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Article *in* Oceanography and Marine Biology · January 2004

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IMPACTS OF HUMAN ACTIVITIES ON MARINE ANIMAL LIFE IN THE BENGUELA: A HISTORICAL OVERVIEW

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Abstract This review provides a historical overview of human activities in the Benguela and documents their effects on marine animal life. Considered are the activities of conventional industrial and inshore fisheries but also nonfishery activities, such as mariculture, regulation of river flow, introduction of marine invasive species, marine contruction and mining, pollution and climate change. Human influences may conveniently be divided into four epochs: aboriginal (c. 10,000 before present (BP)–c. 1652), preindustrial (c. 1652–c. 1910), industrial (c. 1910–c. 1975) and postindustrial (c. 1975–present). The aboriginal epoch is characterised by low levels of mainly intertidal exploitation; the preindustrial epoch by intense exploitation of few large, accessible species; the industrial epoch by technological development and a subsequent massive escalation in catches; and the postindustial epoch by improved resource management and stabilisation of catches, but increasing nonfishery impacts on the system. Over 50 million t of biomass has been extracted from the system over the past 200 yr, resulting in significant changes in community structure. Extraction rates peaked at over 1.3 million t yr^{-1} in the 1960s and have subsequently declined by over 50%. Populations of whales, seals and pelagic and demersal fishes are recovering from historical overexploitation, while those of inshore stocks, particularly abalone, rock lobster and inshore linefishes, remain severely depressed.

Introduction

This review is a product of the History of Marine Animal Populations (HMAP) project, the historical component of the Census of Marine Life Programme (http://www.CoML.org), a decade-long multinational project funded largely through the Alfred P. Sloan Foundation and Consortium for Oceanographic Research and Education (CORE). The initial objective of the HMAP project has been to identify a series of large marine ecosystems (or global fisheries), for which good ecological

0-8493-2727-X/04/$0.00+$1.50
Oceanography and Marine Biology: An Annual Review 2004, **42**, 303–392

and historical catch data exist, and to document the effects of human activities on the structure and functioning of these systems. The Southwest African Shelf, termed the Benguela here, is one of seven such case studies being investigated.

For the purposes of this review the Benguela region (Figure 1) is defined as extending from Cape Agulhas in the south to the Namibian–Angolan border (17˚S) in the north, a distance of some 2500 km. These boundaries also mark the approximate biogeographical transition zones between the cool–temperate biota of the Benguela and those of the warm–temperate South Coast Province of South Africa to the east and the more subtropical Angolan region to the north (Emanuel et al. 1992, Branch & Griffiths 1988). The northern part of this coastline (N of about 32˚S) is extremely arid and virtually linear, the only significant embayments being at Luderitz, Sandwich Harbour and Walvis Bay, and the only river of note the Gariep (Orange), which forms the border between South Africa and Namibia. In the South the coastline becomes more irregular, with several prominent capes (Cape Columbine, Cape Peninsula, Cape Hangklip) and larger bays (St. Helena Bay, Saldanha Bay/Langebaan Lagoon, Table Bay, False Bay, Walker Bay). The seaward boundary of the region is considered to be that of the exclusive economic zones (EEZs) of South Africa and Namibia.

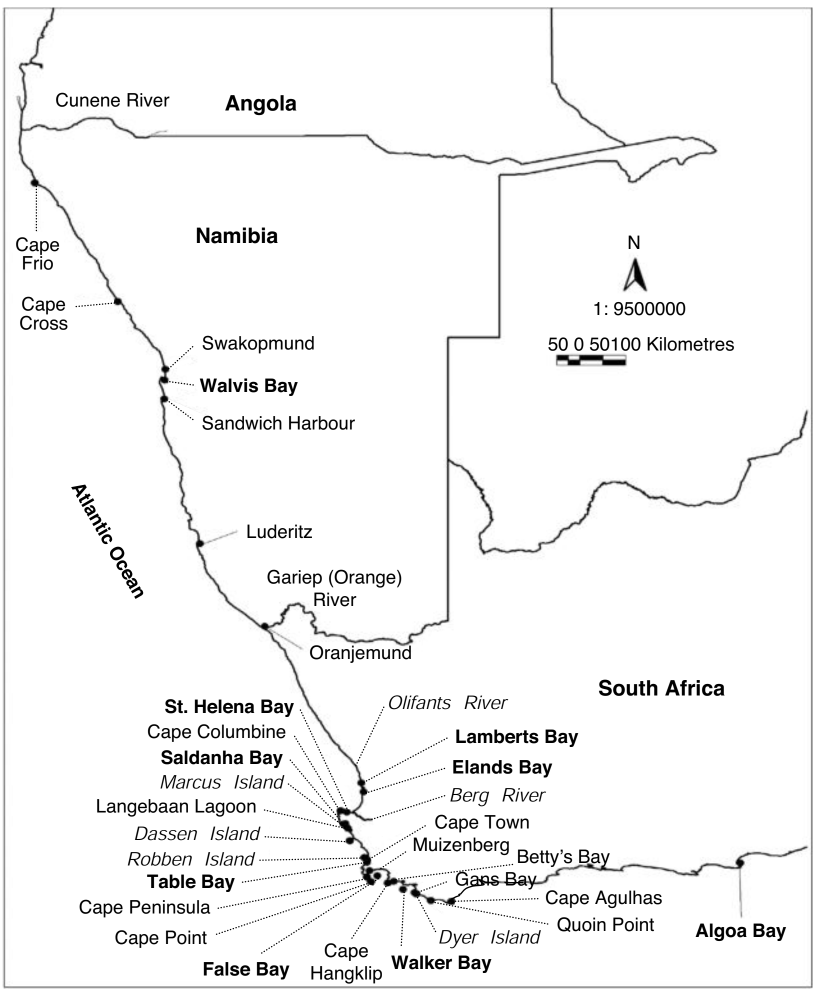

Figure 1 Map of the Benguela region, showing place-names mentioned in the text.

Evidence for a human presence on the shores of the Benguela dates from the Early Stone Age (1–0.5 million yr before present (BP)), but systematic exploitation of marine resources appears only to have commenced during the last interglacial period (120,000 yr BP) (Parkington 2001a; see also below). Marine resources soon became integrated into a hunter-gathering economy. Indeed, the fatty acids contained in the marine food chain are thought to have been important in human evolutionary development (Crawford et al. 1999, Parkington 2001a,b, Broadhurst et al. 2002). Low population levels and rudimentary technology essentially limited the impacts of hunter-gatherers to the intertidal. The establishment of pastoralism (1900–1400 yr BP) ultimately altered human use of marine resources, curtailing human access to the coast to occasional visits determined by the annual movements of their livestock (Smith 1992).

The first European seafarers entered the Benguela in the late 15th century, en route to Asia, but in the mid-17th century the Dutch East India Company (DEIC) established a permanent settlement at Table Bay. The new colony expanded steadily up the west coast to the Berg (c. 1700), Oliphants (1750), and Buffels (1798) Rivers. The DEIC also annexed five bays north of the Orange River (the current Namibia), including Walvis Bay and Luderitz, in 1793. The DEIC's discouragement of private enterprise and the low rainfall in the region limited settlement, and hence the impact of European colonisation on marine resources (Van Duin & Ross 1987) during this period. When the British supplanted the DEIC in 1806 they extended their jurisdiction to the Orange River in 1847 and subsequently annexed all the Namibian islands (1866) and Walvis Bay (1879), while allowing Germany to seize the mainland between the Orange and Cunene Rivers as its colony in 1884. The British allocated land for agriculture and mining and leased out use rights to seabirds, seals and fishes to facilitate settlement and trade on the west coast. This was further encouraged by railway construction and the introduction of steam shipping to the coastal trade in the final quarter of the 19th century (Van Sittert 1992).

British rule ended in 1910, with the amalgamation of its colonies and the Boer republics into the Union of South Africa. South Africa brought the Benguela under a single political administration for the first time in 1915, when it conquered German Southwest Africa during the First World War. Southwest Africa was subsequently administered as a South African colony until its independence as the Republic of Namibia in 1990, and the return of Walvis Bay, a South African enclave, to Namibian administration in 1994. During the 20th century the human population along the Northern Benguela coastline remained small and restricted largely to a series of factory-cum-holiday towns. The economy in most of this region remains based largely on natural resources, including alluvial diamonds, rock lobster, pelagic fishes, and beachfront property. The only major population centre – Cape Town and its satellite settlements – lies in the extreme south. This region has a vibrant and diverse economy quite different from that of the more arid west coast and incorporates substantial industrial, commercial, agricultural and tourism components.

A number of previous reviews have synthesised existing information on various marine components of the Benguela ecosystem. These include articles on physical features and processes (Shannon 1985), chemical processes (Chapman & Shannon 1985), plankton (Shannon & Pillar 1986), the major fishes and invertebrate resources (Crawford et al. 1987), the coastal zone (Branch & Griffiths 1988) and marine geological aspects (Rogers & Bremner 1991). Some of these reviews remain valuable but those dealing with exploitation of biological resources have dated rapidly, because management policies and the status of many marine living resources have undergone radical transformation in recent years. This review aims to provide an updated historical overview of the status of exploited marine stocks in the Benguela region and to consider other, nonexploitative anthropogenic influences that may affect marine animal life in the region. Thus, for the first time, it is possible to see, in a single source, the interactive effects of all types of human impact on the Benguela. These influences are discussed under separate headings below, beginning with the earliest forms of exploitation and the more conventional fisheries and proceeding to other, more indirect environmental influences.

Precolonial exploitation

South Africa's 3000 km coastline is dotted by many thousands of archaeological sites (shell middens and caves) that bear witness to the long-term exploitation of marine resources (shellfish, crustaceans, fishes, seabirds, and marine mammals). The earliest evidence for marine exploitation by people in southern Africa dates to the Middle Stone Age, around 120,000 yr BP, and is found in fossilised open shell middens along the west coast and cave sequences on the south and east coasts (Volman 1978, Klein 1999, Henshilwood et al. 2001, Marean & Nilssen 2002). Much of what is known about prehistoric exploitation of marine animals in southern Africa, however, derives from the far more numerous Later Stone Age (LSA) sites, dating to the last 12,000 yr. The majority of coastal sites dating before that time became submerged along the coastal shelf as a result of rising sea levels from –120 m since the end of the last Glacial Maximum, around 18,000 yr BP (Van Andel 1989).

A diverse range of observations is available for the LSA sites, namely, which species were exploited, their relative abundance in the archaeological record, the technology used to exploit the species and the seasonality of their exploitation (Avery 1987, Buchanan 1988, Jerardino 1996, 1997, Jerardino & Parkington 1993, Jerardino & Yates 1997, Inskeep 1987, Noli & Avery 1988, Parkington et al. 1988, Poggenpoel 1996, Schweitzer 1979, Smith et al. 1992). Palaeoenvironmental conditions prevalent during coastal visits by hunter-gatherer groups are also derived from archaeological sources, along with those obtained from conventional sources, such as geological profiles and cores (Jerardino 1995, Parkington et al. 2000, Compton 2001).

Given the current need to control patterns of resource exploitation by our technologically advanced society, it is interesting to speculate as to the impact prehistoric groups had on marine resources using their simple technology. Broadly speaking, the precolonial exploitation of marine animals consisted mainly of the collection of at least 15 species of molluscs and crustaceans (Buchanan 1988, Jerardino 1997, Jerardino & Navarro 2002) combined with some hunting, but mostly scavenging. Species scavenged included washed up seabirds of about 10 species, Cape fur seals (*Arctocephalus pusillus pusillus*) and cetaceans (Avery 1987, Smith et al. 1992, Jerardino & Parkington 1993). Fishing of at least 10 species was also practiced, with the aid of simple technology such as gorges and fishhooks made out of bone, wooden spears, reed baskets, and nets, as well as stone traps (Avery 1975, Poggenpoel 1996). Judging from the amount of visible archaeological debris and available lists of identified species, the harvesting pressure exerted on marine resources varied in intensity with both locality and time.

A west coast study

Among the small number of coastal projects that have focused on precolonial settlement and subsistence in South Africa, archaeological investigations in the Lambert's Bay and Elands Bay areas (32–32° 35'S) have yielded the best data for evaluating the impact of prehistoric inhabitants on marine resources. Aside from the high density of observations generated for this particular stretch of coastline, there is a relatively good palaeoenvironmental record for the area, and a good level of understanding of present marine animal communities (Branch & Griffiths 1988; marine mammal sections, p. 308–317). These factors make this area the best candidate for the study of precolonial exploitation of marine animal populations in South Africa.

The vast majority of the excavated sites in this study area consist of deposits dominated by marine shell remains and varying quantities of marine and terrestrial vertebrates. The rate at which these deposits were accumulated has clearly changed since the present shoreline was established about 8000 yr BP (Figure 2). The first signs of relatively fast accumulation of shell midden deposits and intensive shellfish collection date to around 3500 yr BP (Jerardino 1996, Jerardino & Yates 1996). Subsequently, between 3000 and 2000 yr BP, shellfish exploitation was greater than at any other time during the Holocene period. During this millennium, enormous shell middens

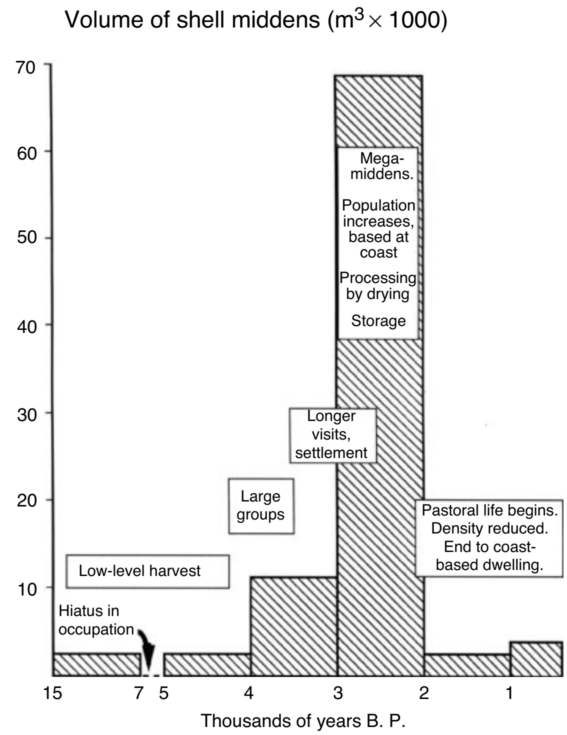

Figure 2 Summary of temporal changes in the volumes of shell middens in the vicinity of Elands Bay and Lambert's Bay, with notes on changes in prehistoric human use of marine resources. (Data from Jerardino & Yates 1996.)

(megamiddens) containing tons of black mussel shells and relatively few bone and cultural remains accumulated immediately behind rocky platforms (Jerardino & Yates 1997) (Figure 2). The overall dietary mix of people (hunter-gatherers), as reconstructed from isotopic measurements on skeletons buried along stretches of the west coast and from archaeological food waste, was more marine between 3000 and 2000 yr BP than either before or after (Lee-Thorp et al. 1989, Jerardino 1996). The scale of shellfish exploitation was dramatically reduced after 2000 yr BP, a period coincident with the arrival of pastoralism to the west coast of South Africa. During the last 2000 yr, the precolonial diet was derived predominantly from terrestrial resources.

Studies focusing specifically on prehistoric shellfish exploitation have shown that human impact on rocky shore molluscs seems likely to have fluctuated in response to a succession of different settlement patterns (from more mobile to more sedentary), demography and palaeoenvironmental conditions at the time of resource exploitation (Jerardino 1997, Klein 1999). Although comparative contemporary data for exploitation levels at these same sites are not available, it is clear that these prehistoric levels of exploitation were very low and almost always sustainable. This is in marked contrast to the extremely high and unsustainable levels of subsistence exploitation currently occurring on the east coast of South Africa (Griffiths & Branch 1997).

The marine bird and mammal records are still insufficiently studied to derive meaningful conclusions as to the impact of prehistoric people on these species. The same applies to the archaeological record of Cape rock lobster (*Jasus lalandii*). The study of this species is particularly important, as it is well known to influence directly and indirectly the abundance and population structure of its prey and other interacting species (Castilla et al. 1994; see also rock lobster section, p. 340–345). Groundwork for the study of the exploitation of Cape rock lobster in the study area was recently laid out (Jerardino et al. 2001, Jerardino & Navarro 2002) and preliminary observations point to an intricate combination of variables such as sea level change, possibly resulting in shrinking

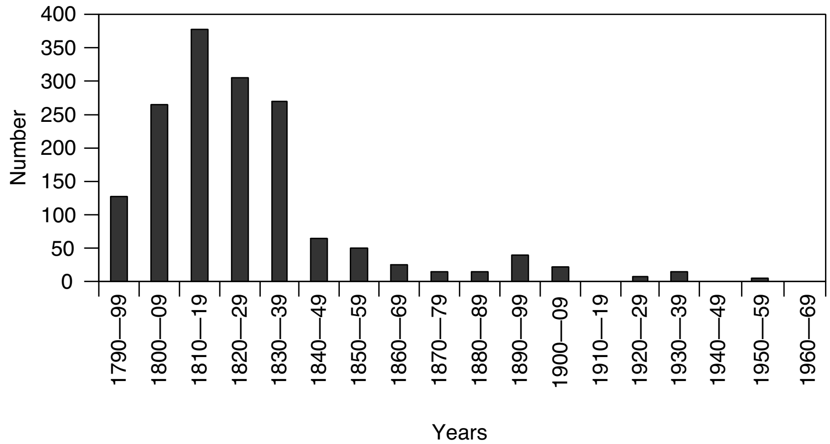

Figure 3 Catches of southern right whales off South Africa by decade, 1790–1940. (Redrawn after Best et al. 1997.)

availability of suitable hideouts for lobsters, and greater exploitation pressures as a result of increasing hunter-gatherer populations together causing fluctuations in the mean sizes of rock lobsters.

