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Spatialized ecosystem indicators in the southern Benguela

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Based on published distribution maps of 15 key fish species, foraging areas of three top predators during their breeding season, and fishing grounds of the main commercial fleets in the southern Benguela ecosystem, seven spatialized ecosystem indicators are derived: biodiversity, connectivity, mean ratio of fished area and area of distribution by species, exploited fraction of the ecosystem surface area, total catch per fished area by fishery, mean bottom depth of catches, and mean distance of catches from the coast. These indicators are compared and their suitability for an ecosystem approach to fisheries is discussed. The first two indicators characterize the ecosystem; the others are pressure indicators that are also compared with conventional (catch rate) indices of abundance.

Keywords: ecosystem indicators, exploitation indicators, GIS, interactions, southern Benguela, upwelling system.

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Introduction

To implement an ecosystem approach to fisheries (EAF), a better understanding of the functioning of marine ecosystems is needed (Larkin, 1996; Jennings and Kaiser, 1998; Hall, 1999; Gislason *et al.*, 2000; Sinclair and Valdimarsson, 2002; Sinclair *et al.*, 2002; Moloney *et al.*, 2004). One way of achieving this is to make use of indicators, i.e. pointers that can be used to reveal or to monitor conditions and trends in the fisheries sector, as well as in the marine environment (FAO, 1999; Smeets and Weterings, 1999; Garcia and Staples, 2000; Garcia *et al.*, 2000; Sainsbury and Sumaila, 2001). It is our view that poor understanding of the functioning of marine ecosystems is largely due to the lack of spatial information on the interactions between their main components, including the physical environment, plankton, other invertebrates, fish, top predators, and the fisheries. Using the southern Benguela ecosystem as a case study, we concentrate here on spatial interactions between fish, top predators, and fisheries to explore potentially suitable indicators for characterizing ecosystems, their level of exploitation, and their health.

The southern Benguela is assumed to extend from the coastline to a depth of 2000 m from 28°S (south of the Lüderitz permanent upwelling cell) along the west coast of South Africa and to 28.5°E (east of East London; Figure 1). It covers an area of 250 000 km^2 (UTM projection) and incorporates the Agulhas Bank and the west coast of South Africa (Shannon and O'Toole, 1998; Shannon *et al.*, 2003). Previous work (Drapeau *et al.*, 2004; Pecquerie *et al.*, 2004) investigated potential interactions between key commercial species of the ecosystem by mapping and quantifying overlaps in distribution. Here, we complement those studies by adding three top predators and two major fisheries, and derive seven spatialized ecosystem indicators: biodiversity, connectivity, mean ratio of fished area to area of distribution by species, exploited fraction of the ecosystem surface area, total catch per exploited area per fishery, mean bottom depth of catches, and mean distance of the catches

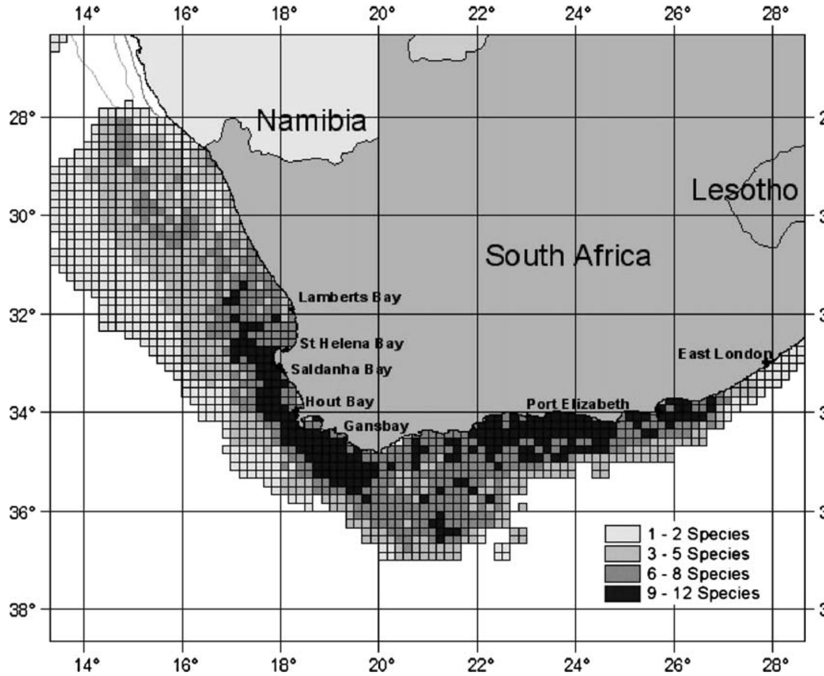

Figure 1. Map of the relative number of species of the 13 main species by combining survey and commercial data sources, 1985−2001.

from the coast. The seven indicators were generated on the basis of three types of data: the area of distribution of the exploited species, the foraging area of predators, and the exploited area.

