 1950s with approximately 14 000 tons of lobsters landed in 1952 (Stuttaford 1994). Despite several management measures put in place (annual quotas, size limits, closed season), catches of rock lobster declined steadily until the late 1960s, when the size limit was abolished. This resulted in severe depletion of the stock, and

catches remained below 2 000 tons, with no sign of stock recovery. An additional rapid stock decline during the 1988/89 fishing season was probably caused in part by anomalously low oxygen concentrations in coastal waters at that time. As a consequence, the annual fishing quota was reduced from 2 000 tons in 1988/89 to 100 tons in 1991/92.

The development and decline during the 20th century of the different industrial fisheries targeting fish species in the northern Benguela (Fig. 2) was reviewed by Boyer and Hampton (2001). The purse-seine fishery targeting sardine *Sardinops sagax* first developed during the late 1940s (Armstrong and Thomas 1995), and increased rapidly in the 1960s after relaxation of the quotas and the introduction of factory vessels. Catches peaked in 1968 at 1.4 million tons after the adoption of a smaller mesh size, but declined markedly over the following years. After a brief partial recovery in the mid-1970s, the stock finally collapsed to extremely low levels by 1977. The purse-seine fleet, while continuing to catch sardine (and probably contributing to keeping the stock at a very low level), shifted its activities to anchovy *Engraulis encrasicolus*. This species dominated the pelagic catches for a few years in the late 1970s and early 1980s, together with juvenile horse mackerel *Trachurus trachurus capensis*.

The second phase in the history of the northern

Benguela commercial fisheries started in the mid-1960s, with the development of a long-distance trawl fishery targeting the two species of Cape hake, the shallow-water *Merluccius capensis* and the deep-water *M. paradoxus*. The fishery was operated mainly by nations other than Namibia, which at that time had not attained its Independence. Catches increased rapidly to peak in 1972 at more than 800 000 tons, but started declining rapidly soon thereafter. Despite some attempts at implementing management measures, such as minimum mesh size (in 1975) and total allowable catches (in 1977), hake landings continued to decline until the early 1980s (van der Westhuizen 2001). Two strong year-classes in the early 1980s likely contributed to a slight increase in the catches of Cape hake, but by the end of that decade the stocks were severely depleted. The reason for the poor performance of the management measures proposed by the International Commission for the Southeast Atlantic Fisheries (ICSEAF) are debatable, but poor implementation and likely biased reporting of catches and efforts are suspected to have played a major role in the decline in Cape hake stocks (Oelofsen 1999, Boyer and Hampton 2001).

As Cape hake catches started to decline in the early 1970s, and after the crash of the sardine fishery, the midwater trawl fishery that was targeting horse mackerel expanded considerably. Annual catches of horse mackerel rapidly increased between the early 1970s and early 1980s to more than 600 000 tons, possibly following an increase in the stock biomass (Vaske *et al.* 1989). While slowly declining from the 1980s through to present, horse mackerel catches have dominated the catches in the region since the late 1970s (Fig. 2).

The development of the commercial fisheries in the northern Benguela until Namibian Independence in 1990 is characterized by sequential exploitation and depletion of the three most important stocks. The total fish catches increased rapidly throughout the 1960s to reach 2.3 million tons in 1968, dominated by sardine (61.3%). Thereafter, catches declined sharply to 1.5 million tons in 1972, dominated by Cape hake (54%), then declined further to 0.9 million tons in 1989, dominated by horse mackerel (44.9%). In addition to these changes in the fisheries and the commercial stocks, the period was also marked by other major changes in the foodweb and possibly in the functioning of the northern Benguela ecosystem as a whole (Cury and Shannon 2004, Heymans *et al.* 2004). In particular, jellyfish biomass seems to have increased markedly over the Namibian shelf since the 1970s (Venter 1988, Fearon *et al.* 1992). The principal forage species of most of the top predators changed

from sardine and anchovy to pelagic goby *Sufflogobius bibarbatus*, following a probable large increase in biomass of the species from the late 1960s (Crawford *et al.* 1985).