Clearly, much remains to be done to improve our understanding of ecosystem change and the role of people in changing marine ecosystems in the precolonial past. In particular, work needs to be done to build the necessary databases with observations already obtained; to generate more detailed observations of sea level change, shoreline configuration, marine productivity and sea surface temperatures; and to carry out additional fieldwork at sites presenting longer and well-resolved sequences. These topics are the subject of ongoing research effort.

Cetaceans

This account is confined to those cetacean species that mainly occur over the continental shelf in the Benguela region, which in the case of large whales restricts the coverage to southern right whales (*Eubalaena australis*), humpback whales (*Megaptera novaeangliae*), and the inshore stock of Bryde's whales (*Balaenoptera edeni*). Despite this geographical restriction, it must be appreciated that at least the first two species are highly migratory, and so subject to human impact quite removed from the Benguela region. Whaling in the Southern Ocean is perhaps the prime example, but because of continuing uncertainty about linkages between breeding and feeding areas, it is difficult to assign pelagic catches to a stock inhabiting a particular coastal region. As a consequence, this review mainly concerns impacts (including catches) that occurred directly within the Benguela.

Prior to the arrival of European settlers, recorded exploitation of cetaceans in the region is confined to reports of the utilisation of stranded whales and dolphins for food and other materials. A few coastal dolphins (probably mostly bottlenose dolphins, *Tursiops truncatus*) were killed by native peoples wading out from the shore (Budack 1977) but it is unlikely that any of these activities adversely impacted the populations.

European colonisation at the Cape in 1652 resulted in an immediate interest in the commercial exploitation of large whales that abounded in the neighbouring bays. The earliest attempts at their capture, however, were desultory and largely ineffective. This changed dramatically with the advent of visiting pelagic whaleships, largely from the U.S., but also from France and Britain, in the late 18th century. Although there was a brief episode of whaling by the Dutch West Indian Company in the Walvis Bay region early in the 18th century, the number of voyages involved was small and unlikely to have significantly affected abundance. The main onslaught began around 1780, when whaleships from New England began to overwinter at the Cape. Their principal quarry was the southern right whale, which yielded large quantities of oil and whalebone. Despite relatively primitive equipment (open boats and hand harpoons), the size of the fleet (up to 30 whaleships in one bay at a time) and the predictability of right whale behaviour led to a rapid decline in whale abundance. By the 1840s the pursuit was largely abandoned by visiting whaleships. Estimates of the landed catch in the South Atlantic by U.S. whaleships from 1805–1909 range from 28,500–32,200 individuals, with the bulk of the catch (24,500–27,000) being taken prior to 1840 (Best 1987). Richards & du Pasquier (1989) have made an independent estimate of the number of right whales taken by all fleets (not just the U.S.) on the coast of southern Africa (including Mozambique) between 1785 and 1812 as 12,000. From their Table 1 it can be estimated that about 34% of these were taken by vessels travelling to Delagoa (Maputo) Bay or the east coast of Africa, and the rest by vessels visiting Walvis Bay or the Cape of Good Hope.

Meanwhile, shore-based whaling for right whales finally began on an organised basis at the Cape in 1792. Despite whaling stations springing up at a number of locations in the Western and Eastern Cape, their catches never reached the levels of those of the visiting whalers. Catches peaked at between 200 and 400 whales decade^{-1} in the early 1880s (Figure 3), whereas the total catch landed from 1792–1912 is estimated at only 1580 whales (Best & Ross 1986). Unlike their foreign counterparts, however, the colonial whalers were able to continue catching because their costs were much lower and the activity could be pursued in association with other fishing enterprises, such as beach-seining. In this way, the catch of only one or two whales a season could still be highly profitable, with almost all the products being exported.

Other species, notably humpback, bottlenose, sperm, blue, finback (probably Bryde's), pygmy right, and killer whales, were taken in this shore-based open-boat fishery (Best & Ross 1986), but the numbers recorded were too low to have been of population significance. As right whales declined in abundance, the pelagic whalers of other nations began to capture other species on the southern African coast, notably humpback whales. Although there are no published estimates of the number of humpback whales taken by the fishery in this region, U.S. catches worldwide between 1815 and 1905 have been estimated at 14,000–18,000 animals, with the peak catch (11,000–15,000) being taken between 1855 and 1889 (Best 1987). Plots of catch positions given by Townsend (1935) indicate a substantial concentration between Gabon and central Angola from June to September. Many of the humpback whales taken in this fishery may therefore have been from the population that is believed to migrate through the Benguela region to its breeding grounds off equatorial West Africa in winter. (A rough estimate from Townsend's chart is that between a third and half of all catches were from West Africa, suggesting total kills of between 4500 and 9000 animals.) This fishery was also characterised by a much higher struck-and-lost rate than for southern right whales, and as several of these animals would have been dead (sunk) or died later, the landed catch is probably an underestimate of the total removals from the population.

No assessment of the effect of these catches on the population has been undertaken, but within 20 yr of the end of the peak catch, humpback whales were again the target of a fishery but this time one potentially far more destructive. In 1909 modern whaling began on the west coast of southern Africa. Instead of open boats powered by sail or oars, with hand harpoons as the principal weapon, whales were pursued by steel-hulled steam-driven catchers of 100 t or more, with the harpoon fired from a mounted cannon and carrying an explosive grenade at its tip. Methods of processing the whale were initially not very different from those of the earlier fishery, with utilisation being largely confined to the blubber and tongue and the rest of the carcass being jettisoned. The escalation in catching effort was enormous, so that by 1913 at least 16 land stations or moored factory ships were whaling between Cap Lopez in Gabon and Hangklip in South Africa. Catches soared accordingly, from about 600 in 1909 to nearly 6000 whales in 1913 (Best 1994). Such a whaling intensity was clearly nonsustainable, and by 1915 catches had crashed to less than 200 (Figure 4). Thereafter the industry largely switched to other species (blue, fin, and sei whales especially), and by 1963 (when humpback whales were finally given protection by the International

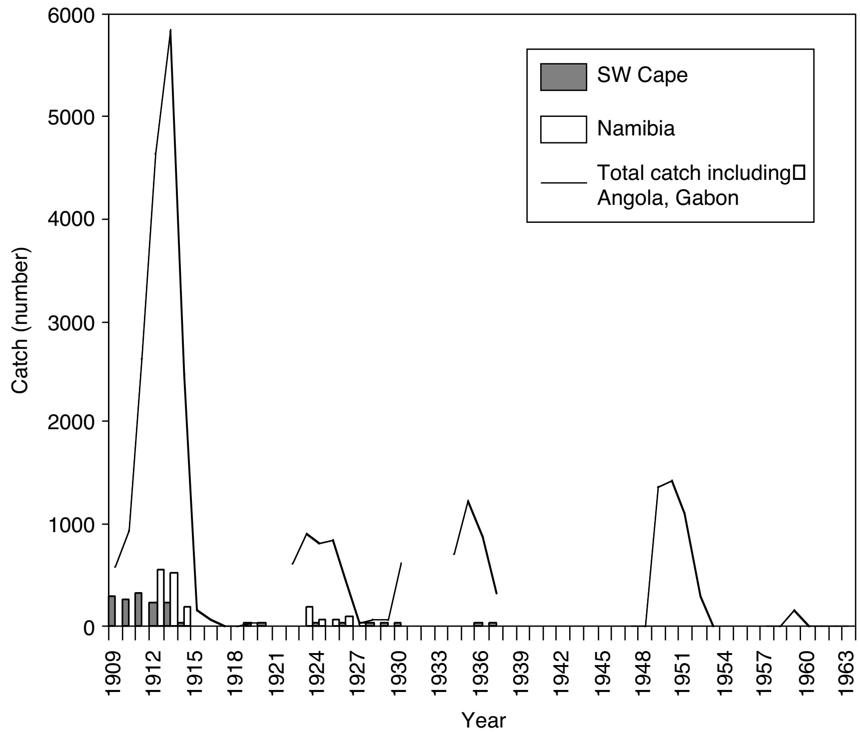

Figure 4 Annual catch of humpback whales off the west coast of South Africa.

Whaling Commission (IWC)) only a handful were being taken annually by the sole surviving land station at Donkergat in Saldanha Bay (Figure 1). Curiously, episodic whaling off Gabon (1934–37, 1949–52) was reasonably successful, suggesting that the humpback whales passing Saldanha Bay may represent a different component of the population, possibly one feeding to the east of the continent, off Queen Maud Land.

Modern whaling also affected right whales. Despite their rarity, right whales were valued by the industry as highly as sperm (and considerably more than blue, humpback, or sei) whales, and this must have encouraged their continued exploitation. It is an indication of just how scarce right whales must have been at the beginning of the 20th century, that only 100 were taken in modern whaling on the South African coast between 1908 and 1937 (Best & Ross 1986, Figure 3). Some projections have indicated that at its lowest point (about 1937), the South African right whale population might have contained as few as 30–68 mature females (Tormosov et al. 1998). Since 1935 the species has been internationally protected but this has not prevented some illegal catching, particularly by pelagic fleets from the Soviet Union, which took at least 3368 southern right whales between 1951–52 and 1970–71 (Tormosov et al. 1998). Such poaching ceased with the introduction of the International Observer Scheme and since 1971 the South African population of right whales has been increasing steadily at 7% a year (Best et al. 2001). In 1997 the population stood at an estimated 659 adult females, equivalent to a total population of some 3100 animals (IWC 2001). This compares to an estimated original population size for southern Africa (both east and west coasts) of 20,000 right whales (Richards & du Pasquier 1989). The latter estimate, however, is difficult to interpret. It includes whales from three widely separated grounds (Walvis Bay, Cape of Good Hope and Delagoa Bay), whose relationship to each other is still unknown, and for which only the Cape of Good Hope can be considered as equivalent to the current South African population. Their estimate also ignores the effect of recruitment during exploitation (resulting in an overestimation of original population size) and only includes catches from coastal waters. Substantial catches of right whales also occurred between Cape Town and Tristan da Cunha in the mid-Atlantic (Townsend 1935). It is now known that these catches are highly likely to have included numbers of right whales that also visit the coast of South Africa (Best et al. 1993; unpublished satellite tagging data). Model projections have shown that overall, the southern right whale population is about 10–14% of its original abundance (Figure 5).

The current status of humpback whales on the west coast of southern Africa is unknown. Incidental sightings (and a preliminary shore-based survey at Cape Columbine in 1993; Best et al. 1995) would suggest that some increase must have occurred since protection, given the size of the catch in the last few years of exploitation and the number of incidental sightings currently being made. However, at present there are no estimates of population size or trend.

The third species of large whale occurring over the continental shelf, the Bryde's whale, was only "discovered" when modern whaling started on the west coast of South Africa. The first published description of its external appearance was based on animals examined at the Donkergat whaling station in Saldanha Bay. Hence one can only speculate that the species was among those rarely taken by open-boat whalers and declared as "finbacks." Unfortunately, publication of the external description was not enough to ensure that the species was always correctly identified in catches thereafter. Confusion with sei or fin whales persisted until well into the 1960s (Best 1994), so it is difficult to reconstruct a reliable catch series for the species. To add further complication, two separate populations of Bryde's whales have been described from the west coast of South Africa, one inshore over the continental shelf (largely nonmigratory) and one offshore, which appears to migrate between equatorial regions in winter and waters off southern Namibia in summer (Best 2001). Both populations feature in the catches, so although there are morphological differences between the two, unless the whales are examined by trained personnel, it is impossible to separate them. In January–February 1983 a shipboard survey was undertaken of the continental shelf of

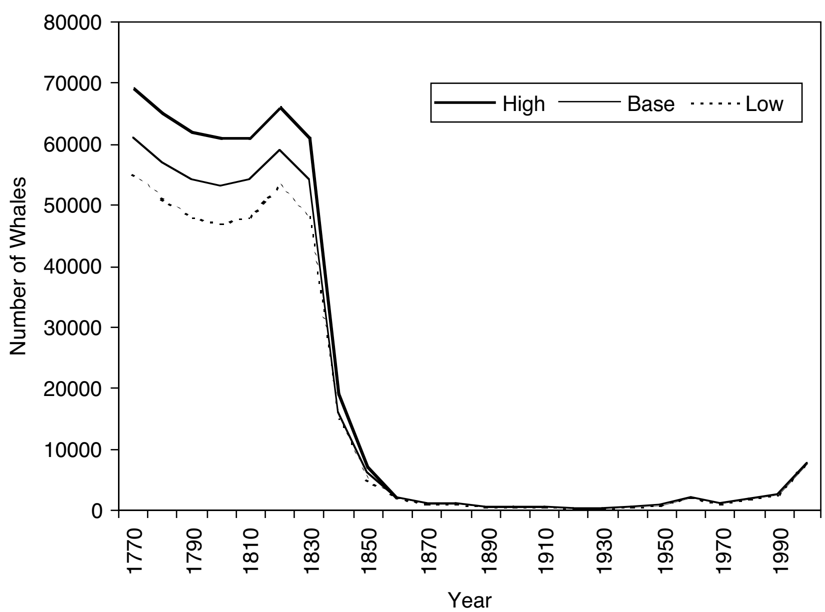

Figure 5 Model projections of the total population size of southern right whales, 1770–1997, using high, low, and base estimates of historic catch scenarios and a 1997 population size of 7571 whales. (From IWC 2001.)

South Africa between East London and St. Helena Bay, corresponding to the known range of the inshore stock of Bryde's whales. This resulted in a line transect estimate of 582 ± 184 whales. This is likely to have been an underestimate, as the survey was carried out in closing mode (Best et al. 1984). Although the status of this stock is unknown, because of its relatively restricted range it is unlikely that it was ever very large.

Cape fur seals

The Cape fur seal *Arctocephalus pusillus pusillus* is the only indigenous pinniped inhabiting the shores of southern Africa. It breeds at 25 colonies, 15 of which are in Namibia and 10 in South Africa (Figure 6). Of these, seven are on the mainland and 18 on islands. There are an additional nine sites where seals haul out, but little or no breeding has been recorded. Cape fur seals preferentially choose nearshore rocky islands on which to breed, which are cooler than the mainland and afford them protection from land predators. However, overcrowding on the small, coastal islands has caused them to overflow onto the nearby mainland and establish new colonies there.

The colonies are distributed around 3000 km of coastline from Algoa Bay in southeast South Africa to Cape Frio in northern Namibia (Figure 6). Although there is no evidence that they breed there, seals have been recorded in Angolan waters up to about 650 km north of the Cunene River. About 90% of the population is found on the west coast, taking advantage of the rich fisheries of the Benguela ecosystem, whereas only about 10% occurs on the south coast, where food resources are less abundant (Rand 1959, Shaughnessy 1979, 1982, David 1987, 1989).

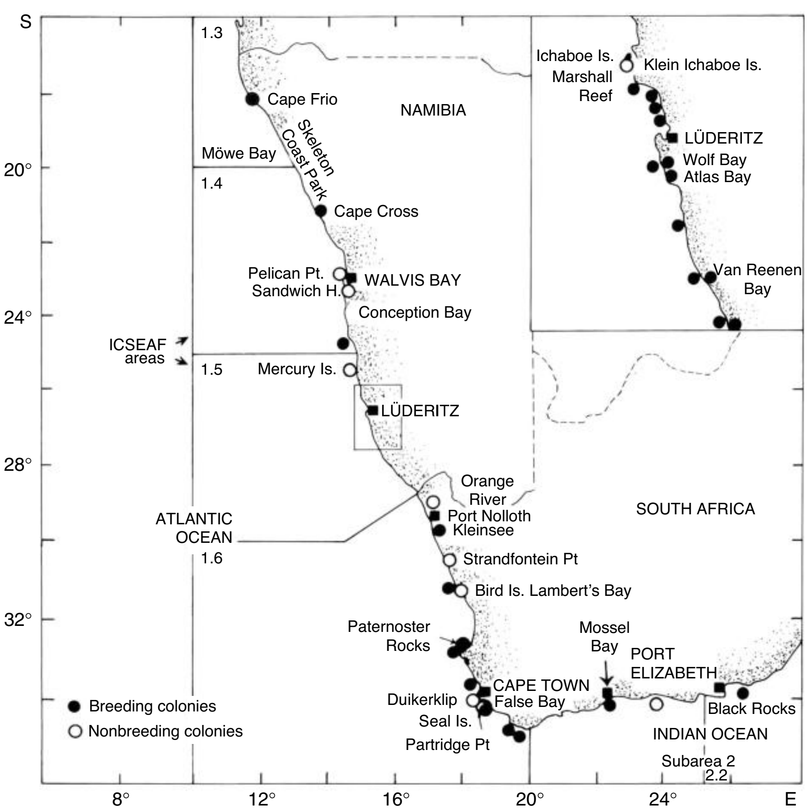

Figure 6 Map showing breeding and nonbreeding colonies of Cape fur seals around the coast of South Africa and Namibia.