Methods

Data

The 13 key exploited species comprise three small pelagic fish (sardine *Sardinops sagax*, anchovy *Engraulis encrasicolus*, round herring *Etrumeus whiteheadi*), two medium-sized pelagic fish (horse mackerel *Trachurus trachurus capensis*, chub mackerel *Scomber japonicus*), chokka squid (*Loligo vulgaris reynaudii*), three demersal taxa (kingklip *Genypterus capensis*, hake *Merluccius* spp., silver kob *Argyrosomus inodorus*), and four large pelagic fish (snoek *Thyrsites atun*, albacore *Thunnus alalunga*, bigeye tuna *Thunnus obesus*, yellowfin tuna *Thunnus albacares*). Density distribution maps for each species were constructed by combining six sources of data in a $10' \times 10'$ cell grid GIS (for details, see Pecquerie *et al.*, 2004): acoustic and demersal research surveys (1988−2001), and commercial pelagic purse-seine, trawl (including coastal and offshore bottom trawl as well as midwater trawl fisheries), and hake- and tuna-directed longline fisheries (1994−2001). Only 95% of the biomass has been represented in the maps to exclude areas of low abundance or sporadic presence (cf. Figure 3 of Pecquerie *et al.*, 2004) and to prevent overestimation of the area of distribution ($DistA_i$ of species i). Because the abundance of the small pelagic species has changed extensively since 1980 (Shannon *et al.*, 2003), two periods (1985−1989, 1990−2001) are defined in some instances.

Top predators were selected on the basis of their numerical abundance and data availability, and include two species of seabird (Cape gannet *Morus capensis*, African penguin *Spheniscus demersus*), and one marine mammal (Cape fur seal *Arctocephalus pusillus pusillus*). All three are colonial and establish breeding colonies (mainly on islands) along the South African coast. Their distribution at sea has been studied by satellite tracking of tagged individuals (Grémillet *et al.*, 2004). Tags were attached to the backs of the animals, and information on their location was provided by satellite geo-referencing. Tracking data (collected in the years 1993−2001) were available only for the most important colonies: four seal (36 animals), eight penguin (16 birds), and three gannet colonies (15 birds). To establish the foraging area of adults from a given colony, the tagged animals were assumed to be representative of all individuals of that colony, even if just one was tagged. Animals were tagged mainly during the breeding season, with a few exceptions for seals. During that season, adults tend to remain adjacent to their colony, because of mating or parental care activities. Initially, we considered that conservative foraging areas might be represented by a circle around each colony with a radius representing the maximum distance recorded. However, that area sometimes appeared far too large relative to the tracking observations, because of an asymmetrical distribution. In such cases, we used two radii, one on each side of the colony, plus a bathymetric limit to estimate the foraging area ($ForA_i$ of species i). When appropriate, the area was modified manually to reflect available observations and a common-sense perception of the real distribution. When just one animal was tagged, an arbitrary 10% of the largest distance recorded from the colony was added to the radius in computing the foraging area. ArcView 3.2 GIS software was used to visualize, explore, query, and analyse the data.

Three types of commercial fishery were considered: pelagic purse-seine, trawl, and longline fisheries. These three fisheries yielded on average 98% of the total catches made in the southern Benguela ecosystem in the 1990s. The exploited area ($ExplA_{i,j}$ for fishery i in year or period j) was estimated by summing the surface of every individual grid cell where at least one catch was taken. Although a sensitivity analysis indicated that the absolute value of ExplA was sensitive to setting a threshold for a minimum number of catches recorded per grid cell, trends and interannual variations remained unaffected. Prior to 1985, ExplA for the pelagic fishery was estimated after digitizing a series of maps of fishing effort distribution in 1951, 1955, 1958, 1964, and 1974, given by Crawford (1986). Data are unavailable for the demersal fishery prior to 1985. Depth is recorded for demersal trawls in skipper logbooks, but not for the pelagic fishery. Because the geo-referencing system of the pelagic fishery is based only on a 10′ grid, the ETOPO2 bathymetry from NOAA with a spatial resolution of 3 km was used to estimate depth at the centre of every grid cell, and this estimate of bottom depth was linked to individual catches (BDC). Similarly, distance from the coast of pelagic and demersal catches (DCC) was estimated by calculating the distance between the centre of a grid cell and the nearest point of the coastline.

Indicators

An index of spatial biodiversity ISB_j during period j was defined as the average relative number of species (exploited and top predators):

$$ISB_j = (100/n) \times \sum_{g=1}^{n} s_{g,j}/S \qquad (1)$$

where ($s_{g,j}$) is the number of species per grid cell g, S the total number of species in the databases, and n is the number of grid cells with observations. This index is associated with the map of the relative number of species per grid cell ($s_{g,j}/S$).

A spatial-overlap (connectivity) index (OI_j) during each period j is defined as the average of relative overlapping areas ($ROA_{a/b,j}$) between any two of the three areas described above ($DistA_{i,j}$, $ForA_{i,j}$, $ExplA_{i,j}$) for species/species or species/fishery pairs (a, b) in the ecosystem characterized by a trophic (or exploitational) relationship (predation, competition), based on available literature and expert judgement (Table 1). ROA is simply the ratio of the overlapping area between two distributions (D_a and D_b) to the total area occupied by these two distributions together (Figure 2):

$$ROA_{a/b,j} = 100 \times \left(D_{a,j} \cap D_{b,j}\right) / D_{a,j} \cup D_{b,j} \qquad (2)$$

where ∩ and ∪ are standard symbols for intersection (overlap) and union (total area), respectively. Although each ROA may be considered individually as a potential indicator of interaction (Drapeau *et al.*, 2004), the mean of all ROAs relating to interactions (Table 1) may serve as an ecosystem indicator of overall degree of interaction (OI_j).

The ratio of fished area and distribution area by species (RED_j) is defined as

$$RED_j = 100 \times \left(DistA_{i,j} \cap ExplA_{i,j}\right) / DistA_{i,j} \qquad (3)$$

Table 1. Qualitative representation of predation (P, heavy; p, medium) and competition (C, strong; c, medium) between all components considered, based on expert judgement and ECOPATH estimated flows (empty cells, no or minor predation/competition; * predation on or competition among mainly juvenile stages; TL, trophic level; after Drapeau *et al.*, 2004, updated by incorporation of the longline fishery).