Fisheries management since Namibia's independence

Only after independence in 1990, by when Namibia's major fish resources were already severely depleted, could the country proclaim full authority over its marine resources. Therefore, at Independence, the first priorities were to stop illegal fishing, to implement management measures to rebuild the depleted stocks, and to ensure sustainable exploitation of the resources for the benefit of the new nation (Oelofsen 1999). After proclaiming its authority over a 200 nautical mile EEZ, Namibia promulgated laws and regulations and implemented management measures to protect its fish stocks. Success was rapid; within a year, illegal fishing by foreign fleets was virtually eradicated, some stocks showed encouraging signs of recovery (Stutaford 1994), and the contribution of the fisheries sector to the Namibian economy started to increase (Kankondi 1994). In order to improve scientific advice to decision-makers on how to manage the marine resources, two new research institutes were established; the national Marine Information and Research Centre in Swakop-mund and the Lüderitz Research Centre.

However, these early successes were also accompanied by shortcomings. For example, some major stocks, particularly small pelagics, failed to recover as expected (Boyer and Hampton 2001), and the status of some vulnerable and endangered species such as Cape gannets *Morus capensis*, African penguins and bank cormorants *Phalacrocorax neglectus* worsened (Crawford *et al.* 1999, Kemper *et al.* 2001, Du Toit *et al.* 2002). Total fish landings in the northern Benguela continued to decline in the 1990s, falling to just over 560 000 tons in 2000, consisting mostly of horse mackerel (58.6%). A moratorium was placed on the sardine fishery in 2001 for one year, the first time since the fishery had begun. This followed some environmental anomalies during the mid-1990s, including the major 1995 Benguela *Niño*. The moratorium highlighted two shortcomings in the management procedures at that time: the single-stock approach does not (cannot) explicitly take into account (1) the effects of environmental anomalies, and (2) trophic effects and multispecies interactions. This is a major challenge for fisheries research and management in the 21st century (Roux 2003).

TOWARDS A MULTISPECIES APPROACH TO FISHERIES MANAGEMENT IN THE NORTHERN BENGUELA

The principles of natural resources (and therefore fisheries) utilization in Namibia are enshrined in the constitution, which states *inter alia*: “The State shall actively promote and maintain the welfare of the people by adopting policies aimed at... the maintenance of ecosystems, essential ecological processes and biological diversity of Namibia and utilization of living natural resources on a sustainable basis for the benefit of Namibians, both present and future...” (Constitution of the Republic of Namibia – Article 95.1; <http://www.orusovo.com/namcon/constitution.pdf>).

After promulgating its authority over a 200-mile EEZ soon after Independence, in accordance with the United Nations Law of the Sea, the Namibian Government promulgated the Sea Fisheries Act “To provide for the conservation of the marine ecology and the orderly exploitation, conservation, protection and promotion of certain marine resources; for that purpose to provide for the exercise of control over sea fisheries; and to provide for matters connected therewith” (Anon. 1992).

Namibia became a signatory of The Rome Declaration on the Implementation of the Code of Conduct for Responsible Fisheries at the Food and Agriculture Organization of the United Nations (FAO 1995) Ministerial Meeting on Fisheries, 1999. Subsequently, the legislation pertaining to fisheries management was updated with the promulgation of the Marine Resources Act, 2000 (Anon. 2000). Namibia was also a signatory to the Reykjavik Declaration on Responsible Fisheries in the Marine Ecosystem, which marked the formal adoption of a commitment to ecosystem approaches to fisheries management at a global level (FAO 2001).

The above demonstrates the commitments Namibia has made to the management of its living marine natural resources and provides clear signs that, together with the principles of sustainability, ecosystem interactions are to be taken into account. In 1998, an “Ecosystem Analysis Section” was activated within the Directorate of Resource Management of the Ministry of Fisheries and Marine Resources, with a threefold agenda:

- study the effects of environmental variability on the living marine resources;
- explore the potential of biological indicators of changes in the ecosystem;
- develop guidelines and provide scientific advice to implement an ecosystem approach to fisheries management.

At the end of 1998, a project entitled “Trophic interactions in the Benguela ecosystem and their implications for multi-species management of fisheries” was initiated in Namibia by FAO, funded by the Government of Japan (as part of the Implementation of the Kyoto Conference Plans of Action). The main objective of this project was to construct an improved, updated, dynamic ecosystem model of the trophic flows of the northern Benguela, to facilitate the development and evaluation of multispecies management techniques for the marine resources of Namibia and possibly the entire Benguela.