Exploitation of seals is one of the oldest commercial "fisheries" in southern Africa. Sealers aboard the old sailing vessels plundered as many colonies as they could find, without any form of control, though there are few early records of numbers taken (Rand 1972, Best 1973). The first known sealers were Dutch and they killed about 45,000 seals near the Cape of Good Hope in 1610. As a consequence, early Dutch sealing destroyed most of the colonies close to Cape Town. Before the arrival of Dutch settlers at the Cape in 1652, French sealers were sometimes active on the islands in and around Saldanha Bay. After the colonisation of the Cape, ships of the English East India Company based in Table Bay, while waiting to resupply the homeward fleet, may have raided local seal islands. There was little further sealing until the late 18th and early 19th centuries, when British and American sealers were active on the west coast of southern Africa (Shaughnessy 1984, David 1989).

In those days there were no legal controls and sealing was completely indiscriminate. Rookeries were invaded even during the breeding season and all age classes were taken, including pregnant cows and small black pups only a few weeks old. As a result, the seal population was reduced to low levels by the end of the 19th century, by which time at least 23 colonies had become extinct in South Africa and Namibia (Rand 1972, Shaughnessy 1984). Subsequently, several of these islands were colonised by seabirds and permanent accommodation was built on some of them for the protection of the guano harvest. This effectively prevented recolonisation by seals, which are sensitive to human presence.

Legislation and harvesting

The desire by the government to improve the control over seal harvesting and to end private sealing prompted it to promulgate the first legal protection under the Cape Fish Protection Act of 1893, which stipulated that no seals might be harvested without a government permit. In 1909 the sealing season was limited to prevent disturbance during the breeding season and was further amended and curtailed in 1936. The union government controlled sealing in Namibia by means of the Sealing and Fisheries Proclamation of 1922 and the Sealing and Fisheries Ordinance of 1949. Both acts prohibited sealing without a licence. Sealing is currently managed under the Sea Birds and Seals Protection Act of 1973, which prohibits landing on any island and the capture or killing of any seal or seabird without a permit. Under this Act the minister is empowered to prescribe the age, size and sex of seals killed, as well as the season and localities where sealing may take place (Shaughnessy 1984).

Despite the low seal population at the beginning of the century, harvesting continued at certain colonies almost every year from 1900 (Wickens et al. 1991). The harvest of pups was small initially but grew progressively as seal numbers increased. From 1900–10 the total annual harvest was 2600–9300 pups. During the economic depression of the 1930s it was between zero and 16,800 pups and by the 1950s it had increased to 27,200–45,000 pups. By the 1970s the harvest had grown to 62,400–81,200 pups and the industry reached its zenith in the 10 yr preceding 1983, when the average harvest was about 75,000 pups (Figure 7).

Until 1965 the bulk of the harvesting was carried out by government sealers of the Guano Islands Division of the Department of Commerce and Industries. From that date the government began handing over concessions for individual colonies to private enterprise by inviting public tenders for the sole right to seal at each colony. By 1979 all concessions were in private hands and government sealing had ceased (Shaughnessy 1984).

No total allowable catch (TAC) was in operation until 1974, because there was inadequate knowledge of the size of each colony. Sealers, therefore, took as many pups as they could, given the weather conditions and the time available. However, this gap in knowledge was filled with the commencement of regular seal research in 1971 and all concessions awarded after 1974 included a TAC.

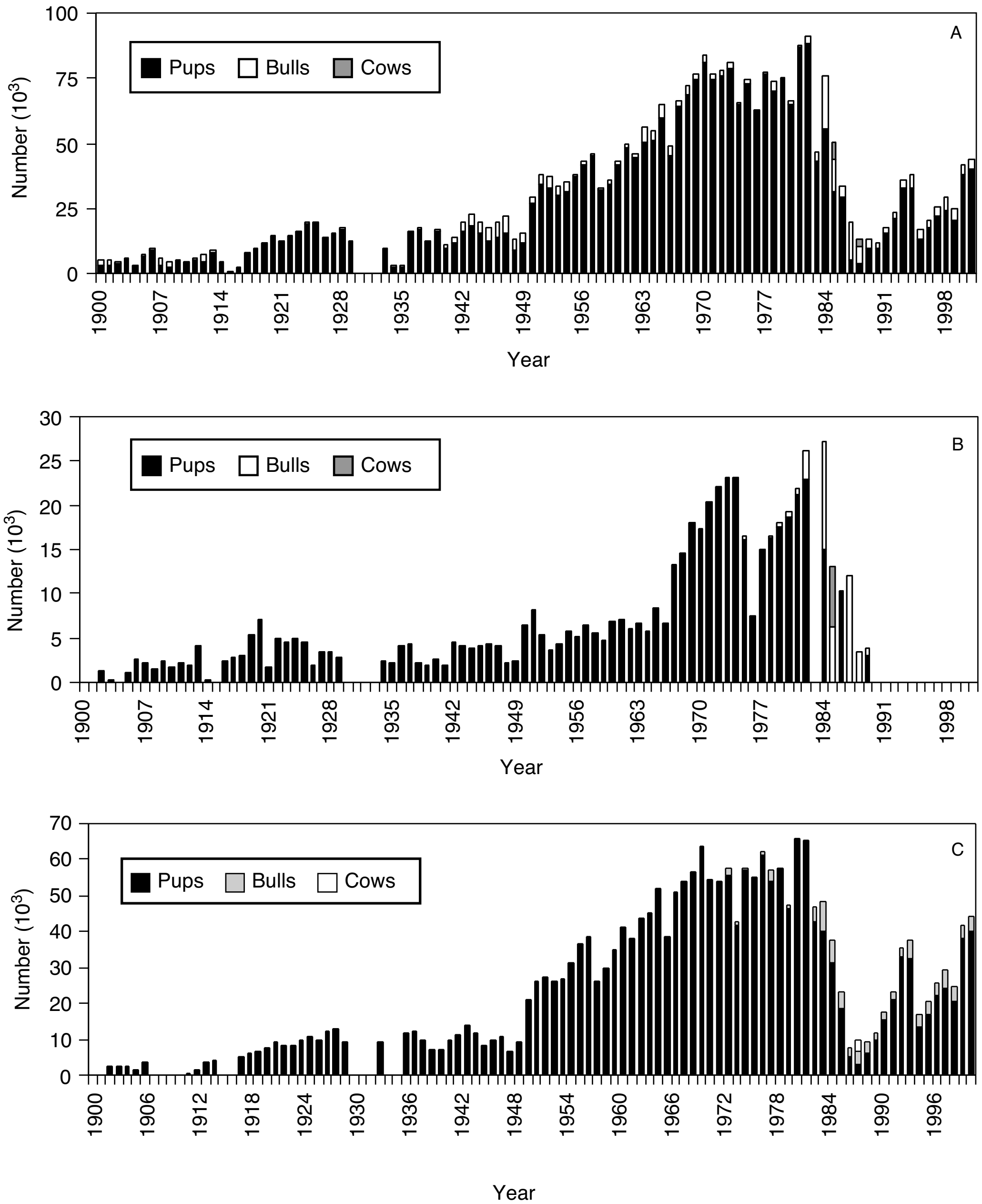

Figure 7 Harvests of pups, bulls, and cows from 1900–2001 at all colonies of the Cape fur seal combined (A) and at South African (B) and Namibian (C) colonies separately.

In addition to the harvest of pups, some adult bulls were taken during the summer breeding season in November–December, when the bulls congregate on the colonies. Prior to 1983 the bull harvest usually totalled 1000–3000 animals but over 5000 were killed during some years in the 1940s and 1960s (Figure 7). However, the bull harvest increased substantially after 1983 and a record number of over 20,000 bulls was harvested in 1984 (see below).

The total harvest of pups and bulls from 1900–2001 was over 3 million (approximate total mass of 100,000 t), made up of about 2.8 million pups (approximate mass of 63,560 t at a mean mass of 22.7 kg pup^{-1}; David 1987) and 244,000 bulls (approximate mass of 36,600 t at a mean mass of 150 kg $bull^{-1}$). The largest number of colonies harvested in a single year was 13 in 1975. The sealing industry thrived until 1983, when the market collapsed as a result of political developments in North America and Europe. In that year militant conservation organisations put pressure on the Canadian government to stop the Canadian harvest of harp seals. The lobby spread to Europe and resulted in the European Parliament requesting member nations to place a voluntary embargo on the import of all seal products. This move effectively caused the collapse of the South African industry because the bulk of the skins were sold in Europe. Sealing in South Africa ceased altogether in 1990, following a ban imposed by the minister of environmental affairs, but continues in Namibia at a reduced level at the Cape Cross and Luderitz colonies (David 1989).

The most valuable product traditionally was always the skins of the pups 7–10 months old, which formed the bulk of the harvest. However, there are some other by-products produced by the industry, the most important of which is seal oil obtained by rendering down blubber scraped from the skin and carcass. This is done in large drums or pots on site. It is also possible to render the stripped carcasses into meat meal and bone meal, provided there is a factory on site, but the value of these products is not high and barely covers production costs. Seal meat has also been processed into pet food but marketing the product was not successful (David 1989).

Another product of considerable value is the genitalia of the bulls, which are dried and sold in the Far East as a supposed aphrodisiac. After 1983 the concessionaires attempted to compensate economically for the loss of the skin market by harvesting more bulls. This form of utilisation is very wasteful because at some colonies no other part of the bull was used and not even blubber oil is produced.

The method of killing pups in the harvest is controversial and has been the subject of sporadic conflict with conservationists and animal rights groups for many years. It is basically the stun-and-stick method used in abattoirs, which is primitive and unaesthetic. The thin skull of the pup is shattered by a blow from a heavy club, which has much the same effect as a bullet. The chest is then immediately opened with a large knife and all the major blood vessels round the heart severed, so that the carcass is rapidly exsanguinated. Although the method is not pretty, it is effective and efficient and no better method has been found. Bull seals are always shot in the side of the head but to shoot thousands of crowded, jostling pups would not be practical or safe (David 1989).

Disturbance programmes

Another impact of human activities on the seal population has been through deliberate disturbance, to cause seals to vacate a particular location. For example, when the permanent human occupation of Mercury Island ceased in the 1980s, seals began to recolonise the island and quickly built up their population to a level where it became established as a new breeding colony. The seals spread over a large part of the island and had a very negative impact on breeding gannets (*Morus capensis*) and African penguins (*Spheniscus demersus*), when they began to overrun the bird colonies. This was considered to be unacceptable because both seabird species are Red Data Book species. A decision was therefore taken to chase the seals and encourage them to move to the adjacent mainland. The island was then reinhabited and the seals were systematically disturbed during three successive breeding seasons. The desired result was eventually obtained when the seals deserted the island completely and moved to the nearby mainland (Crawford et al. 1989).

The population size of seals

No accurate figures of population size were available until census techniques were developed in 1971. The total population in 1997 was estimated to number some 1.5–2 million animals, of which about 60% were in Namibia (Figure 8). It grew at an average rate of about 2.8% yr^{-1} from 1971–92. It seems clear that the great increase in numbers during the 20th century was the normal response of a species recovering from overexploitation. However, the growth in population was not uniform. For example, there was the sharp decline in 1986 due to the decision by the minister to permit extensive bull sealing to extend into December, which caused major disturbance of the colonies at the height of the breeding season. This resulted in significant mortality of pups and disruption of mating. Another major decline in 1995 was caused by the death by starvation of tens of thousands of pups and adults in the Luderitz area, which in turn was due to a warm water oceanographic event that caused the death or emigration of all the local fish populations (Roux 1998). Thus the seal population was unable to find food and the lactating females abandoned their pups, which died of starvation. Many thousands of adults also died.

Only very young black pups (3–6 wk old) are censused from the air. This is because, up to that age, they cannot swim and must therefore stay on land. Furthermore, because of their jet-black natal coat and small size they can be distinguished clearly from adults. Two techniques are used to estimate numbers of pups: aerial photography and tag–recapture. The photography is timed to coincide with the peak of the pupping season, when maximum numbers are expected to be present, and takes place in mid-December. The black pups are counted on large monochrome prints of high contrast.

Whereas all colonies can be photographed within a few days, it is not possible to cover all colonies with a tagging programme. Tagging is labour-intensive and takes much longer to accomplish, so normally only one colony can be tagged per year. The tagging programme is carried out

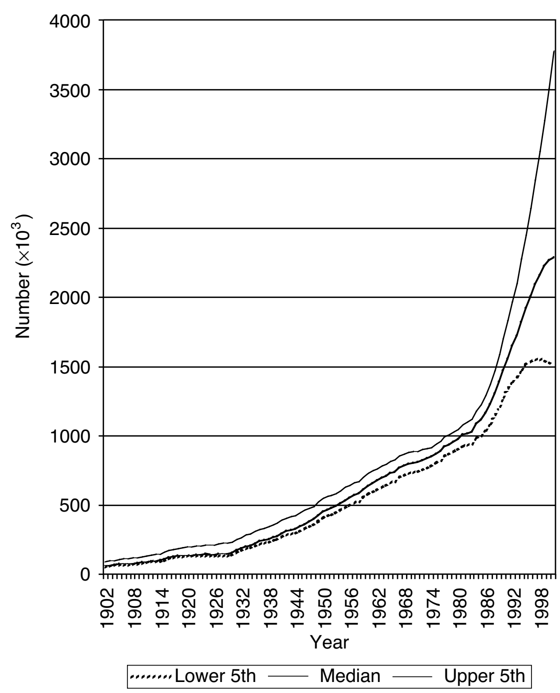

Figure 8 Estimated trend (with 95% confidence limits) in the population size of Cape fur seals, 1900–2001. (After Best et al. 1997).

in mid-January, when the breeding season is over and all bulls have left the colony. The pups are then aged about 6–8 wk and are sufficiently robust to be handled. Tagging is used as a backup to aerial photography and also to study seal movements.

The question naturally arises as to how long the population will continue to expand. This is difficult to answer because the population size before exploitation is unknown. The large size of four of the more recently established mainland colonies (Kleinsee, Cape Cross, Wolf Bay, and Atlas Bay) may more than compensate for the 23 extinct colonies but this is uncertain. In pre-exploitation times it may have been that breeding space was limiting, due to the relatively small size of most of the islands. Now, however, the formation of large mainland colonies, which can spread, has altered this situation. Therefore, human activities, in the form of removal of large predators from, and restriction of human access to, mainland areas, have probably assisted in the recovery of the seal population. The fact that seals have continued to increase in recent years, in parallel with a burgeoning fishing industry, is an interesting observation and may well be indicative of a sufficiency of food to support both the seals and the fishing industry.

Seabirds

There is a rich diversity of seabirds in the Benguela system, including 15 species that breed in the region (8 endemic) and about 60 species that visit it (Ryan & Rose 1985).

Humans have affected seabirds in the region in several ways, including through exploitation of some birds and their products, habitat modification and by-catch mortality. Seabird products that have been harvested include eggs, feathers and guano. Nesting habitat has been greatly modified by human activities, notably guano collection and the provision of alternative nesting sites, such as guano platforms constructed along the northern coast of Namibia. Other anthropogenic changes that have affected seabirds include altered supply of food and pollution, especially oil spills. Seabirds are an incidental by-catch in several fisheries, most importantly longline fisheries. More recently tourism to seabird colonies has developed to the extent where it has the potential to influence populations. Additionally, changes to the structure and functioning of the Benguela system, in particular variations in the abundance of seals, as a result of man's activities (see above), have influenced seabird populations.

The first men from Western civilisations to see African penguins were Bartholomew Diaz and his crew in 1487. Diaz described them as "birds as large as ducks, they do not fly because they do not have feathers on their wings. We killed as many of them as we desired and they bray like asses" (Shelton et al. 1984). This heralded an era in which islands around southern Africa were regularly plundered to provision ships with the products of seabirds and other marine life. The settlement of the Cape in 1652 saw an increase in the frequency of visits to islands for this purpose. Although penguins were killed for food, for fuel to supply ships' boilers and to be rendered down for their fat, the primary attraction was their eggs (Randall 1995).

By the late 1700s exploitation and disturbance by humans had led to the cessation of breeding by African penguins *Spheniscus demersus* at Robben Island and the island was only recolonised by penguins in 1983. Breeding by African penguins stopped, and has not recommenced, at 10 other colonies. At three of these sites the likely cause was again exploitation and disturbance by humans; at five colonies penguins were displaced by Cape fur seals *Arctocephalus pusillus pusillus*; while at one colony scarcity of food was the probable reason for cessation of breeding. At the 10th site breeding was temporary (Crawford et al. 1995b).