Prey species			Predator species															
		TL	Sd	An	Rh	Hm	Cm	Ck	Kk	Hk	Sk	Sn	Se	Pe	Gn	Pf	Df	Lf
Sardine	Sd	3.0		C	C	pc	pc*	pc		pc*	p	Pc*	P	P	P	P		
Anchovy	An	3.5	C		C	pc	pc*	p		pc*	P	Pc*	P	P	P	P		
Round herring	Rh	3.6	C	C		pc	pc*	p		pc*		Pc*	p	p	p	P	p	
Horse mackerel	Hm	3.7	c	c	c		c	p	p	Pc		pc	p	P	P	p	P	
Chub mackerel	Cm	3.9	c*	c*	c*	c				p		pc			p	p	p	
Chokka squid	Ck	3.8	c					P		Pc		p	p				P	
Kingklip	Kk	3.4				c	c			c		c					P	p
Hake	Hk	4.4	c*	c*	c*	c	c	pc	Pc	p*		pc	p				P	P
Silver kob	Sk	4.5															P	
Snoek	Sn	4.5	c*	c*	c*	c	c		c	c							P	
Seal	Se	4.7									c	C		C	C		C	
Penguin	Pe	4.4									c	C	p					
Gannet	Gn	4.4										C	p	C			c	
Pelagic fleet	Pf	-										C	C	C	C			
Demersal fleet	Df	-											c			c		
Hake longline fleet	Lf	-											c				C	

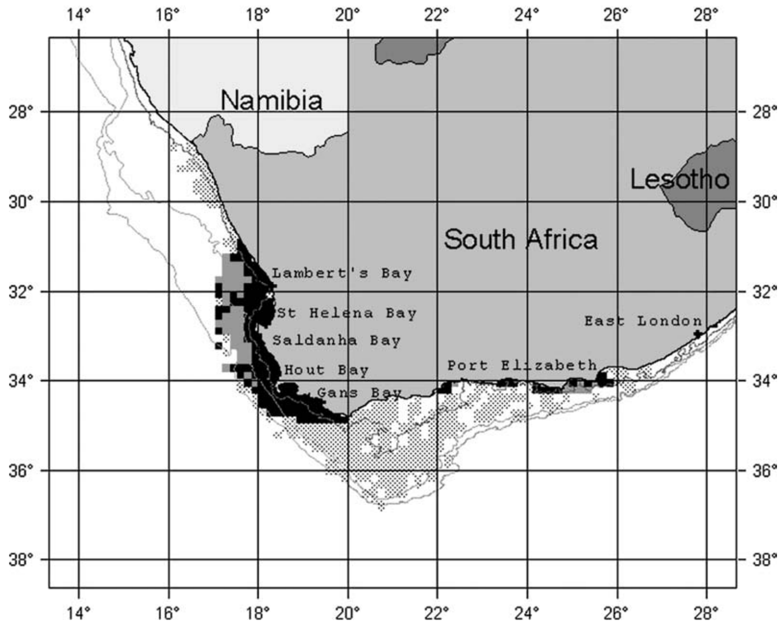

Figure 2. Overlap between the distribution area of anchovy and of the pelagic fleet (black, area of overlap; dotted, anchovy only; grey, pelagic fleet only): Relative Overlapping Index (ROA) is the number of black cells divided by the number of black + dotted cells.

Mean RED values (MRED) may be averaged over species exploited by the same fishery, and an overall value may also be computed for all fleets and species.

The exploited fraction of ecosystem surface ($EFE_{i,j}$) during each year was computed for each fishery as the relative overlapping area between the exploited area ($ExplA_{i,j}$) and the invariant ecosystem area (EA = 250 000 km^2):

$$EFE_{i,j} = 100 \times \left(ExplA_{i,j} \cap EA\right)/EA \quad (4)$$

An overall EFE can be computed by considering all fisheries exploiting the ecosystem (three in this case):

$$EFE_j = 100 \times \left(\left(ExplA_{1,j} \cup ExplA_{2,j} \cup ExplA_{3,j}\right) \cap EA\right)/EA \quad (5)$$

The total catch per exploited area ($CPEA_{i,j}$) is defined as the ratio of total annual catches ($C_{i,j}$) and annually exploited area by fishery:

$$CPEA_{i,j} = C_{i,j}/ExplA_{i,j} \quad (6)$$

The overall annual $CPEA_j$ for the three fisheries is then

$$CPEA_j = \sum_{i=1}^{3} C_{i,j}/\left(ExplA_{1,j} \cup ExplA_{2,j} \cup ExplA_{3,j}\right) \quad (7)$$

The mean bottom depth of catches ($MBDC_{i,j}$) is the average BDC per fishery and per year, weighted by the total catch (C) in each grid cell g:

$$MBDC_{i,j} = \sum_{g=1}^{n} \left(BDC_{i,j,g} \times C_{i,j,g}\right) / \sum C_{i,j,g} \quad (8)$$

The mean distance of catches from the coast ($MDCC_j$) is the average DCC, weighted by the total catch in each grid cell, equivalent to Equation (8).

Results

Using all available data on exploited species (1985–2001), estimated ISB was 38%. The spatial distribution of the relative number of species per grid cell is patchy, with higher biodiversity inside two coastal zones located on the west and south coasts (Figure 1).

ROAs have been computed for the relevant interacting pairs of species (excluding empty cells in Table 1) in terms of overlapping areas of distribution, foraging, and exploitation within the ecosystem (Figure 3). Based on these values, the value of the global indicator OI is 23%.