Ecosystem model of the northern Benguela

The first phase of the project involved the collection and analysis of dietary data from selected important predators (in particular Cape fur seals, shallow-water and deep-water Cape hake, and other predatory species), as well as some basic information on forage fish species (myctophids and pelagic goby). A workshop was convened in August 1999 to collate published and unpublished data on the different components of the ecosystem, with the help of various experts (Roux and Shannon 2000). The likely biases and levels of uncertainty of each dataset were discussed at the workshop, to guide and facilitate the balancing process of the resulting ecosystem model. The modelling approach used was ECOPATH with ECOSIM (EwE; Christensen and Pauly 1992, Walters *et al.* 1997). Local scientists and international experts at the workshop agreed on how the structure of existing ECOPATH models of the region (Jarre-Teichmann *et al.* 1998, Shannon and Jarre-Teichmann 1999) should be modified in order to tailor the model to the fishing scenarios to be tested. The major product of this phase of the project was an updated ECOPATH model that described trophic flows in the northern Benguela in the late 1990s, following the anomalous environmental conditions experienced during 1994 and early 1995 (Roux and Shannon 2000). The model has been further updated and modified in line with the generic structure and parameterization agreed upon for trophic models of upwelling ecosystems (Moloney and Jarre 2003, Moloney *et al.* *subm.*). Model inputs, parameter estimates and summary statistics of this updated model are given in the Appendix.

The sensitivity of model outputs to uncertainty in the input parameters of the main fish groups was explored using that routine in EwE. The model was most sensitive to variability in the input parameters of the Cape hake groups (Shannon *et al.* 2001). In addition, the

Table 1: Selected results from simulations undertaken to investigate the ecosystem implications of multispecies management strategies (from Shannon *et al.* 2001). Unless otherwise stated, simulations were run for 50 years, starting from the balanced model for the period 1995–1999 when biomasses of many species were at very low levels. Default vulnerability parameter values (mixed flow control) were assumed, except where otherwise indicated (sensitivity tests)

Management strategy	Estimated outcome for selected groups (% change in biomass)
<i>Closure of fisheries to investigate potential for stock recovery</i>	
Closure of sardine fishery	All model groups: between –4 and +4 Sardine: +40
Closure of demersal trawl hake fishery	<i>M. paradoxus</i> : +63 <i>M. capensis</i> : +39 Forage fish stocks: –12 to –14 Predators on forage fish: –14 to –16
Moratorium on catches of small and large horse mackerel	Large horse mackerel: +38 <i>M. capensis</i> : +7 Fur seals: +9 Seabirds: –84 Most other groups: between –5 and +5
<i>Implications of altered fishing strategies for potential competitors</i>	
Opening of fishery on pelagic goby (e.g. removing 50% of goby production)	Goby: –87 Anchovy: +51 Sardine: +27 Small horse mackerel: +21 <i>M. capensis</i> : –25 <i>M. paradoxus</i> : +16
Increasing fishing mortality of small and large horse mackerel (e.g. fourfold)	Sardine: negligible effects Seabirds: +70 Large pelagics: +35 <i>M. capensis</i> : –30 <i>M. paradoxus</i> : +17
Onset of fishery removing 50% of gelatinous zooplankton production	Anchovy: +22 Cephalopods and horse mackerel: +33 Large pelagic fish and whales: +42 <i>M. capensis</i> : +20 <i>M. paradoxus</i> : +33 Pelagic goby: –8 Seabirds: –6
<i>Implications of altered fishing strategies for predator and prey groups</i>	
50% reduction in Cape fur seal population	Anchovy: +16 Sardine: +10 Most other groups: between –4 and +5
95% reduction in Cape fur seal population	Sardine: +21
Moratorium on sealing	Fur seals: +8 Other model groups: between –2 and +2
Moratorium on sealing (assuming top-down control by seals of their prey)	Fur seals: +53 (after 75 years) Sardine: –16 (after 75 years) Anchovy: –17 (after 75 years)
Fourfold increase in small horse mackerel fishing mortality (purse-seine fishery)	Horse mackerel: –18 <i>M. capensis</i> : –8 <i>M. paradoxus</i> : –2 Seabirds: +13 Goby: +8 Total catches (all groups): +20
Fourfold increase in large horse mackerel (mid-water trawls) fishing mortality	Horse mackerel: –88 <i>M. capensis</i> : –30 <i>M. paradoxus</i> : +17 Goby: +33 Seabirds: +59 Total catches (all groups): –39

(continued)

Table I: (continued)

<i>Exploring interactions between different fisheries</i>	
Fourfold increase in fishing mortality of large <i>M. capensis</i>	Large <i>M. capensis</i> : -36 Small <i>M. capensis</i> : -27 <i>M. paradoxus</i> : between -5 and +5 Dolphins: +17 Seabirds: +22

ECORANGER routine of EwE was used to seek alternative balanced models, by drawing input parameter values in a Monte Carlo fashion from specified ranges and frequency distributions of input parameters (Christensen and Walters 2004). The difficulty in balancing the original model was reflected by the paucity of alternative possible models found using this routine, showing that the foodweb of the northern Benguela simulated in the model is tightly constrained (Shannon *et al.* 2001).