Unsustainable harvests of penguin eggs were taken at several islands in the 19th and 20th centuries, leading to large decreases in the sizes of some penguin colonies. Almost 600,000 eggs were collected at Dassen Island in 1919, after which harvests decreased steadily (Figure 9). On the assumption that the decreasing egg harvests reflected the trend in the population of adult penguins at the island, it was estimated that 48% of all eggs produced were harvested and that the population of penguins aged 2 yr or older was at least 1.45 million in 1910 (Shannon & Crawford

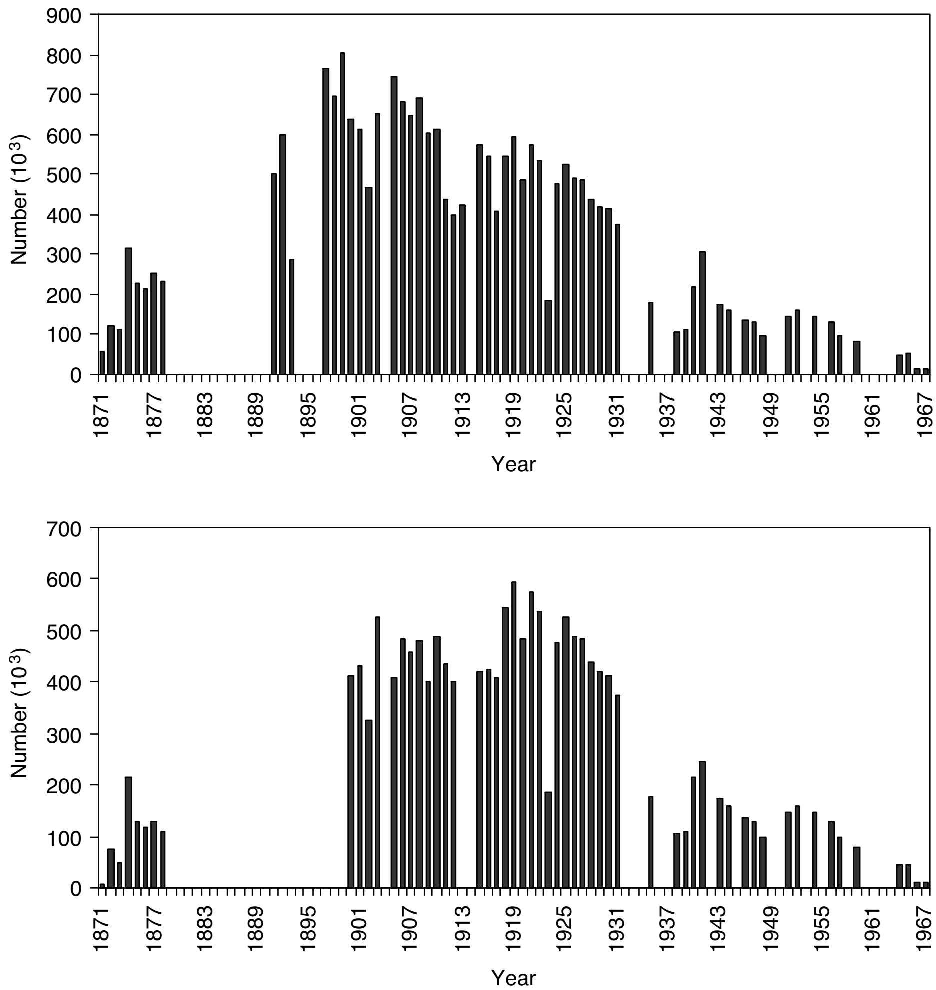

Figure 9 Harvests of the eggs of African penguins (top) off southern Africa and (bottom) at Dassen Island, 1871–1970. (Redrawn from Best et al. 1997.)

1999). By the early 1990s, this had decreased to 30,000 (Crawford et al. 1995c). Sanctioned collections of penguin eggs in southern Africa stopped in 1967 (Shelton et al. 1984). Feathers were collected as a form of down in the early part of the 20th century but this activity probably did not have a great influence on seabird populations.

Collection of guano in southern Africa commenced in March 1843 at Ichaboe Island. Management of this industry is documented by Hutchinson (1950), Shaughnessy (1984) and Best et al. (1997). In 1845, licences were issued to vessels collecting guano. From 1847 at Ichaboe Island and 1869 at certain other islands, concessions were granted for the harvesting of guano. These concessions terminated in the 1890s. From the 1890s until 1975, management of the islands and the collection of guano were primarily undertaken by the South African government. In 1976, the collection of guano was again leased out to private enterprise. Concessions were not renewed in the 1990s, when the harvesting of guano at islands lapsed. In the 1930s, seawalls were built around some islands to protect nesting birds from high seas and to reduce losses of guano into the ocean. For the period 1844–95 it has proved possible to reconstruct the annual harvest of guano from Namibia (but not from South Africa) using records of guano imported to and exported from several countries (Van Sittert & Crawford 2003). Accurate records of guano harvests are available since

1892 for South Africa and 1896 for Namibia (Figure 10). The guano harvest was decreased in some years by heavy, unseasonable rains and in others by the periodic scarcity of fish (Hutchinson 1950). After commercial purse-seine fisheries became established off South Africa and Namibia, the amount of food available to seabirds was reduced and the production of guano decreased (Crawford & Shelton 1978). Commencing in 1931, platforms to attract seabirds to breed and to deposit guano were constructed along the coast of northern Namibia between Walvis Bay and Cape Cross. Guano is still collected annually at these platforms (Figure 10).

The main producers of seabird guano in the Benguela system are Cape gannets *Morus capensis* at six islands and Cape cormorants *Phalacrocorax capensis* at the platforms and most islands. To a lesser extent, guano is also produced by Bank cormorants *Phalacrocorax neglectus*, other cormorants, and African penguins. Guano was collected from April or May, at the conclusion of the breeding seasons of Cape gannets and Cape cormorants, but at Malgas Island in 1977, commencement of collecting at too early a date caused the deaths of 800 chicks after they had been displaced from nests. About the same time unrecorded numbers of chicks at Ichaboe and Possession Islands suffered a similar fate (Crawford et al. 1983).

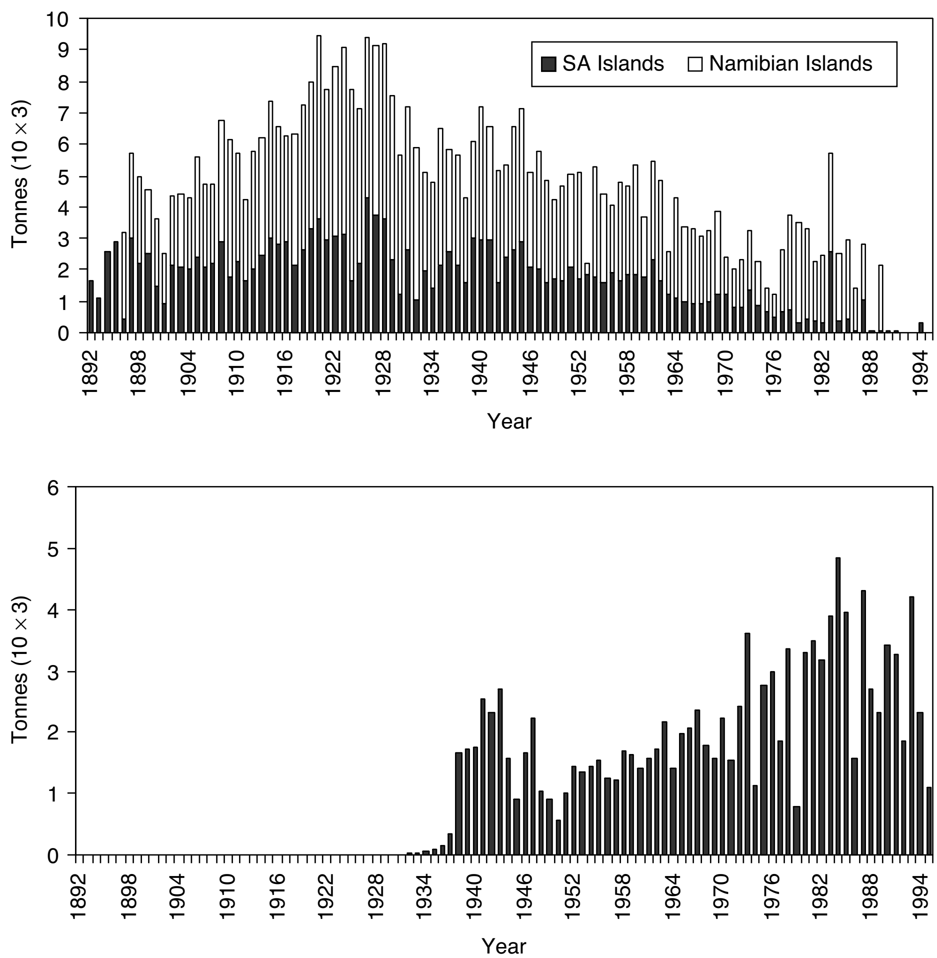

Figure 10 Collections of seabird guano at (top) islands off Namibia and South Africa and (bottom) platforms off Namibia, 1892–1995. Sardine stocks collapsed off Namibia in the 1970s and off South Africa in the 1960s. (Redrawn from Best et al. 1997.)

As a result of guano collecting Cape gannets had little material at islands with which to construct nests. This caused them to breed later and sometimes to nest on flat ground. Although there were attempts to offset this by placing phosphatic sand (collected from penguin colonies) at gannet breeding areas (Ross & Randall 1990), eggs were probably lost from nests constructed with too little guano (Jarvis 1970). Parts of some gannet and penguin colonies also became basin-shaped as a result of guano collection, allowing rainwater to accumulate, flooding nests and decreasing breeding success (Randall & Ross 1979). Collection of guano during the breeding season of penguins caused considerable disturbance at breeding sites. The removal of accumulated deposits of guano from islands also forced penguins to nest on the surface (Frost et al. 1976). This reduces breeding success, since burrows have a more constant microclimate than surface nests and shield birds from direct insolation, which may cause penguins to abandon nests (Randall 1983).

Populations of seabirds, especially kelp gulls *Larus dominicanus vetula* and great white pelicans *Pelecanus onocrotalus*, that ate the eggs or young of, or disturbed, those species that produced guano were reduced in the 20th century through deliberate destruction of eggs and chicks, shooting, and harassment (Crawford et al. 1982, 1995a). Control of kelp gulls (mainly the destruction of eggs and chicks) commenced in 1937, was discontinued as a policy in the early 1960s, but continued at some localities until 1978 (Crawford et al. 1982). It was recently reintroduced at two Namibian islands and at Bird Island in Algoa Bay.

The Western Cape population of great white pelicans bred at Robben Island in the early 1600s. At some stage in the next 250 yr it abandoned this island. As with penguins, exploitation and disturbance by humans were the most likely reasons. From at least 1869 until at least 1919, pelicans bred at Dyer Island, sometimes being persecuted by island staff to decrease their predation on guano-producing birds. Between 1894 and 1904, some bred at Quoin Rock, from where they were eventually displaced by Cape fur seals. From 1930–54, they bred at Seal Island (False Bay), where human disturbance included riflemen shooting at seals, commercial harvesting of seals, and use of the island for naval target practice. From 1956, they bred at Dassen Island, where they were left relatively undisturbed. The population in the Western Cape had been reduced to about 30 pairs between 1930 and 1956, but following cessation of persecution, it increased to 504 pairs in 1993 (Crawford et al. 1995a) and 603 pairs in 2000 (Figure 11).

The construction of the guano platforms off northern Namibia between 1930 and 1971 provided alternative nesting space for Cape cormorants and great white pelicans after sand islands in Cape Cross Lagoon and Sandwich Harbour, where they had previously bred, became joined to the mainland (Cooper et al. 1982). It is not known to what extent, if any, the construction of these platforms increased the populations of these two species off northern Namibia. However, construction of Bird Rock Platform north of Walvis Bay and the grounding of *Meisho Maru No. 8* near Cape Agulhas enabled extensions of the breeding range of crowned cormorants *Phalacrocorax coronatus* 415 km to the north and 16 km to the east. Various man-made structures, including jetties, disused boats, artificial islands in sewage or salt works, and roofs of buildings, have provided some seabirds with safe nesting habitat. Other natural sites have been rendered unsuitable for some species through, for example, islands being joined to the mainland to provide safe anchorage for fishing boats but thereby allowing access to mainland predators.

Up until the mid-20th century, factors influencing trends in seabird populations had mainly been those operating at breeding localities, such as harvesting of eggs and the collection of guano. Toward the end of the 20th century, seabirds were increasingly influenced by factors at sea, the most important of which were an altered supply of food, marine pollution, and incidental mortality of seabirds caused by fisheries (e.g., Crawford et al. 1995c). For some species the food supply decreased and brought about large decreases in population sizes. For others additional sources of food were made available.

The seabirds most affected by a decreased supply of food have been those that feed mainly on fish exploited by commercial fisheries. Colonies of African penguin between Lüderitz and Dassen Island and of Cape gannet in Namibia underwent massive decreases after collapses of sardine

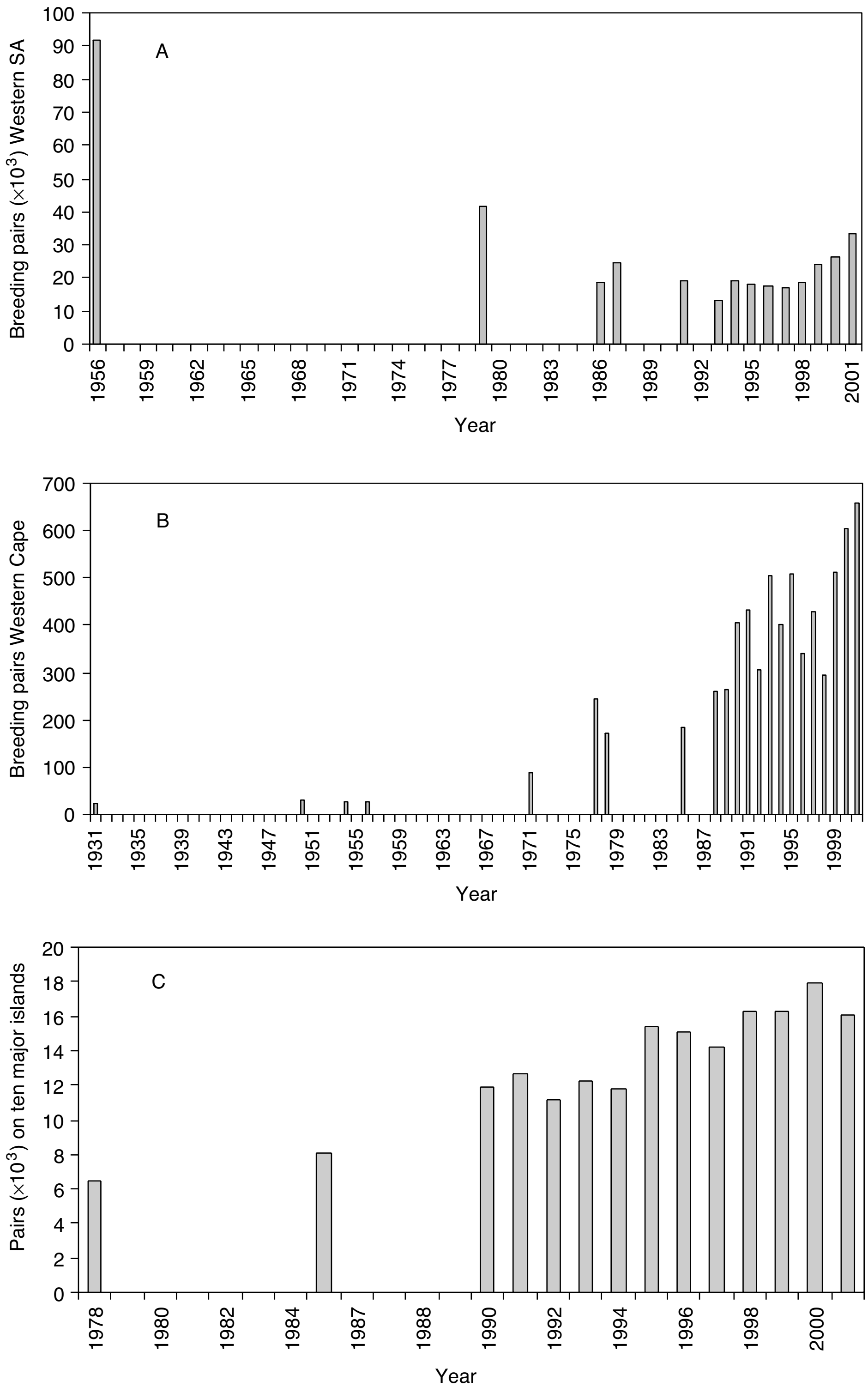

Figure 11 Trends in populations of (A) African penguins, (B) great white pelicans, and (C) kelp gulls in Western Cape in the 20th century.

Sardinops sagax resources off South Africa during the 1960s and Namibia during the 1970s. Off South Africa, sardine was replaced by anchovy *Engraulis encrasicolus* (formerly *Engraulis capensis*; Grant & Bowen 1998). Cape gannets switched their diet to this species. However, anchovy stocks on the Agulhas Bank were beyond the foraging range of African penguins breeding at colonies to the north of Table Bay, causing many young penguins to emigrate from the west coast to breeding localities on the Agulhas Bank. More recently, as South Africa's sardine stock has recovered, there has been a reversal of this trend. In Namibia, there was no replacement of sardine in the epipelagic zone. The population of African penguins at Namibian localities to the south of Lüderitz decreased from more than 40,000 pairs in 1956 to only about 1000 pairs in 2000. The area occupied by breeding Cape gannets at Namibian islands fell from 6.24 ha in 1956 to 0.63 ha in 1996. There was some emigration of young gannets from Namibia to islands off the Western Cape (Crawford et al. 1985, 2001, Crawford 1998, 1999). Bank cormorants decreased off the Western Cape following a decrease in production of Cape rock lobster *Jasus lalandii* (Crawford et al. 1999).