The RED values for pelagic species did not exceed 40% (e.g. Figure 1), whereas for demersal species they were always >70% (silver kob 98%; Figure 4). MRED values were 31% for the pelagic fleet, 78% for the demersal fleet, and overall (all fleets combined) 58%.

The pelagic fishery exploited just 0.8% of the ecosystem surface area in the early 1950s, whereas >20% is currently exploited (Figure 5). Quantitative historical data for the demersal fisheries are unavailable, but the exploitation area was likely comparatively small prior to the late 1950s because at that time demersal stocks were mainly exploited

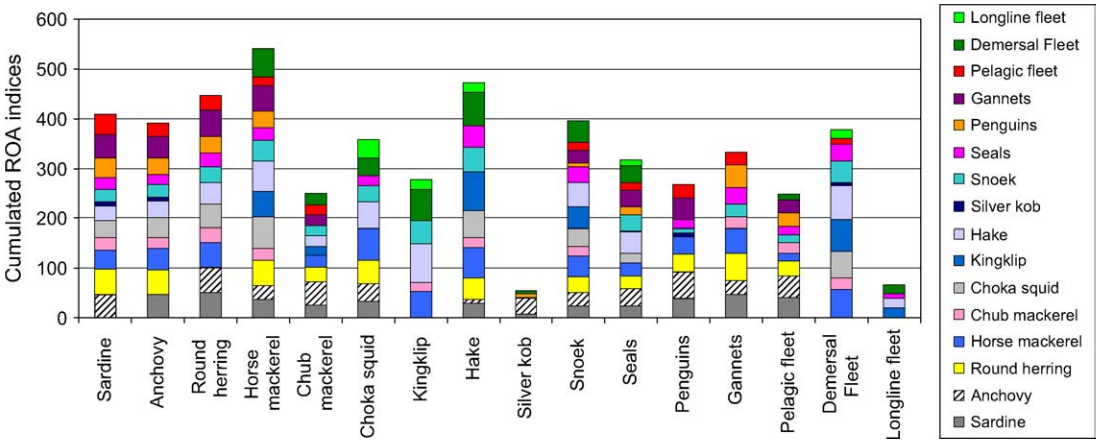

Figure 3. Cumulative Relative Overlapping Indices (ROA) for the major species and fleets of the southern Benguela ecosystem.

by small local fleets. In the early 1960s, foreign fleets started to operate off South Africa, then increased rapidly in size (Payne, 1995), so that by the late 1980s, up to 90% of the entire area may be considered to have been exploited. In 1977, South Africa declared a 200 nautical mile Fishing Zone and excluded many of the foreign fleets. This presumably resulted in a reduction in EFE.

The area exploited by the pelagic fishery expanded from about 2030 km^2 in 1951 to 30 000 km^2 in 1974. The catch per exploited area decreased rapidly in the 1950s (from about 100 $t\,km^{-2}$ in 1950 to 20 $t\,km^{-2}$ in 1956), then decreased progressively further until the end of the 1980s (Figure 6). There is no available quantitative information for the effort distribution for the years 1975–1986, but the stability of total pelagic landings during this period suggests that the exploitation area was probably quite uniform during those 12 years. In recent years, CPEA has tended to stabilize at around 10 $t\,km^{-2}$. The CPEA of the demersal fishery varied from 0.65 to 1.1 $t\,km^{-2}$ between 1985 and 2001, with no linear trend (Figure 6). The combined CPEA of the two fisheries is not shown, but because the volume of pelagic landings is much greater, it is completely dominated by the pelagic fishery.

The MBDC of the pelagic fishery displays a trend towards deeper water, from about 47 m at the end of the 1980s to >85 m in recent years, associated with an increase in MDCC from 10 to 15 km (Figure 7a). The MBDC and MDCC of the demersal fishery have been more variable, but in general show the opposite trend (Figure 7b). The MBDC of all fisheries combined increased from 180 to 225 m, whereas the corresponding MDCC was variable, but seems to have decreased from 50 to 40 km.

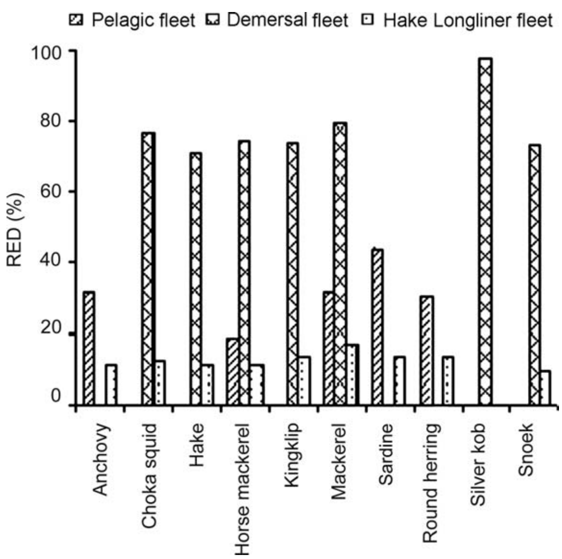

Figure 4. Ratio of fished area and distribution area by species (RED) for pelagic and demersal fleets (average 1985–2001).

Discussion

Although some important species (plankton, mesopelagic fish, sharks) and fisheries (handline, squid jig, coastal and small-scale) are still missing from the analysis, and despite the partial overlap between the fisheries (1985–2001) and top predator data (1993–2001), the spatialized indices provide a framework for evaluating ecosystem state and fishing impacts. They can be split into two categories: (i) indicators characterizing the ecosystem (ISB and OI); (ii) pressure indicators, "representing the pressure on the environment exerted by different driving forces" (Grieve *et al.*, 2003; MRED, EFE, CPEA, MBDC, and MDCC).