Dynamic simulations of the possible ecosystem effects of altered fishing strategies can be explored using EwE. Given hypothetical changes in fishing mortalities, the rates of change in biomass of model groups over time can be simulated. Based on the “foraging arena” concept, dynamic consumption rates are calculated according to assumptions of flow control from predator to prey, prey-handling times, relative feeding times of predators and prey, etc. (Christensen and Walters 2004). In EwE, consumption by a model group changes over time, depending on the model abundances of the prey groups as well other predators that may be competing with the group. In this way, possible ecosystem implications of competition and indirect trophic effects can be explored.

In the second phase of the project, scenarios of different plausible fishing strategies were identified. Fisheries scenarios selected for testing were grouped thematically. Hereafter, examples of simulations undertaken in each of the fishing scenario categories are presented to illustrate the kind of questions and interactions that can be explored using an approach such as EwE applied in the Namibian context, and to highlight the types of results that could prove helpful in providing qualitative inputs for fisheries management advice. All simulations presented below were undertaken using ECOSIM. Simulations were run for 50 years or until equilibrium was reached, assuming default model parameter settings. In particular, the default vulnerability setting for mixed flow control was assumed. In upwelling systems in particular, the system’s dynamics are sensitive to the assumptions of flow control between groups at mid-trophic levels (Cury *et al.* 2000, Shannon *et al.* 2000). The effects of assumptions of top-down versus bottom-up control on the outputs of the latter scenario were tested. In EwE, the vulnerability parameter is used to deter-

mine the type of control between groups (Christensen and Walters 2004). Flow control is expressed by means of the vulnerable and non-vulnerable portions of a prey group to each of its predators.

The results from some of the original simulations (Shannon *et al.* 2001) to address important questions are given below as indications of the direction and relative magnitude of the changes. However, they should not be treated in a quantitatively predictive sense, because the models have recently been updated and revised (see Appendix). In addition, uncertainties around input model parameter values need further consideration.

Given the present structure of the ecosystem, what is the potential for recovery to historical levels of the main depleted stocks by altering the fishing strategy on those stocks?

Several examples of simulations of this nature were run, ranging from slight changes in present fishery mortalities to complete closure of some fisheries (Table I). In the case of sardine, a simulated closure of the fishery had little effect on the other components of the system, whereas the sardine stock only increased moderately, to about 1% of the maximum estimated biomass before the stock collapsed in the early 1970s.

A hypothetical closure of the hake fishery resulted in larger increases in the target stocks than for sardine. The effect was more pronounced for large *M. paradoxus* than for large *M. capensis*. This is not surprising because *M. paradoxus* is more heavily exploited. Other likely consequences of such closure are declines in forage fish stocks and a decline in other predators such as seabirds and pelagic-feeding demersals that depend on those forage fish species.

A modelled moratorium on horse mackerel catches resulted in a moderate increase in the biomass of large horse mackerel and tended to favour some horse mackerel predators, such as *M. capensis* and fur seals, but caused seabirds to decline dramatically (Table I).

These extreme examples illustrate that, given the present assumed trophic structure of the northern Benguela, altering the major fisheries would not cause a recovery of the small pelagics to the levels seen 40 years ago, suggesting that the original foodweb in the region may have been altered dramatically. They also suggest that trophic effects may have large indirect

consequences on some components of the system (e.g. seabirds).

What is the potential strength of competitive interactions between stocks, and how would a target stock be affected by altered fishing strategies on potential competitors?

Since the collapse of the Namibian sardine stock, concerns have been voiced about the potential effect of perceived competitors in limiting sardine stock recovery. For example, heavy exploitation of anchovy was encouraged at the time of the sardine collapse, in the forlorn hope that this would promote the rapid recovery of sardine (Shelton 1992). The only significant result of this management attempt was the rapid decline of the Namibian anchovy stock in the late 1970s. Subsequently, similar suggestions have been made about juvenile horse mackerel and goby, with the hope that a decrease in those stocks would favour the recovery of sardine.