Additional food that has been made available to seabirds includes unwanted fish and offal discarded by fishing boats, food at coastal rubbish tips and abattoirs and fish present in freshwater impoundments that have been built in proximity to the coast (Berruti et al. 1993, Crawford et al. 1995a). These sources of food have been used especially by great white pelicans, Cape gannets, kelp gulls, Hartlaub's gulls *Larus hartlaubii*, and some of the nonbreeding visitors to the Benguela system (Ryan & Rose 1985). The populations of great white pelicans and kelp gulls increased in the Western Cape in the past two decades (Figure 11). This was probably a response to the lifting of controls on these populations. However, the overall supply of food for these two species may be greater now than previously, so their populations may rise above pristine levels. Because both species feed on the eggs and chicks of other seabirds (Crawford et al. 1995a, 1997), the reproductive success of prey species may be reduced below levels that would pertain in an undisturbed system. The eggs and chicks of penguins, which at many islands can no longer burrow into guano, are now more accessible to predators than previously. Availability of an artificial supply of food may have led to an extension in the breeding season of Hartlaub's gulls (Ryan 1987).

Oiling is a major threat to several seabirds. It causes feathers to clump, leading to a breakdown in their insulating properties. As a result, birds become hypothermic and are forced to leave cold waters. They dehydrate, mobilise stored energy reserves and may lose up to 13% of their body mass within a week (Morant et al. 1981). Unless rescued, they will eventually starve. Oil ingested by preening can also cause ulceration of the mouth, oesophagus, and stomach and, in severe cases, can lead to substantial blood loss. Oil absorbed into the system can cause red blood cells to rupture, leading to anaemia (Birrel 1994). Furthermore, an immunosuppressant effect makes birds more susceptible to diseases (Morant et al. 1981) such as pneumonia and aspergillosis. Ingested oil may produce a greater diversity of pathogenic bacteria. If a bird gets oil in its eye, it can lead to ulceration of the cornea and blindness unless treated (Crawford et al. 2000). From 1970–2001, more than 46,000 African penguins and 5000 Cape gannets were oiled (Morant et al. 1981, Adams 1994, Underhill et al. 1999, Crawford et al. 2000). Many have been successfully cleaned and returned to the wild, although there has also been heavy mortality. Some of the rehabilitated birds breed again in the wild (Randall et al. 1980).

There is limited mortality of African penguins from entanglement in fishing nets, discarded line and other material. There are unconfirmed reports of penguins being used as bait in rock lobster traps (Ellis et al. 1998). Off Namibia, Cape gannets are killed as accidental by-catch in longline fisheries. Tuna fisheries off southern Africa also kill albatrosses and petrels (Ryan & Boix-Hinzen 1998). Tourism to seabird colonies may decrease breeding success through disturbance.

The population of Cape fur seals in the Benguela system was reduced to low levels at the start of the 20th century, but has recovered during the 20th century (see seal section, p. 312). Increasing seal numbers adversely influence specialist seabirds in three ways: they compete for breeding space, compete for food and inflict mortality (Crawford et al. 1989, 1992, 2001).

In summary, humans have caused large decreases in the populations of abundant seabirds in the Benguela system, initially through unsustainable exploitation and later by competing with them for food. Seabirds have also been killed by oil spills and as incidental by-catch in fisheries. However, for some seabirds and seals, humans have created additional breeding habitat and additional sources of food. Populations of these opportunistic species increased during the 20th century, following cessation of uncontrolled harvesting or the removal of population controls and they in turn are now adversely influencing populations of some of those seabirds that have more specialised requirements for feeding and breeding.

Pelagic fisheries

The pelagic fishery in the Benguela region uses purse-seine nets to target visible, near-surface shoals of small pelagic fishes, which are generally caught at night. Once a shoal has been located, the net is set in a circle around it and the bottom of the net pulled closed by means of a foot rope. The net is then brought alongside the vessel and fishes are sucked out with a suction pump and transferred to the hold. Depending on the species targeted, the fishes are kept in refrigerated seawater or a strong brine solution. Once its hold is full, the vessel returns to harbour, where the catch is processed. Most pelagic fishing trips last a single night and landings of up to 200 t trip^{-1} are not uncommon. Because of the high degree of targeting and selectivity of purse-seine operations, by-catch of other species is generally low.

The anchovy *Engraulis encrasicolus* and sardine *Sardinops sagax* have been the dominant species of the Benguela pelagic fishery, with round herring *Etrumeus whiteheadi* and horse mackerel *Trachurus* spp. occasionally important. Off South Africa, where anchovy has dominated pelagic landings for the past three decades, over two thirds of anchovy caught are juveniles of around 6 months old, migrating from the west coast nursery grounds to the spawning grounds off the south coast. Anchovy, round herring, and juvenile sardine and horse mackerel that shoal together with anchovy or round herring are reduced to fish meal and fish oil. Adult sardine are either canned or frozen for human consumption or frozen for use as bait in other fisheries. Currently, the pelagic fishing industry operates primarily from ports along South Africa's west and southwest coasts, although commercial pelagic fishing also takes place from Mossel Bay and Port Elizabeth. In Namibia, the purse-seine fleet is based in Walvis Bay. As is the case for pelagic fisheries worldwide, that of the Benguela is characterised by high volumes and relatively low values compared to other fisheries.

Development of the South African pelagic fishery took place after World War II. Fishing for sardine and horse mackerel started in St. Helena Bay and soon expanded along South Africa's entire west coast. Rapidly increasing effort from the late 1940s led to a peak catch of almost 500,000 t in 1962, over 80% of which was sardine (Crawford et al. 1987). Catches of both species declined rapidly after 1962, the sardine stock collapse being ascribed to overfishing, expansion of fishing grounds to the south and variable recruitment (Beckley & van der Lingen 1999). Reduced horse mackerel catches were ascribed to the depletion in the population of the exceptionally strong year classes of 1946, 1947, and 1948 (Crawford et al. 1987). Since their collapse horse mackerel landings have consisted primarily of juvenile fishes and have remained low, generally less than 10,000 t yr^{-1}. Annual sardine catches were low and relatively stable at around 50,000 t during the period 1967–94 but have since shown a steady increase to 190,000 t in 2001. In 1964 the South African pelagic fishery switched to smaller-meshed nets to target anchovy, which then replaced sardine as the mainstay of the South African pelagic fishery and has dominated purse-seine catches since. Average catches of anchovy off South Africa have been in the region of 230,000 t yr^{-1} and have varied between 40,000 t in 1996 and 596,000 t in 1987. A time series of pelagic landings by major species is given in Figure 12.

Sardine was also the dominant species initially targeted by Namibian purse-seine vessels. The collapse of the South African sardine stock in the early 1960s resulted in increased fishing effort

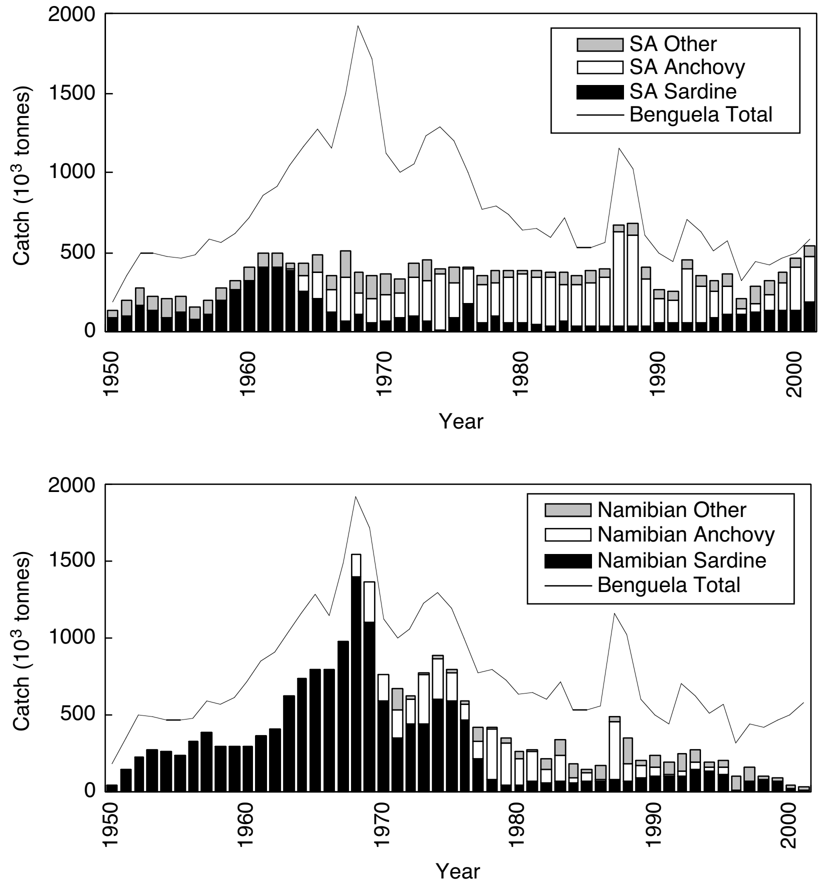

Figure 12 Pelagic fisheries catches in the Benguela ecoregion, 1950–2001. Top, landings of anchovy, sardine, and other species (horse mackerel, chub mackerel *Scomber japonicus*, round herring, and lanternfish *Lampanyctodes hectoris*) by the South African fishery. Bottom, landings of anchovy, sardine, and other species (horse mackerel only) made by the Namibian fishery. In both panels the total pelagic catch from the entire Benguela region is shown by the solid line. (Data for South Africa updated from Crawford et al. 1987; and for Namibia from Boyer and Hampton 2001, and A. Kreiner, Ministry of Fisheries and Marine Resources, Namibia, personal communication.)

off Namibia, and annual pelagic catches rose rapidly from around 200,000 t (composed entirely of sardine) in the 1950s to a maximum of 1.55 million t (1.4 million t of which was sardine) in 1968 (Boyer and Hampton 2001). In addition to the local purse-seine fleet operating out of Walvis Bay, factory ships from the eastern European midwater fleet, fishing outside territorial waters, also targeted sardine in the late 1960s. Poor catch records from this fleet indicate that the sardine catches reported during the 1960s must be regarded as a minimum (Boyer & Hampton 2001). After 1968, the Namibian pelagic fishery experienced a decline in landings to around 625,000 t in 1972, followed by slightly increased catches for a few years, before another abrupt collapse in the early 1980s (Figure 12). These collapses have been primarily attributed to overfishing, although poor sardine recruitment resulting from adverse environmental conditions exacerbated the decline. As in South Africa, smaller-meshed nets were used to target anchovy after the collapse of the sardine stock, with annual anchovy catches fluctuating around 200,000 t during the 1970s and 1980s. A pronounced peak of around 350,000 t in anchovy landings was recorded in 1987, the same year the highest anchovy catches were recorded off South Africa. Since then catches have rarely exceeded

50,000 t and have declined to zero following the anomalous environmental event recorded in the mid-1990s.

Overall, it can be seen that landings by pelagic fisheries in the Benguela ecoregion have fluctuated 10-fold over the period 1950–2001, between 185,000 t and 1.9 million t, with a long-term average of 770,000 t (Figure 12). Peak landings were made in the decade 1965–75, primarily by the Namibian purse-seine fleet (80% of total pelagic landings in 1968 and 1969). Whereas landings by South African purse-seine vessels have remained relatively constant over the past 50 yr, those by the Namibian fishery have exhibited a 40-fold variation. Pelagic landings in the Benguela region are no longer dominated by Namibia. Since 1979, over 50% of pelagic landings have been taken by the South African purse-seine fleet and this percentage has increased steadily. By 2001, 94% of total pelagic catches from the Benguela were made off South Africa.

Fluctuations in stock size

The large-scale fluctuations in population size and changes in the relative abundance of anchovy and sardine indicated by pelagic fishery catch data are typical of upwelling ecosystems. Analysis of scale deposits in sediments from upwelling systems suggests that these population fluctuations, or regime shifts as they have become called, are a natural phenomenon that occurred in the absence of fishing (Schwartzlose et al. 1999). Estimates of anchovy and sardine stock size in the Southern Benguela, derived initially from fishery-dependent data, but more recently from fishery-independent surveys, corroborate the catch data (Figure 13). Off South Africa, anchovy replaced sardine as the dominant small pelagic species following the collapse of the sardine population. This dominance lasted from 1965 until the mid-1980s, after which a steady increase in the sardine population resulted in populations of approximately equal size during the latter half of the past decade. Estimates of Namibian sardine biomass indicate that the stock collapsed from more than 11 million t in 1964 to well below 1 million t by the mid-1970s, and the population has not attained more than 500,000 t since (Figure 13). Unfortunately, biomass estimates for Namibian anchovy are not available. Unlike the Southern Benguela, however, anchovy in the Northern Benguela did not replace sardine to a great degree following the latter's collapse. After sustaining moderate catches from the 1970s to the mid-1990s, the anchovy population became severely depleted following the Benguela Niño of 1994–95 (Boyer & Hampton 2001).

Ecological impacts of the fishery

Because of their intermediate trophic level and massive population sizes, small pelagic fishes in upwelling ecosystems exert top-down control of zooplankton and also bottom-up control of predators such as other fishes and marine birds. Small pelagic fishes therefore constitute a crucial link in mediating energy flows between lower and upper trophic levels and, because of the low numbers of species comprising this group, have been termed *wasp-waist* populations (Cury et al. 2000). Their critical position means that changes in the abundance of these small pelagic fishes have substantial impacts on the ecosystem. For example, overfishing and collapse of the sardine stock during the 1960s were followed by collapses of colonies of African penguin along the west coast of southern Africa (Crawford 1998 and see above). Whereas the recovery of the Southern Benguela sardine population has resulted in a stabilisation of penguin colonies between Lüderitz and Table Bay, this has not arrested the overall decline in penguin numbers. The inability of the penguins to cope with recent shifts in dominance in prey species may have resulted from increased competition for food with fishermen during the 20th century (Crawford 1998). Similarly, the decline of the sardine stock off Namibia resulted in severe decreases in numbers of Cape gannets, which were not able to successfully exploit the mesopelagic horse mackerel and the goby *Sufflogobius bibarbatus* that replaced sardine (Schwartzlose et al. 1999).

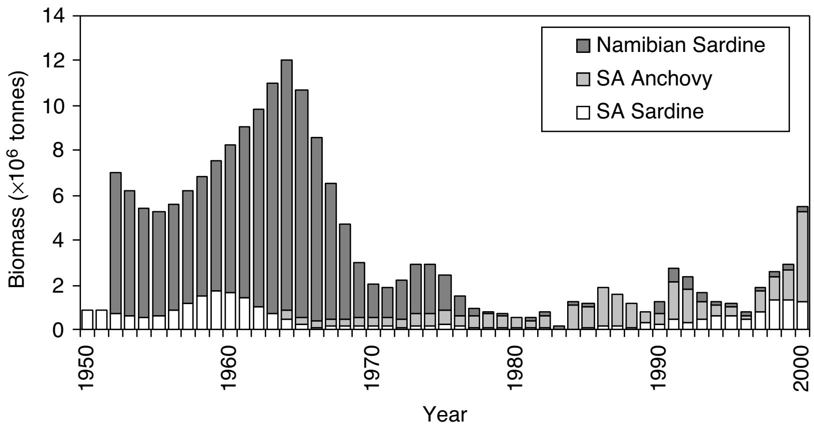

Figure 13 Estimates of biomass of some important species of pelagic fish in the Benguela ecoregion, 1950–2000. Estimates of South African sardine and anchovy biomass up to 1982 derived from virtual population analysis (VPA) and from 1984 onward from hydroacoustic surveys. Estimates for Namibian sardine were VPA-derived from 1952–85, and from hydroacoustic surveys from 1990–2000. Note that the VPA-derived estimates are considered to reflect only large-scale trends in abundance because of several limitations in the data (see Schwartzlose et al. 1999). (Data updated from Beckley and van der Lingen 1999, for South African and Namibian sardine; from Butterworth 1983, for anchovy VPA estimates; and from Barange et al. 1999.)

Explicit analyses of the ecosystem effects of pelagic fishing have been achieved through computer simulation in which a dynamic, mass-balanced trophic model (called Ecosim) of the Southern Benguela was subjected to various types of fishing pressure (Shannon et al. 2000). These simulations have suggested that increased fishing on small pelagic fishes in systems where wasp-waist (both top-down and bottom-up) control dominates results in major perturbations propagating through the ecosystem. Competing species are favoured through enhanced availability of zooplankton prey, and the increased abundance of these competitors delays recovery of the target species once fishing pressure is reduced.