ISB is an indicator of the average biodiversity per cell, regardless of the total number of species in the ecosystem. Nevertheless, the same mean value may be obtained for different patterns of patchiness, and the indicator should be

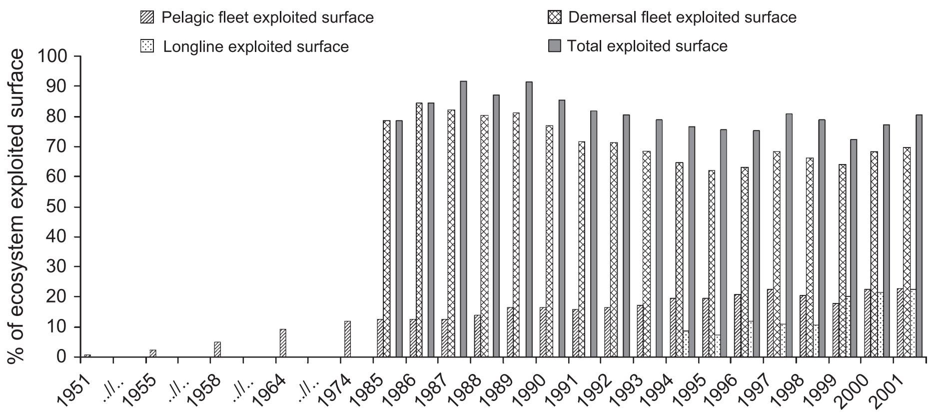

Figure 5. Exploited fraction of the ecosystem surface area (EFE) for demersal (1985–2001), pelagic (1951–2001), and longline (1994–2001) fleets separately and combined.

complemented with a map that allows for computation of a spatial indicator of dispersion of biodiversity (e.g. fractal dimension; Hastings and Sugihara, 1993). Moreover, ISB may be affected by the subset of species selected.

To test the sensitivity of ISB to species selection, we used all species (602) identified in the demersal trawl surveys and computed the value for the two periods. ISB_{85-89} was 7.5% and ISB_{95-01} 6.1%, compared with 38% when using only 13 species (tunas excluded), which appears to be due to the incorporation of rare species.

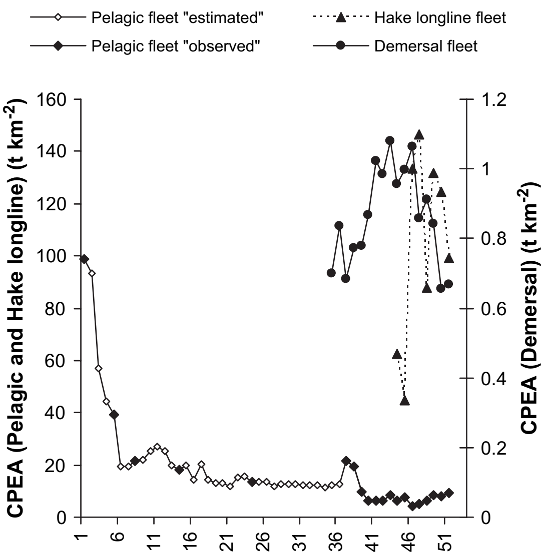

Figure 6. Total catch per exploited area (CPEA) in the pelagic (1951–2001), demersal (1985–2001), and hake longline (1994–2001) fleets. Catches in the pelagic fishery are known for all years, whereas exploited areas have been partly interpolated (circles) based on years for which such data were available (dots) and assuming a linear trend between two measurements of exploited area.

Weighting ISB by the abundance would be expected to limit this effect. In addition, the three corresponding maps are quite different (Figures 1 and 8), even between ISB_{85-89} and ISB_{95-01}, despite the small difference in indicator values. During the first period, there were no clear spots of high biodiversity except along the 200 m isobath (Figure 8a). During the second period, the area along the south coast also displayed high biodiversity (Figure 8b), possibly caused by intrusion of Indian Ocean warm water onto the Agulhas Bank favouring the presence of subtropical species. ISB might be a useful indicator for comparing spatial biodiversity across ecosystems, but standardization of the number/types of non-exploited species included in the analyses would require careful attention. Also, annually integrated ISB in combination with changes in distribution of number of species might be used to track differences in spatial biodiversity associated with the impacts attributable to human activity.

OI reflects the overlap among the areas occupied by the fish, or among them and the forage areas of their predators, including fishers. Connectivity is an important property of ecosystems that allows the different components of the ecosystem potentially to interact and thus respond as a whole to internal or external perturbations. A high OI underlines the importance of an ecosystem approach to fisheries, because the effect of exploitation will propagate to many other species and fisheries. OI might be considered a measurable form of the Mixed Trophic Impact indicator (Ulanowicz and Puccia, 1990), calculated from models as the net trophic effect of one species or fishery on every other species and fishery. The high ROA value for horse mackerel suggests that fishing or environmental impacts on that species have a stronger effect on the ecosystem as a whole than would have impacts on species with substantially lower ROA values, such as silver kob. Indeed, trophic models of the southern Benguela show that horse mackerel has large net trophic impacts on species and fisheries (Shannon *et al.*, 2003). In the southern Benguela, the OI global value of