In order to test these effects, opening a fishery on pelagic goby was modelled. An extreme scenario (Table I) resulted in a drastic decline of the goby biomass, effectively "vacating the niche". However, the net effect on sardine was not a dramatic recovery, but a modest increase as a result of increased predation pressure. The side effects of such a strategy were important for other stocks. *M. capensis* (preying on pelagic goby) declined, whereas *M. paradoxus* increased, favoured by reduced predation by *M. capensis* as well as increases in some of the prey of small and large *M. paradoxus* (zooplankton and mesopelagics).

Similarly, several scenarios affecting horse mackerel were simulated, by increasing the fishing mortality of small horse mackerel (targeted by purse-seiners), large horse mackerel (targeted by midwater trawlers) or both. Again, curbing a perceived competitor did not result in any substantial recovery of the sardine stock. However, potentially important negative effects of some strategies that were not modelled in this study could have devastating effects on the very stocks that such scenarios would be expected to benefit. For example, the directed purse-seine fishery for horse mackerel takes sardine as a bycatch. As a consequence, a drastic increase in this fishery could impact sardine stocks negatively. Similarly, the midwater trawl fishery could potentially impact many species (particularly sardine and both species of small hake).

Given the large biomass of gelatinous zooplankton in the northern Benguela, this group may have potentially important competitive as well as predatory effects on other stocks. The simulated removal of jellyfish had positive effects on most groups (Table I). However, understanding of the ecological role of gelatinous zooplankton in Namibian waters is limited at present. For example, the impact of the group on fish recruitment

(particularly as a result of predation on eggs and larvae) could be considerable (Möller 1984, Boyer *et al.* 2001).

What trophic effects would altered fishing on predators or prey of a given stock have on its trajectory, as well as on the rest of the system?

The possibility of limiting the potential predation pressure (particularly by seals) on a fished stock to improve its recovery has been suggested for many of the world's depleted stocks, including those of the Benguela (Shaughnessy 1985, Butterworth *et al.* 1988, Wickens and Shelton 1988). Several scenarios of reduction of the Cape fur seal population were modelled to examine the potential benefit for commercially important prey species (e.g. sardine and hake). Following a 50% reduction in seal biomass, effects on other groups were small (Table 1). Even by increasing sealing to the point of reducing the seal population to only 5% of its initial estimate, the benefit to the sardine stock was only moderate. However, Plagányi and Butterworth (2004) note that, as a consequence of the underlying functional response assumptions in the model, simulated responses to reduced top predator abundance can be dampened. Furthermore, in the absence of age-structured parameterization (e.g. of growth efficiency), ECOSIM is likely to underestimate the ecosystem effects of top predator removal (Aydin 2004).

Opposite scenarios were tested to estimate the potential impact of a reduction or closure of sealing on different prey species. Again, there were only small changes: model biomasses of all other groups were retained within 2% of their original values. However, equilibrium seal biomass was sensitive to the vulnerability parameter setting. Under the closure to sealing scenario, there were large increases in model seal biomass when top-down control by seals of their prey was assumed (Table I).

Different scenarios of altered fishing levels on horse mackerel, which is an important prey of Cape hake in the northern Benguela (Traut 1996), were modelled to test their potential effects on the hake stocks (Table I). Hake were negatively impacted when a heavier fishery on small horse mackerel was simulated, although the modelled decline of *M. capensis* was greater than that of *M. paradoxus*. This could be a consequence of the larger contribution of horse mackerel to the diet of *M. capensis* (Traut 1996, Shannon *et al.* 2001). Again, this scenario did not take into account the potentially serious detrimental effect of sardine bycatch (bycatch was not modelled in this study).

A similar increase in fishing mortality of large horse mackerel (targeted by midwater trawlers) caused the horse mackerel stock to collapse, with implications for its predators and prey (Table I). Simulations such as these illustrate that ecosystem effects of altered fishing scenarios are often not of a magnitude or even

a direction that would be expected from consideration of predator-prey relationships in the absence of indirect trophic effects.

What are the interactions between the different fisheries?

Several of the examples given above highlight interactions between different fisheries in the northern Benguela. Additional scenarios were also considered, in particular involving the fishery on the two species of Cape hake. Both stocks are exploited by the same fishery, but *M. paradoxus*, which is preyed upon by *M. capensis* (Punt and Leslie 1995), dominates the hake catches. Possible effects of differential changes in fishing pressure on those two species were examined (Table I).