While intense fishing on small pelagic fishes in the Benguela has had large and obvious ecosystem impacts, more subtle effects on the fish populations have been suggested, principally the erosion of intraspecific diversity (Cury et al. 2000). For example, the composition of pelagic fish schools has been shown to reflect the relative species abundance within the pelagic community. When anchovy or sardine are abundant they tend to form relatively pure schools, whereas when their abundance is low, they join schools of other more abundant species. These variations in average percentages of a pelagic species contained in schools track the overall relative population abundance remarkably well (see Figure 7 of Cury et al. 2000 for an example from the Southern Benguela) and have led to the development of the school trap hypothesis (Bakun & Cury 1999). This hypothesis postulates that "a fish species driven to school together with a more abundant species must effectively subordinate its specific needs and preferences to the 'corporate volition' of a school largely driven by a different set of needs and preferences." The school trap thus constitutes a mechanism for adverse population interaction between co-occurring small pelagic fishes and could maintain a collapsed population in a depleted state for lengthy periods, as well as affecting spatial dynamics such as migrations. Possible evidence for the school trap hypothesis has been suggested from a study of changes in the spawning habitats of small pelagic fishes off South Africa over the past two decades. During the 1980s and early 1990s, sardine and anchovy in the Southern Benguela showed a broad-scale overlap in their spawning habitats. From 1994 onward, however, their spawning habitats became markedly distinct, with sardine spawning principally off the west coast and anchovy spawning predominantly off the south and east coasts (van der Lingen et al. 2001).

This was also the first year of the acoustically estimated biomass time series when the relative biomass of sardine was higher than that of anchovy (Barange et al. 1999).

The school trap may not only operate between species, but intraspecifically as well. From the point of view of a fish population, a fishery represents nothing more or less than a massive source of predation pressure, and sustained pressure on a particular component of the population could result in removal at the population level of the tendencies expressed by the targeted component. Off Namibia, sardine primarily spawned near the coast between two intense upwelling centres (near Cape Frio and Luderitz) that appeared to provide the best available reproductive habitat, some spawning also taking place offshore, in the region of the Angola–Benguela front (Bakun 2001). The development of the pelagic fishery occurred directly within the inshore, primary reproductive habitat, and the inflated predation pressure arising from fishing may have resulted in a steady preferential removal from the sardine population of individuals with strong affinities for the primary reproductive habitat. As a result, essentially all of the Namibian sardine's reproductive output in recent years has been concentrated in the Angola–Benguela frontal zone (Bakun 2001). These two examples indicate that overfishing not only can alter the abundance of small pelagic fish populations, and hence have a substantial impact on ecosystem structure, but also can affect ecosystem functioning by altering the composition and spatial distribution of such populations.

Demersal and midwater trawl fisheries

The demersal fishery off southern African started in the early 1900s (Lees 1969). Initially the prime target species were Agulhas sole *Austroglossus pectoralis* and west coast sole *Austroglossus microlepis* (Payne & Badenhorst 1989), but this changed with the discovery of the vast hake resource off the west coast during the First World War (Payne & Punt 1995). From these humble beginnings, the demersal fishery, based on the Cape hakes *Merluccius capensis* and *Merluccius paradoxus*, has developed into the most valuable fishery in the region. The historical background to the hake fishery of southern Africa and the management of the stocks have been described in detail by Andrew (1986), Payne (1989), Punt (1991), Payne & Punt (1995), Gordoa et al. (1995), Boyer & Hampton (2001), and van der Westhuizen (2001), and hence only a brief summary is provided below.

Initially the demersal fishery was concentrated close to Cape Town and total annual landings of hake were around 1000 t (Lees 1969). The expansion of the fishery was initially slow, and hake landings were less than 50,000 t yr^{-1} by the end of the Second World War. Improved technology enabled the expansion of the fishing grounds and the annual hake landings began to increase more rapidly, reaching 50,000 t by 1950 and 160,000 t by 1960 (Payne & Punt 1995). Exploratory fishing by Japan and Spain in the early 1960s showed that catch rates were higher off Namibia than off South Africa (Gordoa et al. 1995). This resulted in the expansion of the fishery into Namibia waters, an increase in the foreign distant-water fleets, and a rapid escalation of hake landings, reaching over 1 million t by 1972 (Figure 14).

This rapid escalation of fishing effort prompted the establishment of the International Commission for the Southeast Atlantic Fisheries (ICSEAF) in 1972. Over the next few years ICSEAF introduced a minimum mesh size of 110 mm, a system of international inspection, and allocated quotas to member countries. South Africa declared a 200-nautical mile exclusive economic zone on November 1, 1977, and excluded all but a small amount of foreign effort, thereby reducing the hake catches off South Africa (Figure 14). South Africa then embarked on a rebuilding strategy for the Cape hake resource by setting conservative annual catch limits.

At that time, South Africa governed the ex-German colony of Southwest Africa (as Namibia was then known). In the absence of an internationally recognised government, an EEZ could not be instituted in the Northern Benguela. Nonetheless, some foreign fishing effort was withdrawn due to the falling hake catch rates. The international fishery off Namibia was managed by ICSEAF until independence in 1990. The newly recognised nation then declared a 200-nautical mile EEZ and

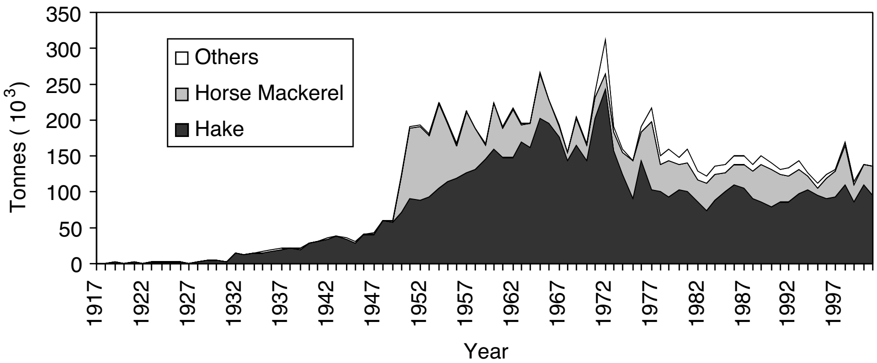

Figure 14 Catches taken by the demersal fishery in the Benguela, 1917–2000.

excluded all foreign fishing effort, resulting in a dramatic drop in hake catches (Figure 14). Since 1990 there has been a perceived gradual recovery of the hake resource and an increase in hake catches.

The wider ecological effects of the hake fishery are difficult to estimate. There are three main aspects to be considered: discarding of hake, catches of other species, and substratum damage. The minimum marketable size for hake was certainly greater in the early days of the fishery than today, resulting in a higher proportion of discards. The "official" hake catch statistics (Figure 14) prior to 1972 were thus increased by 39% in accordance with a decision made by ICSEAF in 1978 (Andrew 1986). In the 1980s the hake fishing industry was forced to develop a market for small hake because of the depleted nature of the resource. A consequence of the rebuilding strategy has been an increase in the availability of large hake, which has allowed the industry to diversify the market. There has been a gradual shift to deeper water as the industry has started targeting larger fishes (Glazer & Butterworth 2002). Walmsley (2004) estimated that 17,000–25,000 t of fish (including 7000–12,000 t of hake) were discarded in 1997 by the demersal trawl fleet off the west coast of South Africa. However, there are consistent rumours of high grading off South Africa, particularly by small quota holders trying to maximise the financial returns on their limited hake quotas.

Some by-catch species such as Agulhas sole, kingklip (*Genypterus capensis*), and monkfish (*Lophius vomerinus*) are sought after because they fetch a higher price per unit mass than hake. Other species (e.g., horse mackerel *Trachurus trachurus capensis* and ribbon fish *Lepidopus caudatus*) are retained or discarded depending on market demand, whereas a third group (e.g., grenadiers *Caelorinchus symorhincus* and *Malacocephalus laevis*) are unwanted incidental by-catch and are routinely discarded.

For a brief period in the early 1970s there was a directed fishery for west coast sole in the vicinity of the Orange River. The fishery was closed in the late 1970s due to a stock collapse and remains closed in South Africa, but there has been a small directed fishery off Namibia since 1994 (Boyer & Hampton 2001). In 1983 an experimental kingklip-directed longline fishery was started off South Africa (Japp 1988, 1989). Catches increased rapidly from 1042 t in 1983 to a peak of 8684 t in 1986; the fishery was closed due to collapse of the stock in 1990. Subsequent modelling showed that the kingklip resource was already under pressure as by-catch in the hake fishery before the longline fishery commenced (Punt & Japp 1994). There is a directed fishery for monkfish off Namibia (Boyer & Hampton 2001, Maartens & Booth 2001a,b) and some targeted fishing off South Africa (Walmsley et al. 2004).

Horse mackerel are semipelagic and are exploited by purse-seine nets and by both midwater and bottom trawls. After the peak hake landings off Namibia in the early 1970s, some effort was diverted to midwater trawl fishing for horse mackerel in response to declining hake catch rates (Payne & Crawford 1989). The midwater trawl fishery for horse mackerel subsequently became the largest fishery by volume off Namibia, with catches of around 500,000 t yr^{-1} between 1978 and 1987 (Boyer & Hampton 2001). Adult horse mackerel were exploited by purse-seine off the west coast of South Africa in the 1950s and 1960s, with catches peaking at 102,600 t in 1952 (Johnston & Butterworth 2001), but this resource has largely disappeared. Small quantities of adults are taken in bottom trawls and adults in spawning condition have been observed during research cruises to the west coast. (Note that the horse mackerel landings shown in Figure 14 include catches taken by midwater trawl off the south coast of South Africa.) It is generally assumed that the horse mackerel resource off the west coast of South Africa was a southern extension of the large Namibian resource. With the collapse of the sardine fishery off South Africa, some purse-seine effort was redirected toward juvenile horse mackerel, but catches have generally been small, except for 1989 and 1996, when the purse-seine catch of juvenile horse mackerel exceeded 25,000 t. The purse-seine catch of juvenile horse mackerel of the South African west coast is limited to 5000 t in terms of current regulations. However, in some years substantial quantities of juvenile horse mackerel (up to 100,000 t) are taken in pelagic purse-seine nets off Namibia (Boyer & Hampton 2001).

The historical proportion of hake in the catches is not known because only data on the landings (i.e., retained catch) are available. Walmsley (2004) found that the proportion of hake in the catches taken off South Africa increased with increasing depth. Therefore, one would expect that the proportion of unwanted by-catch was higher during the early stages of the fishery, when vessels were small and fished in shallow waters close to Cape Town. Payne & Punt (1995) show that the hake contribution to the landings off South Africa decreased from around 90% in the early 1960s to around 60% in the early 1990s. They attribute this decline in the hake proportion to a combination of channeling effort into mixed species fisheries and to landing a greater proportion of the by-catch.

Bottom trawls are unselective, and there is a growing body of literature on damage they cause to the seabed (see, for example, Hollingworth 2000). As a result, there is increasing pressure to develop less destructive fishing methods. A widely used alternative is longlines. These are lines of over a kilometre in length with hooked branch lines called snoods placed at regular intervals. An experimental longline fishery for hake was started off South Africa in 1983 and currently some 5000 t of hake are caught by this method off South Africa and 5000–10,000 t off Namibia. Although longlines are more selective than bottom trawls and do not damage the seabed, they are not without their ecological problems. In particular, longlines are responsible for a substantial mortality of seabirds (Barnes et al. 1997). Due to their very low reproductive rate and delayed onset of reproductive maturity, these *k*-selected species cannot withstand the observed levels of mortality (Ryan et al. 2002 and references therein). In addition, there is substantial loss of catch to predators such as Cape fur seal and through fish breaking off when the line is hauled in slightly rough weather.

Inshore net fisheries

Both beach-seine and gill nets are used by the Benguela inshore net fishery. Beach-seines are mobile nets, usually rowed out into the surf zone under the directions of a spotter, to encircle a shoal of fish (most commonly mullet *Liza richardsonii*). A crew of 6–30, depending on the size of the net (up to 275 m in length) and the length of the haul, then haul the head ropes shoreward. As the net approaches the shore, the ends, or wings, of the net are brought together, and the trapped fish driven into the bag in the middle of the net. Occasionally no spotter is used and a “blind seine” is made in areas or at times when fishes are likely to occur. Smaller 50- to 100-m beach-seine nets may also be deployed by walking out into the surf to encircle fish and are locally referred to as foot nets. Beach-seine fishing has remained essentially unchanged since the technique was introduced to South Africa during the mid-1600s. The only technological improvements relate to the use of woven nylon rather than cotton nets, glass fibre as opposed to wooden boats and four-wheel drive vehicles to transport the rigs on sandy beaches.

Gill netting is normally a passive form of fishing in which nets are deployed, usually from a boat, in the hope that fishes will swim into them and become entangled. Two main types of gill nets have been used in the region: positively buoyant, 44-mm stretch-mesh drift nets (used for catching *Liza richardsonii*) and negatively buoyant set nets set along the seafloor and buoyed and anchored at both ends. In the past, set nets of various mesh sizes (44–178 mm) were used, but since 1982 the only legal mesh size has been 178 mm, used to target St. Joseph shark *Callorhinchus capensis*. Although cotton and multifilament braided nylon mesh were used in the past, most modern gill nets are made from monofilament nylon mesh. Technological advances (such as outboard motors, echo sounders, and spotlights) have also allowed commercial gill netters to employ a more active type of gill netting. Shoals may be completely encircled, or the nets set in a semicircle in the path of the shoal. The fisher then scares the fishes into the net by revving the outboard motor and completing the circle behind the shoal.

Origins and history of inshore net fishing

The origins of inshore net fishing in South Africa date back to the use of beach-seine nets by English and Dutch seafarers during the 17th century (De Villiers 1987). By the end of that century, the Dutch had established beach-seine fishing outposts as far afield as Langebaan Lagoon (Poggenpoel 1996). Most of the catch was salt cured, dried, and used to supply the Dutch East India Company trading boats, troops, and slaves at the castle in Cape Town. With the expanding colonial settlement in the Cape, beach-seine fishing became widespread and was the most frequently employed fishing method in most areas. Thompson (1913) writes:

> Nearly every sea-board hamlet and coast farm have a net or two, whilst at Struis Bay, Keimouth and similar stations “trekking” is practically the only mode of fishing carried on. The St. Helena Bay and Saldanha Bay areas and False Bay are the chief centers of the seining industry, which in these localities has long been the means employed for dealing with the large shoals of harders, white stumpnose, elft, albacore and young white steenbras that are periodically on the move close inshore.

By this time the link between agriculture and beach-seine fishing was well established. Coastal farmers either worked their own nets or purchased fish from fishing settlements on the coast. The fishes were still processed into sun-dried “bokkoms” and given to farm labourers as “rantsoen vis” in partial payment for their work.

The first major threat to the supremacy of beach-seine nets as the gear of choice for inshore fishing, particularly along the west coast, was the introduction of gill nets by Italian immigrants during the late 1800s (Thompson 1913). User group conflict between the traditional beach-seine fishers and the new, more technologically advanced gill net fishers began to occur. Despite their admiration of the gill net, the state was also not keen to alienate the beach-seine fishers or farmers and drafted various regulations aimed at minimising conflict between the two sectors (Thompson 1913, Van Sittert 1992). During the early 1900s many of the Italian gill net fishers began to work in the expanding rock lobster fishery and competition for fish and the rantsoen vis market between the two groups decreased (Van Sittert 1992). By 1912, however, over 300 gill nets were in use in the Berg River and in the sea to the south of the river mouth (Van Sittert 1992). This indicates the extent to which these nets had been incorporated into the inshore fisheries along the west coast and beach-seining was no longer the dominant net fishing method.

The importance of inshore gill and beach-seine fishing along the west coast was further reduced with the start of the purse-seine fishery during the 1930s. Purse-seine fishers were more efficient at catching large quantities of harders *Liza richardsonii* and maasbanker *Trachurus trachurus* than beach-seine and gill net fishers. The large quantities of purse-seine-caught fishes thus flooded the rantsoen vis market, causing prices to collapse and considerable hardship for gill and beach-seine net fishers (Van Sittert 1992). This prompted many traditional beach-seine and gill net fishers to either join the purse-seine fishery or seek alternative employment. Increased demand for fish during the Second World War led to the construction of numerous canning factories and the purse-seine fleet began targeting the more abundant pelagic pilchard *Sardinops sagax* and anchovy *Engraulis encrasicolus*. Smaller purse-seine vessels did, however, continue to catch considerable quantities of *L. richardsonii*. Despite numerous complaints by gill and beach-seine fishers, management was slow to respond, officially requesting purse-seines not to target *Liza richardsonii* in 1973 and banning the practice in 1984.

Management of inshore net fisheries

After the early attempts to resolve conflict between gill and beach-seine fishers around the turn of the last century, the inshore net fisheries received little management attention for almost 50 yr. Any person could fish anywhere along the coast with gill or beach-seine nets, the only restriction being a minimum mesh size of 44 mm, aimed at preventing catches of juvenile *Liza richardsonii*. In 1967, the old conflict between beach-seine and gill net fishers again threatened to develop. An investigation by the then Sea Fisheries Research Institute revealed that the numbers of beach-seine and gill nets in use was increasing rapidly. This was possibly due to the collapse of the west coast sardine stock in the late 1960s and the subsequent need for purse-seine fishermen to once again supplement their incomes by inshore net fishing. With the growth of recreational line fishing, anglers and conservation-minded members of the public also became increasingly opposed to inshore net fishing, which they alleged threatened stocks of numerous fish species.