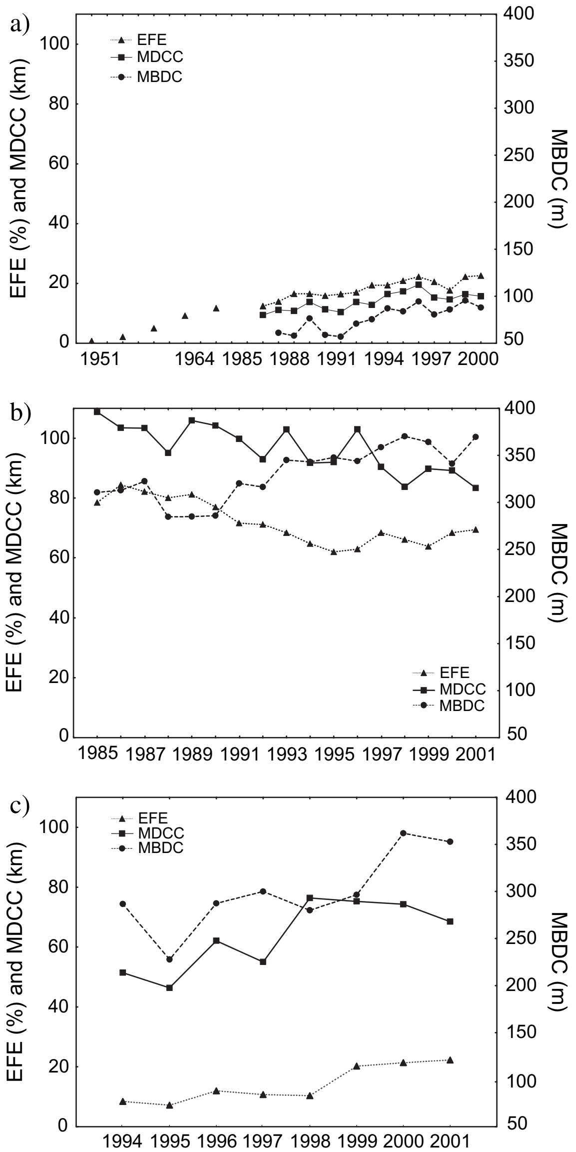

Figure 7. Mean bottom depth of catches (MBDC), mean distance from the coast of catches (MDCC), and the exploited fraction of ecosystem surface area (EFE) for (a) the pelagic fishery (1951–2001), (b) the demersal fishery (1985–2001), and (c) the two combined (1994–2001).

22.5% reflects a medium level of connectivity with other species, likely attributable to the absence or limited connection between some of the sedentary coastal subtropical species inhabiting the south or east coasts. Current survey and tagging data are insufficient to provide good estimates of $DstA_{i,j}$ and $ForA_{i,j}$ on an annual basis, so accurate annual estimates of OI cannot be expected.

The five pressure indicators represent fishing pressure on the ecosystem in all its spatial aspects. Pressure indicators may be easier to estimate and interpret, and less prone to bias than conventional indices such as standardized fishing effort or average catch per unit effort (cpue). However, they cannot replace conventional indicators, because such crucial factors as changes in fishing unit capacity and technological improvements are not taken into account. Rather, they add other dimensions.

The mean ratio of fished and distribution area by species (MRED) may be useful for deciding on the need to implement marine protected areas (MPA) or equivalent measures. If the fisheries for most species operate over nearly their entire area of distribution, such management measures may be essential. In the southern Benguela, RED is close to 70% for many demersal species, despite several MPAs having been implemented progressively (MPAs cover 19% of the ecosystem today). Without their implementation, RED would likely be close to 100% for all species. Interpretation may be difficult for highly migratory species, because such species may be easily overexploited with low values of RED.

The exploited fraction of the ecosystem surface (EFE) indicates how the fishery exploits the ecosystem spatially. In contrast to OI and MRED, EFE may be computed reliably on an annual basis, which should allow ecosystem management to be more reactive, for instance by limiting fishing areas.

The total catch per exploited area per fishery (CPEA) has been used commonly for demersal fisheries (e.g. Craig *et al.*, 2004), but less so for migratory (pelagic) fisheries or for fisheries combined. Because most production and exploitation in marine ecosystems occurs within the euphotic and bottom layers, an index based on surface area appears preferable to one computed on the basis of volume of water. CPEA is an absolute ($t\,km^{-2}$) rather than a relative index, and therefore should allow for comparison between ecosystems or for indicating trends in productivity and/or exploitation rate of the ecosystem. The marked drop in CPEA in the pelagic fisheries of the Benguela from 1951 to 1956 reflects the expansion/development of these, while its stabilization in the recent period may reflect the effectiveness of the operational management procedures implemented for anchovy and sardine beginning in 1994 (De Oliveira, 2002). The increased values observed in 1987 and 1988 reflect the large (∼600 000 t) anchovy catch in those years, most of which was made in relatively shallow water close to the coast. The available time-series for the demersal fishery is too short to draw any such conclusions.