In another scenario, by decreasing fishing mortality of large *M. capensis* and *M. paradoxus* while differentially increasing that of small hake, the overall hake catches could be kept virtually constant (although at the cost of a decrease in the size of fish in the catch), whereas the biomass of large *M. paradoxus* could experience considerable gains. Although not very realistic, these scenarios nevertheless illustrate how the interplay of size-selection devices and depth limits to fishing could differentially affect the two Cape hake stocks, with little change caused to other groups.

Implications for an ecosystem approach in the northern Benguela

The simulations presented provide some insights into the nature of the trophic interactions between the different biotic groups of the northern Benguela, and how the effects of fishing can propagate through the ecosystem as a result of trophic interactions. At present, Namibian fisheries, like all other major marine capture fisheries around the world, are managed largely on a single-species basis, in which it is assumed that the dynamics of any given stock are purely a function of the characteristics of that stock. The results presented here give an indication of the weaknesses and potential pitfalls of that assumption. For example, the development of a fishery for goby, while probably bringing new economic returns to Namibia, could result in reductions (of 10% or more for the scenarios presented here) in the biomass of *M. capensis*. This reduction in biomass could be anticipated to lead to a reduction in long-term yield of the resource. Similarly, an increase in fishing mortality on small horse mackerel is likely to lead to a reduction in abundance of both Cape hake stocks, particularly *M. capensis*. In the ongoing development of an optimum strategy for sustainable utilization of the northern Benguela ecosystem and effective management of Namibia's fisheries, inter-specific interactions need to be considered. The ecosystem modelling approach taken here is one source

of potentially useful inputs to this process.

Several important ecosystem issues need to be addressed, which are not formally considered in management decisions at present:

- effect of fishing on non-target components of the ecosystem;
- interactions between different fisheries;
- changes in ecosystem state (regime shifts, competition between small pelagic fish species);
- species of conservation concern;
- biodiversity issues;
- impact of opening new fisheries (or closing fisheries) on the system;
- protected areas, including fishing restriction zones (e.g. 200-m depth limit for trawlers).

THE WAY FORWARD

Simulations such as those examined in this study, including the consideration of uncertainties, give valuable insight into the nature and possible implications of the trophic interactions in the northern Benguela ecosystem and how the effects of fishing may propagate through the ecosystem as a result of the interactions. However, given the many uncertainties in the way any ecosystem model represents the interactions and how ecosystem structure might respond to changes, it would not be possible to use such a model as a predictive tool at this stage. Additional analyses and alternative models are needed to reduce the uncertainties and to test the underlying assumptions. Ecosystem models can constitute a useful synthesis of current understanding of the trophic structure of the ecosystem and can highlight critical gaps in knowledge of the system, so helping to set research priorities.

The future role of multispecies models in the management framework applied to northern Benguela marine resources was considered by Roux and Shannon (2002), and future steps to take this process further were recommended. These include in particular the incorporation of new research data into the models, and motivations for specific projects in order to address some of the critical uncertainties. In addition, the concept of ecosystem management needs to be formalized in the northern Benguela through a consultative process between the fishing industry, fisheries researchers and managers, as well as other stakeholders.

Stock assessment is an imprecise discipline, for many reasons. Multispecies stock assessment and forecasting involve even more uncertainties than single-species projections, and are therefore intrinsically less precise. However, the information generated by multispecies and ecosystem models, when appropriately

interpreted, is valuable on a different scale. Ecosystem/multispecies models can give important insights into the realities of ecosystem dynamics, predator-prey and competition interactions, among other multispecies effects. Such effects cannot be adequately addressed by means of traditional approaches that consider a single stock in isolation from others and the ecosystem with which it interacts, either directly or indirectly. Traditional single-stock assessment tools usually remain the best available tools for short-term tactical management decisions. They have the advantage of incorporating more details on age and size structure of the fished stocks and on gear selectivity than is normally possible in ecosystem models, and they provide reference points for single stocks. However, to address some of their shortcomings, they can be complemented by multispecies approaches, especially for medium and long-term, strategic fisheries management. Ecosystem models are also valuable tools to assist in setting multispecies objectives for fisheries management, for testing the robustness of long-term changes in management strategies and policies, and for assessing the ecological sustainability of present and future fisheries management scenarios.