As a result, numerous control measures were introduced, including compulsory registration of all nets in 1974 (with permit holders required to submit daily catch returns on a monthly basis), gear and catch restrictions, and the restriction of net fishing to certain areas. A reduction of inshore net fishing effort became official policy and permits were only awarded to so-called bona fide fishers or pensioners with a history of participation in the fishery. The number of beach-seine permits issued for the popular angling areas of False Bay and Walker Bay were drastically reduced and gill netting was confined to the west coast (north of Cape Town). With the exception of False Bay, the inshore net fishery has received negligible management action since the early 1980s, although along with other South African fishing sectors, net fishing rights are currently under review and a substantial reduction in effort is likely.

Spatial distribution of effort

Although limited gill net fisheries targeting *Liza richardsonii* and other species appear to have developed around Cape Town and along the southwest coast, management measures aimed at restricting their use, in favour of beach-seines, were proposed as early as 1926. Gill net and beach-seine fishing were probably introduced to Namibian waters by South African pelagic fishers during the 1950s and 1960s, and a limited market probably constrained the expansion of the fishery beyond the ports of Walvis Bay and Luderitz. The area between Elands Bay and Langebaan on the west coast and False Bay and Walker Bay on the southwest coast were the main areas of net fishing activity for much of the last century.

It was only with the compulsory registration of gill and beach-seine nets in 1974 that information on the full extent of this net fishing activity became available. A total of 1303 net permits were issued to 593 permit holders, who operated between Cape Agulhas and Walvis Bay. In Namibia, only 17 net permits were issued initially, but this increased rapidly to 185 permits issued to 34 permit holders by 1978. By 1985, however, only six of these permit holders were reporting catches. Approximately 70% of all permits issued were for set or drift gill nets, the remainder being beach-seine permits. By far the majority (77%) were issued to fishers operating between Elands Bay and Langebaan on the west coast. Along the southwest coast 191 beach-seine permits were issued in 1974. In line with official policy to reduce the impact of beach-seine netting on so-called linefish species, the number of permits issued for the southwest coast was reduced by 66 to 64% by 1985.

There has been only a moderate reduction (23%) in the number of net permits issued for the west coast. Potential netting effort, however, actually increased by allowing all gill net permit holders north of Saldanha the option of using 178-mm set nets, in addition to their small-mesh harder nets. The numbers of permits issued also only reflect potential netting effort. A recent study has shown that a large number of permit holders (up to 40% in some areas) are inactive, or only operate a few times per year in an apparent recreational fashion (Hutchings 2000).

Long-term trends in reported catches

Anecdotal evidence indicates that historically a large portion of the inshore net catch comprised species other than *Liza richardsonii*. Linefish species such as white steenbras *Lithognathus lithognathus*, elf *Pomatomus saltatrix*, kob *Argyrosomus inodorus*, white stumpnose *Rhabdosargus globiceps*, yellowtail *Seriola lalandi* and galjoen *Dichistius capensis* contributed over 90% to the total beach-seine catch at Muizenberg (False Bay) around the turn of the century (Lamberth 1994). The contribution of these species to net catches has, however, decreased substantially since then, and *Liza richardsonii* made up over 77% of the total mass landed by the period 1977–87 (Figure 15). Despite legislation introduced in 1974 preventing the targeting of linefish by net fishers, illegal net fishers still land in the region of 150 t yr^{-1}. A recent survey (Hutchings 2000) indicated that at least 29 species are caught as by-catch in the legal gill net fishery along the west coast and contribute approximately 5% to the total catch. Lamberth (1994) recorded 65 by-catch species in False Bay beach-seine hauls, and beach-seines along the rest of the southwest coast probably have a similar catch composition

Although a variety of species were taken by inshore net fisheries over much of their history, it is impossible to separate these net-caught fractions of the catches from those made by other fishing sectors. Reporting of by-catch by net fishers has also been shown to be grossly inaccurate (Lamberth 1994, Hutchings 2000). It is therefore not possible to determine the impact of net fishing on the stock size of these species. However, *Liza richardsonii* species were caught almost solely by net fishing and some historical catch data for this species do exist.

Gilchrist (1914a) provides the earliest record of *Liza richardsonii* catches in the St. Helena Bay area. In this report, evidence of decreasing catches in St. Helena Bay is provided in the form of annual catches of adult and juvenile *Liza richardsonii* by Messrs. Stephan Bros. for the years 1880–1913. Assuming these are accurate (figures are not rounded off), the recorded catches for a 33-yr period provide a valuable insight into the net fishery in St. Helena Bay at the time. A drastic reduction in annual catches occurred, with an average annual catch prior to 1900 of approximately 102 t (calculated from a conversion ratio of five adults kg^{-1} and eight juveniles kg^{-1}), declining to only 16 t thereafter, a 85% decrease (Figure 16). It appears likely that the observed decline was due to the high fishing effort by both estuarine and marine net fishers. The particularly noticeable

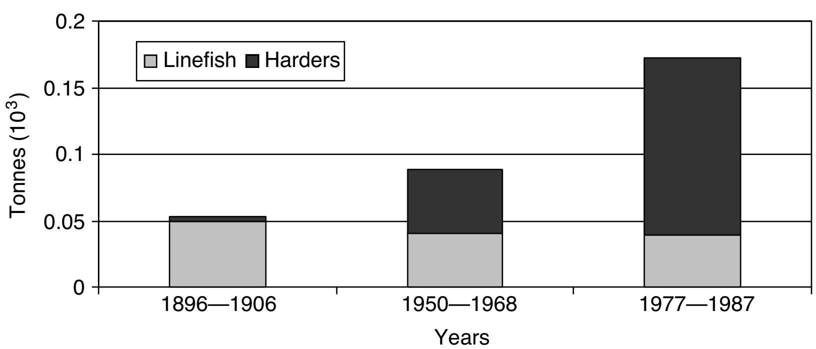

Figure 15 Beach-seine catches in False Bay for three different time periods. (After Lamberth 1994.)

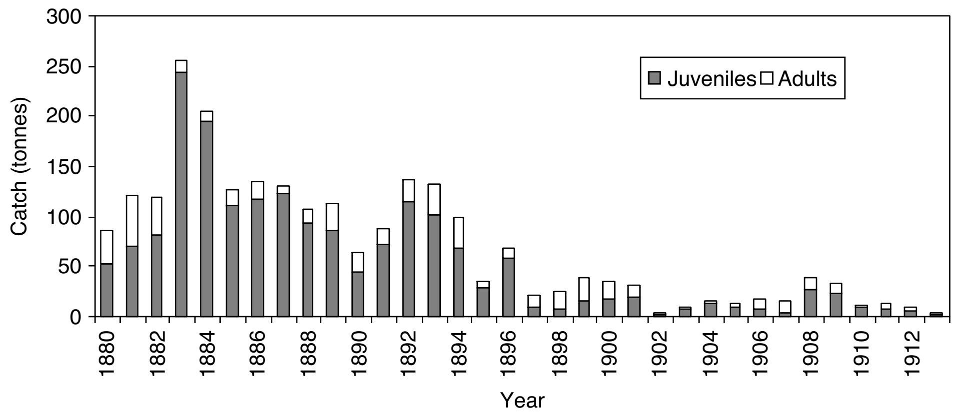

Figure 16 Recorded net catches of adult and juvenile mullet (*Liza richardsonii*) by Messrs. Stephan Bros. of St. Helena Bay, 1880–1913. (After Gilchrist 1914a.)

decrease in the number of juveniles caught suggests that a degree of recruitment overfishing had occurred.

Despite the decline in the St. Helena Bay catch for this period, the total *Liza richardsonii* landings for the region as a whole (Figure 17) rose to 1775 t for the period 1927–31 (Marine and Coastal Management, unpublished data), substantially more than the 259–323 t recorded for 1898–1900 (Gilchrist 1899, 1900, 1901). The total catch for the period 1974–99 appears to show a sustained decline since 1987–99 (Figure 17). These data, however, are based on compulsory catch returns by net fishers. Such returns have since been shown to be inaccurate, with many permit holders under- or overreporting catches or failing to submit returns. Indeed, as little as 8% of the total catch is reported in some areas, and the actual total catch of *Liza richardsonii* was estimated at around 5000 t for 1998 (Hutchings 2000). It is therefore unlikely that these statistics are accurate reflections of the real trends and data for this period should be treated with extreme caution.

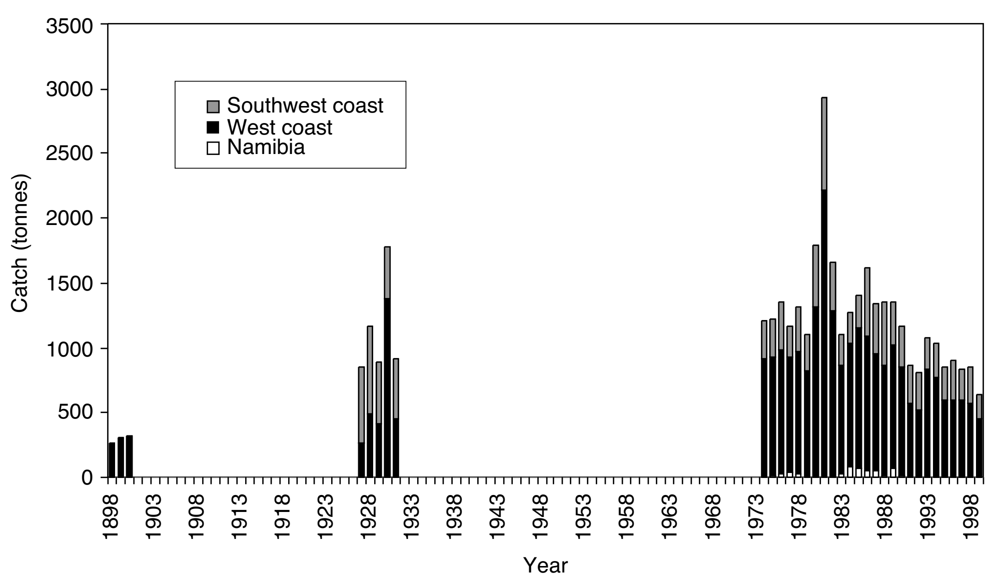

Figure 17 Total reported mullet (*Liza richardsonii*) catch over time.

Given the paucity of catch data for most of the net fisheries history and the inaccuracy of more recent catch statistics, it is impossible to make informed comments on the stock status of *Liza richardsonii* in the Benguela. Undoubtedly over 300 yr of inshore net fishing has substantially reduced the biomass of *Liza richardsonii* and of other species in the region. Indeed, there is compelling circumstantial evidence that the *Liza richardsonii* stock is overexploited in some areas, but there is simply insufficient data to quantify this.

Linefishes

Commercial linefishing is one of the oldest fishing industries in the Benguela system. Fleets of row- and sailboats were used for handline fishing off South Africa from the beginning of the 19th century and off Namibia about a half-century later (Kinahan 1991, Griffiths 2000). Although snoek (*Thyrsites atun*) comprised the mainstay of this fishery, warm-temperate species, chiefly croakers (Sciaenidae) and seabreams (Sparidae), have played substantial roles in warmer waters, both north of Walvis Bay and east of Cape Point. While a total of 15 species are targeted by commercial line fisheries in the Benguela, this review focuses on the six most important (based on annual catch). As with other line fisheries, catch and effort data are unevenly distributed in space and time. In order to accommodate this limitation, but also allow for the presentation of data for separate stocks of the same species, the study area was divided into the following regions: Southwestern Cape (Cape Point to Cape Agulhas), Western Cape (Cape Point to Orange River), and Namibia. The Namibian coastline was not subdivided because commercial vessels are based only at Swakopmund and Walvis Bay, operating between Meob Bay in the south and Rocky Point in the north.

Fishing effort in the Southern Benguela rose dramatically during the 20th century, numbers of commercial vessels increased 10-fold in the Southwestern Cape and 20-fold in the Western Cape. This trend was not, however, reflected in the northern part of the system. During the 1990s the average numbers of active commercial vessels were 577 in the Southwestern Cape, 986 in the Western Cape, and 10 in Namibia, equating to 1.2, 2.1, and a mere 0.006 vessels km^{-1} coastline, respectively (Griffiths 2000, Holtzhausen, personal communication).

Overall catches of the six main target species over three periods for which adequate data are available are shown in Figure 18. Catches in the Western Cape, which are dominated by snoek, have increased dramatically over the past century, whereas those on the Southwestern Cape, which are made up of a more diverse mixture of species, have declined over the same period. The reasons for this become more evident when examining the biology of the component species individually (see below).

Snoek (Thyrsites atun)

Snoek is a medium-size, pelagic predator attaining a weight of 9 kg (Nepgen 1979) inhabiting the coastal waters of the temperate Southern Hemisphere. It is found off southern Africa, Australia, New Zealand, the east and west coasts of southern South America, Tristan da Cunha, and the islands of Amsterdam and St. Paul (Nakamura & Parin 1993). Southern African snoek have been recorded from northern Angola to Algoa Bay, but are mostly found between the Cunene River and Cape Agulhas, i.e., in the Benguela ecosystem. This snake mackerel is a valuable commercial species and an important predator of small pelagic fishes. It apparently has two separate subpopulations in the Benguela ecosystem, one off Namibia and the other off South Africa, with medium-term (c. 5 yr) exchange in response to environmental events and food availability (Griffiths 2003). Taken with handlines since the early 1800s, *Thyrsites atun* became a significant by-catch of the hake-directed trawl fishery that developed during the 20th century. Initially discarded, snoek were retained by trawlers only after 1972.

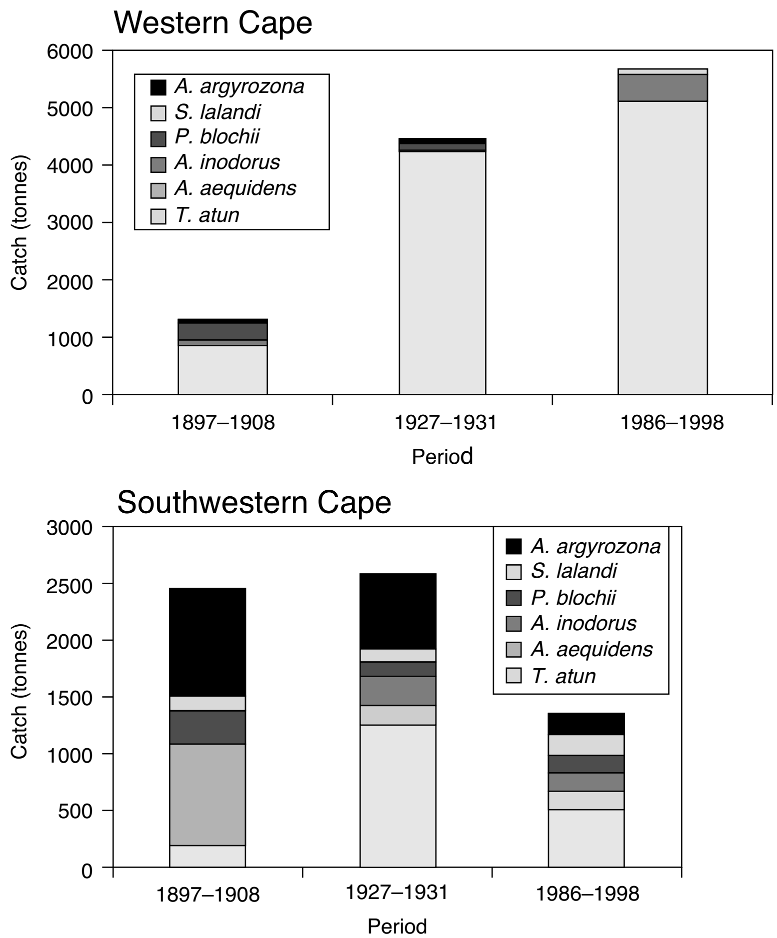

Figure 18 Mean annual commercial handline catches during three periods of the 20th century of the six most important linefishes of the Southern Benguela: carpenter (*Argyrozona argyrozona*), yellowtail (*Seriola lalandi*), hottentot (*Pachymetopon blochii*), silver kob (*Argyrosomus inodorus*), geelbek (*Atracoscion aequidens*), and snoek (*Thyrsites atun*). (Constructed from data presented by Griffiths 2002.)