Mean bottom depth of catches (MBDC) and mean distance of catches from the coast (MDCC) may reflect the evolution of fisheries, because fishers first tend to exploit, and deplete, the more accessible and productive coastal areas before moving farther afield and into deeper water, where more sophisticated fishing techniques are required (Pauly *et al.*, 2003). The deeper areas of the southern Benguela have been exploited by both pelagic and demersal fisheries in recent years (Figure 7). Histograms (not shown) of fishing effort per depth class indicate that this trend is not

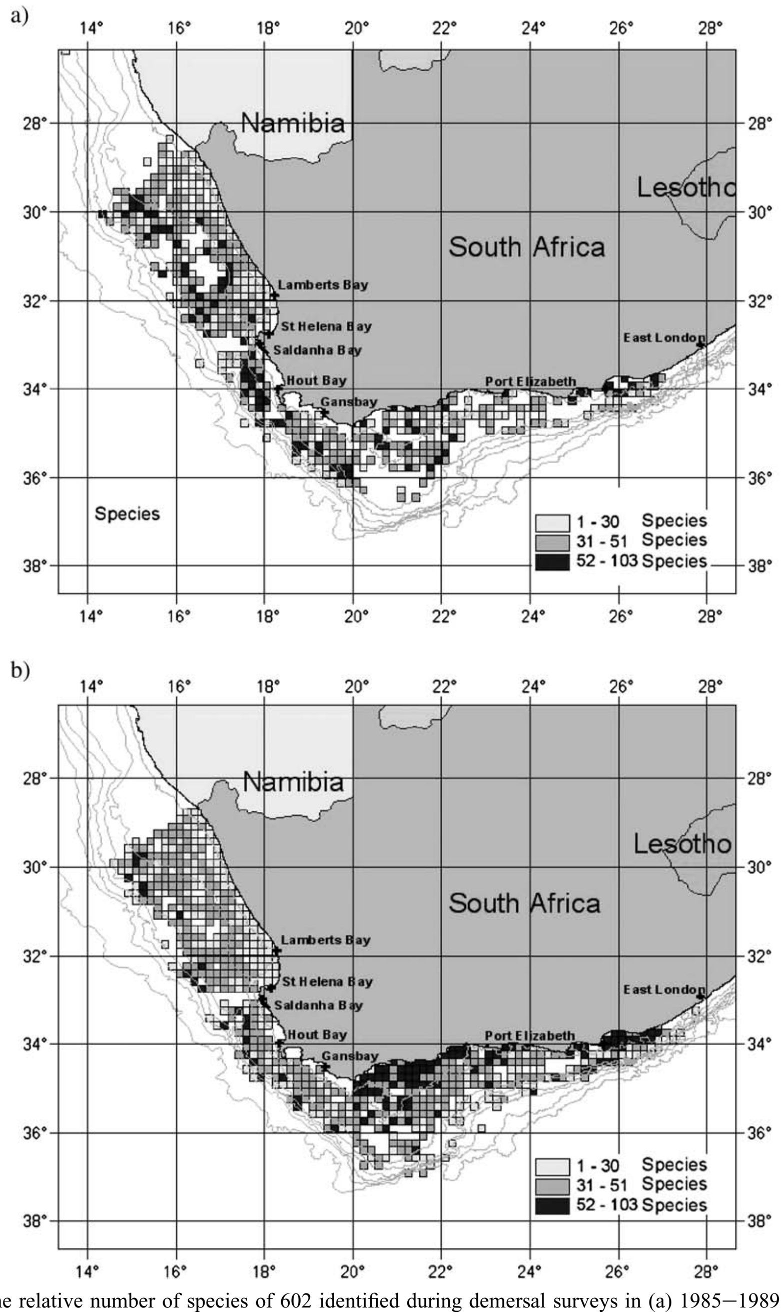

Figure 8. Map of the relative number of species of 602 identified during demersal surveys in (a) 1985–1989 and (b) 1990–2001.

due to a decrease of effort in the coastal area, but rather to expansion offshore. The trends in both time-series may be considered as an index of the overall type of exploitation of the ecosystem, whereas local variations within the time-series reflect resource availability, which is often related to interannual variations in abundance of the dominant species. For example, in the pelagic fishery, the increasing trend in MBDC and MDCC over the period 1987–2001 (from 5% to 35%) most likely reflects the increasing proportion of sardine, which are generally caught farther offshore than anchovy. In addition, the larger number of holders of fishing rights from 2000 onward (>100 compared with <20 before) was associated with a subsequent increase in the number of smaller vessels, which could explain the recent decline in MDCC. Finally, the large increase in abundance of both main small pelagic target species in recent years (Griffiths *et al.*, 2004) increased the probability of finding fish closer to home than before.

Rather than ranking different ecosystem indicators for electing the best, more might be learned about ecosystem state and fishing impacts by trying to interpret the similarity and discrepancies among them. For instance, the high correlation among EFE, MBDC, and MDCC for the pelagic fleet ($|0.87| < r < |0.97|$) appears to reflect the expansion of that fishery, whereas the correlation between those three indicators and both CPEA and cpue was weaker ($|0.64| < r < |0.71|$). The latter seems mainly to be a consequence of the exceptionally good recruitment of anchovy and sardine from 2000 on. For the demersal fishery, the correlation between all indices is weaker ($|0.49| < r < |0.78|$), indicating that each indicator responds differently to the development in the fishery.

The seven indicators proposed meet the three main criteria often applied to the evaluation of indicators, i.e. simplification, quantification, and communication (Grieve *et al.*, 2003): easy to compute and interpret; quantitative; and expressed in units (% for the first four; $t\,km^{-2}$, m and km for the others) that facilitate communication. Nonetheless, the practical use of OI and MRED is limited by data for computing them on an annual basis, and their value depends on the time scale of measurement. Therefore, OI and MRED may be more useful as "scientific" indicators for comparing different ecosystems under controlled applications. The others are more appropriate for ecosystem monitoring and the effects of policy on an annual basis. EFE, for instance, could be translated to a target reference point and operational objective in an ecosystem approach to fisheries management.

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References

Craig, J. F., Halls, A. S., Barr, J. J. F., and Bean, C. W. 2004. The Bangladesh foodplain fisheries. Fisheries Research, 66: 271–291.