It is clear that incorporating ecosystem (multispecies) considerations into the Namibian fisheries management framework has become a necessary target. Ecosystem modelling provides the basis from which a structured framework can be developed to implement an EAF in Namibia. Good multispecies models can provide measurable ecosystem-based indicators and be used in fisheries management to assess biodiversity and conservation issues, as well as ecological sustainability. A goal of future work to improve the advice given to managers should therefore be to combine the insights gained from both single-stock and ecosystem/multispecies approaches for ecologically sustainable fisheries management.

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APPENDIX

Specifications of the ECOPATH model for the northern Benguela for the late 1990s

The ECOPATH model used in the simulations is fully described by Shannon *et al.* (2001). The area considered was the shelf from about 20 to 600 m depth, and from the Orange River in the south to 15°S off southern Angola (the approximate average latitude of the Angola–Benguela Front). The resulting area encompassed by the model was 186 500 km², and included the distribution of the small pelagic stocks in southern Angola (particularly sardine), but excluded littoral communities (and rock lobster), as well as deep-water stocks (in particular orange roughy).

For the dynamic simulations, the species groups were aggregated into 28 compartments, including four groups for the two hake species (small and large for each), two groups of cetaceans (toothed whales and dolphins in one group, baleen whales in another) and

an additional group of forage fish to distinguish the small pelagic community near the Orange River (mainly anchovy from the South African stock).

Similarly, the fishing fleets were disaggregated by fish groups for the dynamic simulations, and a “squid fishery” was introduced (there was no directed squid fishery in the northern Benguela in the mid-1990s, and squid catches were a bycatch in other fisheries). Also note that the “demersal fishery” in Table. App III consists of the demersal trawl fisheries (for hake, monkfish and sole) and the demersal longline fishery (targeting hake), but not the deep-water trawl fishery (targeting orange roughy).

The model was subsequently updated with the incorporation of new diet information and aggregated following Moloney and Jarre (2003).

App. Table I: Basic estimates and basic inputs (bold) in the balanced ECOPATH model of the northern Benguela for the late 1990s

Group	TL	Biomass (tons km ⁻² year ⁻¹)	P/B (year ⁻¹)	Q/B (year ⁻¹)	EE (proportion)	P/Q (proportion)
Phytoplankton	1	203.664	35.7	–	0.644	–
Microzooplankton	2.06	3.192	482	1 928	0.999	0.25
Mesozooplankton	2.53	17.5	40	133.333	0.999	0.3
Macrozooplankton	2.61	19.989	13	31.707	0.999	0.41
Gelatinous zooplankton	3.23	245.66	0.44	1.467	0.151	0.3
Macrobenthos	1.51	192.111	1.2	10	0.999	0.12
Sardine	2.65	0.443	1.35	14	0.999	0.096
Anchovy	3.03	0.15	1.8	18	0.959	0.1
Pelagic goby	3.22	17.368	1.8	12	0.999	0.15
Mesopelagics	3.56	9.575	1.23	12.3	0.999	0.1
Cephalopods	3.99	1.935	5	15	0.999	0.333
Other small pelagics	3.52	7.995	0.958	9.349	0.93	0.102
Small horse mackerel	3.56	4.429	1.2	10	0.999	0.12
Large horse mackerel	3.56	5.362	0.8	5.333	0.606	0.15
Large pelagics	4.57	0.188	0.5	5	0.276	0.1
Small <i>M. capensis</i>	4.03	2.068	2	13.333	0.999	0.15
Large <i>M. capensis</i>	4.5	3.328	1.228	7.824	0.934	0.157
Large <i>M. paradoxus</i>	4.31	1.894	1.14	7.278	0.666	0.157
Benthic-feeding demersals	3.59	2.681	1	5	0.97	0.2
Pelagic-feeding demersals	3.97	1.341	1	5	0.975	0.2
Chondrichthyans	3.35	0.36	0.5	3.333	0.667	0.15
Seabirds	4.36	0.004	0.156	120.3	0.608	0.001
Seals	4.65	0.253	0.29	18.25	0.171	0.016
Cetaceans	4.29	0.019	0.15	7.418	0	0.02
Detritus	1	840	–	–	0.555	–

TL: Trophic level

P/B: Production/biomass

Q/B: Consumption/biomass

EE: Ecotrophic efficiency

P/Q: Production/consumption

App. Table II: Diet matrix of the balanced ECOPATH model for the northern Benguela in the late 1990s. Predators are numbered along the horizontal axis