Crawford, R. J. M. 1986. Long term changes in the distribution of fish catches in the Benguela. *In* Long Term Changes in Marine Fish Populations, pp. 449–480. Ed. by T. Wyatt, and M. G. Larrañeta. Consejo Superior de Investigaciones Cientificas, Vigo.

De Oliveira, J. A. A. 2002. The development and implementation of a joint management procedure for the South African pilchard and anchovy resources. PhD thesis, University of Cape Town. 319 pp.

Drapeau, L., Pecquerie, L., Fréon, P., and Shannon, L. J. 2004. Quantification and representation of potential spatial interactions in the southern Benguela ecosystem. African Journal of Marine Science, 26: 141–159.

FAO. 1999. Indicators for sustainable development of marine capture fisheries. FAO Technical Guidelines for Responsible Fisheries, 8. 68 pp.

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

Garcia, S. M., Staples, D. J., and Chesson, J. 2000. The FAO guidelines for the development and use of indicators for sustainable development of marine capture fisheries and an Australian example of their application. Ocean and Coastal Management, 43: 537–556.

Gislason, H., Sinclair, M., Sainsbury, K., and O'Boyle, R. 2000. Symposium overview: incorporating ecosystem objectives within fisheries management. ICES Journal of Marine Science, 57: 468–475.

Grémillet, D., Dell'Omo, G., Ryan, P. G., Peters, G., Ropert-Coudert, Y., and Weeks, S. J. 2004. Offshore diplomacy, or how seabirds mitigate intra-specific competition. A case study based on GPS tracking of Cape gannets from neighbouring breeding sites. Marine Ecology Progress Series, 268: 265–279.

Grieve, C., Sporrong, N., Coffey, C., Moretti, S., and Martini, N. 2003. Review and gap analysis of environmental indicators for fisheries and aquaculture. Institute for European Environmental Policy Report, London. 61 pp.

Griffiths, C. L., Van Sittert, L., Best, P. B., Brown, A. C., Cook, P. A., Crawford, R. J. M., David, J. H. M., Davies, B. R., Griffiths, M. H., Hutchings, K., Jerardino, A., Kruger, N., Lamberth, S., Leslie, R. W., Melville-Smith, R., Tarr, R. J. Q., and van der Lingen, C. D. 2004. Impacts of human activities on marine animal life in the Benguela: a historical overview. Oceanography and Marine Biology. An Annual Review, 42: 303–392.

Hall, S. J. 1999. The Effect of Fishing on Marine Ecosystems and Communities. Fish Biology and Aquatic Resources Series 1. Blackwell, Oxford. 274 pp.

Hastings, H. M., and Sugihara, G. 1993. Fractals. Oxford Science Publications, Oxford. 235 pp.

Jennings, S., and Kaiser, M. J. 1998. The effects of fishing on marine ecosystems. Advances in Marine Biology, 34: 201–352.

Larkin, P. A. 1996. Concepts and issues in marine ecosystem management. Reviews in Fish Biology and Fisheries, 6: 139–164.

Moloney, C. L., van der Lingen, C. D., Hutchings, L., and Field, J. G. 2004. Contributions of the Benguela Ecology Programme to pelagic fisheries management in South Africa. African Journal of Marine Science, 26: 37–51.

Pauly, D., Alder, J., Bennett, E., Christensen, V., Tyedmers, P., and Watson, R. 2003. The future for fisheries. Science, 302: 1359–1361.

Payne, A. I. L. 1995. Cape hakes. *In* Oceans of Life off Southern Africa, 2nd edn, pp. 136–147. Ed. by A. I. L. Payne, and R. J. M. Crawford. Vlaeberg, Cape Town.

Pecquerie, L., Drapeau, L., Fréon, P., Coetzee, J. C., Leslie, R. W., and Griffiths, M. H. 2004. Distribution patterns of key fish species of the southern Benguela ecosystem: an approach combining fishery-dependent and fishery-independent data. African Journal of Marine Science, 26: 115–139.

Sainsbury, K., and Sumaila, U. R. 2001. Incorporating ecosystem objectives into management of sustainable marine fisheries, including ‘best practice’ reference points and use of marine protected areas. *In* Responsible Fisheries in the Marine Ecosystem, pp. 343–361. Ed. by M. Sinclair. CABI Publishing, Oxford. 448 pp.

Shannon, L. J., Moloney, C. L., Jarre, A., and Field, J. G. 2003. Trophic flows in the southern Benguela during the 1980s and the 1990s. Journal of Marine Systems, 39: 83–116.

Shannon, L. V., and O’Toole, M. J. 1998. Integrated overview of the oceanography and environmental variability of the Benguela Current region. Second Regional Workshop on the Benguela Current Large Marine Ecosystem (BCLME). BCLME Thematic Report, 2. 58 pp.

Sinclair, M., Arnason, R., Csirke, J., Karnicki, Z., Sigurjonsson, J., Skjoldal, H. R., and Valdimarsson, G. 2002. Responsible fisheries in the marine ecosystem. Fisheries Research, 58: 255–265.

Sinclair, M., and Valdimarsson, G. (Eds). 2002. Responsible Fisheries in the Marine Ecosystem. CABI Publishing, Oxford. 448 pp.

Smeets, E., and Weterings, R. 1999. Environmental indicators: typology and overview. European Environment Agency Technical Report, 25. Copenhagen. 19 pp.

Ulanowicz, R. E., and Puccia, C. J. 1990. Mixed trophic impacts in ecosystems. Coenoses, 5: 7–16.