Prey	Proportion of prey in diet																							
	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	
1. Phytoplankton	0.47	0.5	0.6	0.02		0.56	0.33	0.007			0.007													
2. Microzooplankton	0.06	0.5				0.08	0.04	0.006			0.006										0.003			
3. Mesozooplankton			0.4	0.62		0.18	0.31	0.255	0.6	0.02	0.774	0.7	0.15		0.1				0.18		0.003		0.05	
4. Macrozooplankton				0.12		0.18	0.32	0.413	0.4	0.54	0.17	0.3	0.65	0.2	0.4	0.1	0.3	0.06	0.5		0.043		0.416	
5. Gelatinous zooplankton				0.04							0.026													
6. Macrobenthos					0.08			0.325					0.1					0.01	0.4	0.4				
7. Sardine										0.005				0.02	0				0.01	0.01	0.038	0	0.009	
8. Anchovy										0.001				0	0				0	0.01	0.149	0	0.005	
9. Pelagic goby										0.123			0.08	0.05	0.5	0.4	0.07		0.04	0.03	0.465	0.4	0.066	
10. Mesopelagics										0.123			0.02	0.13	0	0.1	0.26	0.03	0.04	0.03	0.05	0.1		
11. Cephalopods										0.01				0.3	0.1	0	0.07	0.27	0.15	0.1	0.045	0	0.254	
12. Other small pelagic										0.068				0.08	0	0.1	0.14	0.03	0.04	0.07	0.083	0	0.004	
13. Small horse mackerel										0.1				0.03	0.1						0.005	0.1	0.05	
14. Large horse mackerel															0			0.01	0.02	0.02		0.1	0.043	
15. Large pelagics																				0	0.003	0.2	0.019	
16. Small <i>M. capensis</i>										0.01				0.02	0.1	0.01				0	0.002	0.2		
17. Large <i>M. capensis</i>														0.03	0.1	0.03		0.03		0	0.052	0.2	0.024	
18. Large <i>M. paradoxus</i>														0.05	0	0.03		0.03	0.02	0	0.046		0.024	
19. Benthic-feeding demersals															0	0.08	0.04		0.05					
20. Pelagic-feeding demersals														0.1	0	0.04	0.01	0.01	0.02	0.015	0			
21. Chondrichthyans																				0.1			0	
22. Seabirds																							0.003	
23. Seals																						0.001	0.033	
24. Cetaceans																								
25. Detritus	0.47										0.017													
Import				0.2	0.05																0			
				0.87																	0.16			

App. Table III: Average landings estimated for the period 1995–1999 for the different fisheries considered in the balanced ECOPATH model of the northern Benguela for the late 1990s

Group	Average landings (tons km ⁻²)							
	Pelagic	Midwater	Demersal	Tuna	Linefish	Sealing	Squid	Crab
Macrobenthos	0	0	0	0	0	0	0	0.01
Sardine	0.189	0	0	0	0	0	0	0
Anchovy	0.01	0	0	0	0	0	0	0
Pelagic goby	0.046	0	0	0	0	0	0	0
Cephalopods	0	0	0	0	0	0	0.001	0
Other small pelagics	0.03	0	0	0	0	0	0	0
Small horse mackerel	0.345	0	0	0	0	0	0	0
Large horse mackerel	0	1.307	0.02	0	0	0	0	0
Large pelagics	0	0.004	0	0.009	0.004	0	0	0
Large <i>M. capensis</i>	0	0	0.267	0	0	0	0	0
Large <i>M. paradoxus</i>	0	0	0.455	0	0	0	0	0
Benthic-feeding demersal	0	0	0.083	0	0	0	0	0
Pelagic-feeding demersal	0	0.041	0	0	0	0	0	0
Seals	0	0	0	0	0	0.005	0	0
Total	0.62	1.352	0.825	0.009	0.004	0.005	0.001	0.01

App. Table IV: Summary statistics from the northern Benguela ecosystem model for the period 1995–1999

Parameter	Model output (ton km ⁻² year ⁻¹)
Total biomass (excluding detritus)	741.509*
Sum of all consumption	12 009.35
Sum of all exports	2 682.454
Sum of all respiratory flows	6 259.718
Sum of all flows into detritus	5 514.971
Total system throughput	26 46
Sum of all production	10 194
Net system production	1 011.087
Total catches	2.826
Mean trophic level of the catch	3.71

* tons km⁻²