The catches of snoek can be analysed in a number of ways: in terms of total domestic and foreign catch, capture location, or method of capture. Total annual catches from the Benguela (Figure 19A) increased markedly from 5000–19,000 t in 1960–77 to 26,000–82,000 t in 1978–90 with the influx of foreign trawlers in 1978, and then dropped again to 12,000–25,000 t following their withdrawal in 1991. The proportion of the catch made in Namibian waters (Northern Benguela) rose commensurately from 10–20% to 40–80% during the period of foreign involvement, and then declined to 5–9% after they were excluded (Figure 19B). Looking at the method of capture (Figure 20), handline catches of snoek in the Southern Benguela during the 20th century, despite high interannual variation, demonstrate some noteworthy trends. Annual catches increased concomitantly with the increasing effort between 1896 and 1950. Between 1950 and 1978 they fluctuated between 2500–13,000 t, and between 1979 and 1999 they showed a steady increase from a low of 2000 t to around 18,000 t. Factors such as the abolishment of a 5-month closed season (1960s–1981) in 1981 and increased targeting of snoek following the demise of other linefishes (Griffiths 2000) may have played a role in the latter. Annual trawled by-catch of snoek (Figure 20A) made by South African vessels in the Southern Benguela also fluctuated considerably, but increased generally from around 1500 t in 1972 to 15,600 t in 1991. This peak was probably caused by the redirection of

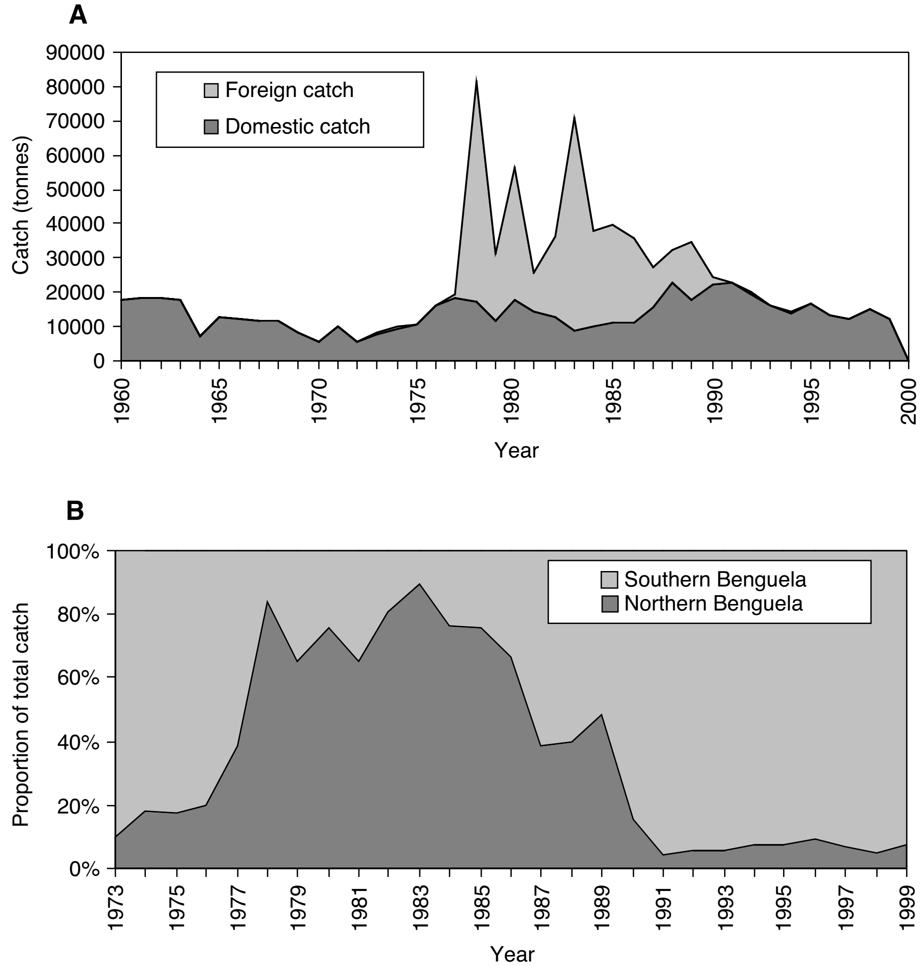

Figure 19 (A) Domestic and foreign snoek (*Thyrsites atun*) catches in the Benguela ecosystem. (B) The division of annual catch between northern and southern subregions (1973–1987). Data were obtained from International Commission for the Southeast Atlantic Fisheries statistical bulletins (ICSEAF 2 & 17) and *FAO Yearbooks of Fishery Statistics* (FAO 46, 52, 70, 88/1). Catch for the (ICSEAF) division 1.5 (1973–1987) was divided according to the ratio of Namibian and South African coastlines comprising the region. Because FAO yearbooks do not report catch by subregion within the Benguela, country totals for the period 1988–90 were assigned to the Northern and Southern Benguela based on previous fishing patterns (i.e., Israeli and Portuguese catches assigned to the south and all other foreign catches to the north); foreign involvement ceased after 1990.

excess trawl effort at snoek and horse mackerel when South African vessels were excluded from Namibia (the South African hake catch is limited by a TAC and therefore could not accommodate the extra effort). A steady decline in the annual trawl catch of snoek since 1991 is attributed to a general increase in operational depth, associated with a recent focus on the production of prime quality hake for overseas markets, rather than with declines in snoek abundance. Given the influences of effort levels and fisher behaviour on the trawl and handline catches of snoek, it is impossible to draw any conclusions on the long-term abundance of the resource. However, handline catch per unit of effort (CPUE) during the 1990s (available for the periods 1897–1906, 1927–31, and 1986–98) was only 42% lower than the historical maximum (1927–31) (Table 1) suggesting that the stock is currently not overexploited (Griffiths 2000). Handline catches off Namibia, on the other

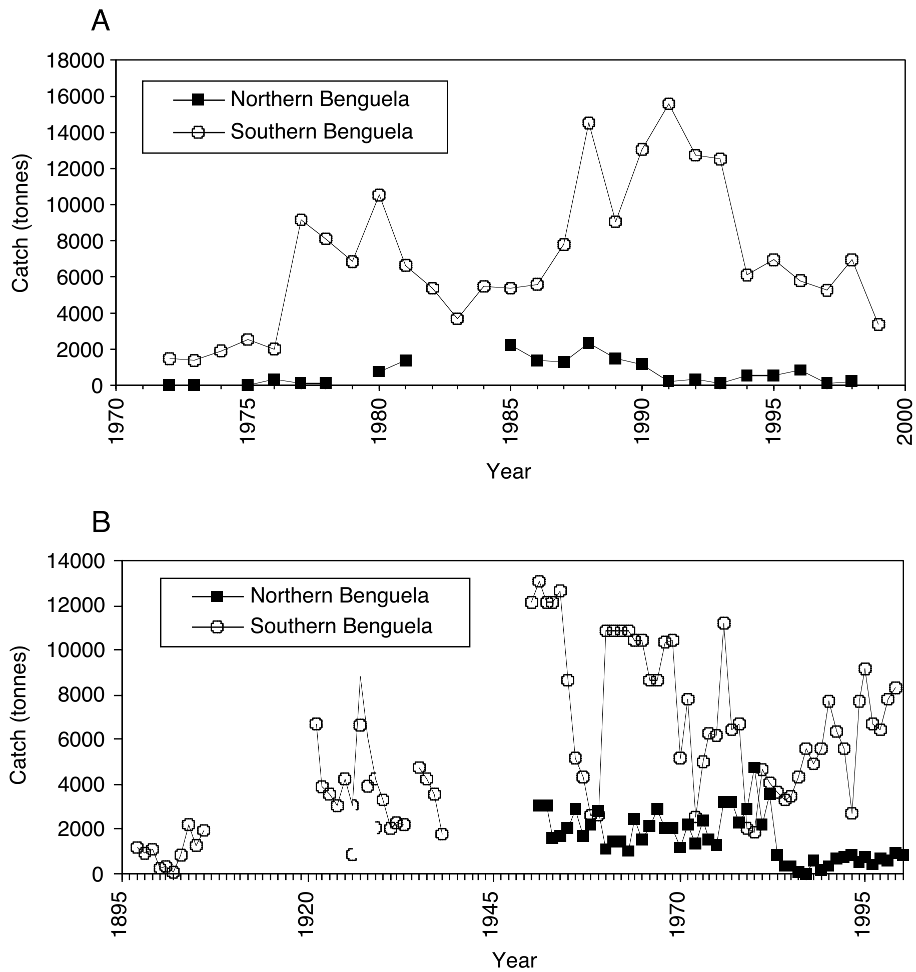

Figure 20 Annual (A) trawl and (B) handline catches of snoek made in South African and Namibian waters by domestic vessels.

Table 1 Vital statistics available for linefish stocks of the Benguela ecosystem

Species	Area	% CPUE	Stock assessment	Reference
Snoek	Namibia		?	Griffiths 2000
	South Africa	57.6	?	
Silver kob	Namibia		B = 40%	Griffiths 1997b, 2000, Kirchner 2001
	Southwestern Cape	9.2	SB/R = 8%	
Geelbek	Southwestern Cape	2.5	SB = 5%	Griffiths 2000, Hutton et al. 2001
Carpenter	Southwestern Cape	3.2	?	Griffiths 2000
Hottentot	West Coast	22.4	?	Griffiths 2000
	Southwestern Cape	37.9	?	
Yellowtail	South Africa	59.8	SB = 45%	Griffiths 2000, Penney 2000

Note: Data include current catch per unit of effort expressed as a proportion of historical peak (% CPUE) and stock assessment details. Remaining biomass (B), spawner biomass (SB), and spawner biomass per recruit ratios (SB/R) are given as percentages of pristine under the stock assessment column.

hand, dropped dramatically from 1500–4700 t (1950–83) to <1000 t in 1984, following the period of high foreign catches in the region. Lack of any indication of recovery may well have been the result of the persistent low levels of primary prey biomass, particularly clupeoids, since 1980 (Boyer & Hampton 2001).

Silver kob (Argyrosomus inodorus)

Silver kob is a medium to large sciaenid attaining a weight of 37 kg (Griffiths & Heemstra 1995). Two stocks have been identified within the Benguela ecosystem: Meob Bay to Cape Frio (Kirchner & Holtzhousen 2001) and Cape Point to Cape Agulhas (Griffiths 1997a). Although annual yield from the Southwestern Cape stock declined by 40% during the 20th century, CPUE dropped by at least 90% (Table 1 and Figure 21). This evidence of severe overexploitation is further supported by an estimated spawner biomass per recruit ratio of as little as 8% of pristine and an instantaneous fishing mortality rate threefold that of natural mortality (Griffiths 1997b). Owing largely to the lower commercial effort (above), and also to the fact that around 80% of the Namibian coastline is inaccessible to shore-based recreational activity (due to mining operations), this silver kob stock is considerably more healthy. Annual commercial handline catches of Namibian silver kob fluctuated between 78 and 2800 t between 1964 and 2000, depicting no clear trend. Remaining biomass is estimated at around 9000 t, or 40% of pristine (Kirchner 2001). It should, however, be noted that a commercial ski boat fishery established in the Swakopmund area in the early 1980s has since collapsed (Holtzhausen, personal communication), indicating a case of localised depletion. Recreational fishers catch approximately 30% of the estimated 1200-t annual catch (1995) of silver kob in Namibia (Holtzhausen et al. 2001). Most of the annual catch from the Southwestern Cape stock is taken by the commercial handline fleet (90.8%), with the remainder divided between recreational anglers (7%) and the beach-seine fishery (3.8%).

Geelbek (Atractoscion aequidens)

The geelbek is a predatory sciaenid that attains a weight of 15 kg and feeds almost exclusively on sardines and anchovy (Griffiths and Hecht 1995). Once the most important linefish of the Southwestern Cape (Figure 18), catch rates in this region declined by more than 95% during the 20th century. Although geelbek caught in the Southwestern Cape form part of a single South African stock, similar declines in CPUE throughout its range confirm the severe stock depletion (Griffiths 2000). Stock assessment indicates that spawner biomass had declined by at least 95% in 1996 due to overfishing (Hutton et al. 2001).

Carpenter (Argyrozona argyrozona)

Endemic to the warm-temperate waters of South Africa, this sparid reef fish is the second most important linefish of the Southwestern Cape (Figure 18). Recent tagging studies suggest that carpenter of the western and central Agulhas Bank comprise a single stock (Griffiths and Wilke 2002). Despite its current importance, catch rates of carpenter on traditional linefish grounds of the Southwestern Cape and Southern Cape (Cape Agulhas to Plettenberg Bay) have declined by as much as 97 and 95%, respectively (Table 1; Griffiths 2000). Stock assessment indicates that the reproductive potential (egg per recruit) of the western carpenter stock has been reduced to between 6 and 14% of pristine levels (Brouwer and Griffiths, unpublished data).

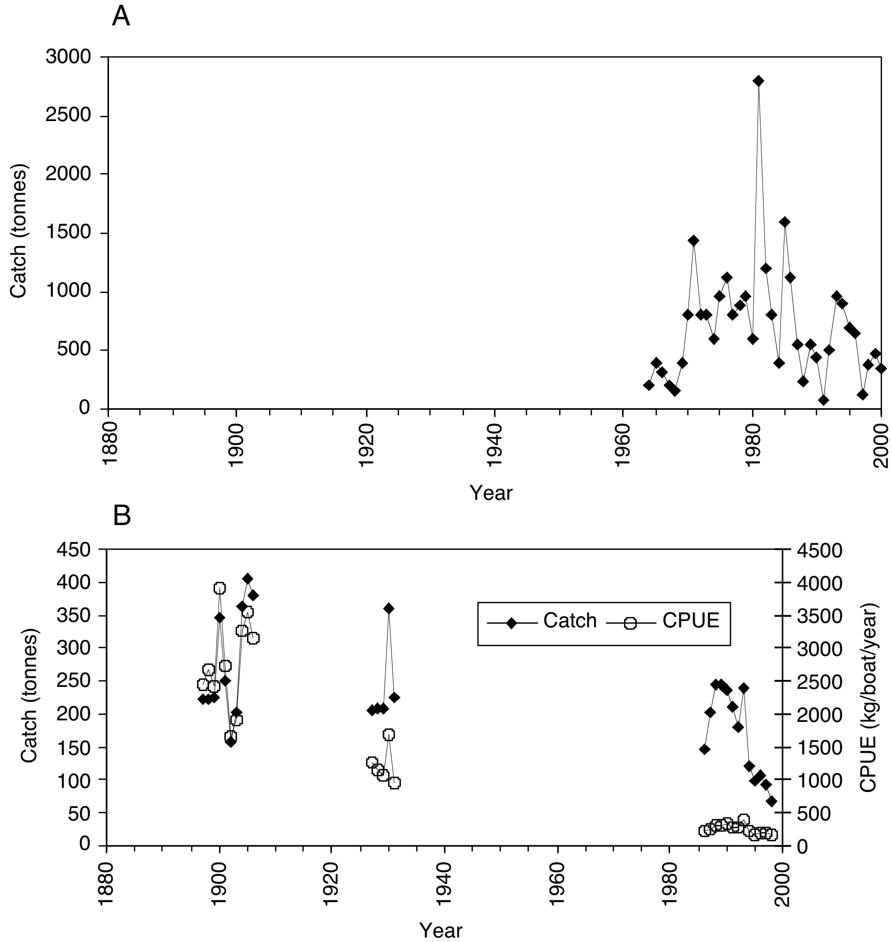

Figure 21 Commercial handline catch and catch per unit of effort for silver kob (*Argyrosomus inodorus*) stocks in (A) Namibia and (B) the Southwestern Cape.

Hottentot (Pachymetopon blochii)

Hottentot is an omnivorous cool-temperate sparid attaining a weight of 3 kg and inhabiting kelp forests and subtidal reefs to a depth of about 40 m. It is most abundant between Cape Agulhas and Luderitz in southern Namibia. Although targeted when larger and more lucrative species are less abundant, catch rates of this resident endemic fish have declined by 77 and 62% in the Southwestern Cape and Western Cape, respectively (Griffiths 2000). A quantitative assessment of this resource in 1985 (Punt et al. 1996) indicated that, with the exception of Gans Bay, hottentot were not overexploited. However, the growth rate used in this analysis (Pulfrich & Griffiths 1988) was overestimated, as it was based on whole (vs. sectioned) otoliths (maximum age being underestimated by 30%), and further assessment is required.

Yellowtail (Seriola lalandi)

This circumglobally distributed carangid, which attains a maximum weight of 34.2 kg, comprises a number of disjunct populations. The southern African population occurs along the entire South African seaboard and is found from the shore to the edge of the continental shelf. It is, however, most abundant in the area extending from the central Agulhas Bank (within the 200-m contour) to the central Western Cape. Tagging studies reveal that fish move freely within this range and that they comprise a single stock (Wilke & Griffiths 1999). Although a traditional linefish in South Africa, purse-seining for yellowtail was introduced in 1972. Commercial handline catch rates around Cape Agulhas dropped by two orders of magnitude by the early 1980s, but then recovered after netting was banned in 1982 (Penney 2000). Stock assessment (virtual population analysis (VPA) with extended survivor analyses) indicates that the stock is presently optimally exploited (Penney 2000).

Overview of linefish resources

Stock assessments and trends in CPUE reveal that, with the exception of snoek and yellowtail, other linefish species targeted in the Southern Benguela were heavily overexploited during the last century. Factors contributing to their demise have included unchecked commercial effort and biological factors such as predictable locality (coastal migrants and resident reef fishes), longevity (12–25 yr), and late maturity (7–55% of maximum age) (Griffiths 2000). Plans to rebuild South African linefish stocks include drastic reductions in commercial effort and the introduction of more stringent bag limits for recreational anglers.

Apart from the loss of long-term yield, overfishing of teleost predators, particularly in the Southwestern Cape, is thought to have had other effects on the Benguela ecosystem. Sardine *Sardinops sagax* and anchovy *Engraulis encrasicolus* are important food of pelagic and several reef-associated linefishes. These two prey species spawn on the Agulhas Bank and utilise the Benguela current to transport eggs and larvae onto nutrient-rich west coast nursery areas (see pelagic fish section, p. 323). Thereafter, the juvenile sardine and anchovy return to the Agulhas Bank, where they are spawned, thereby biologically pumping energy/carbon from the highly productive west coast upwelling system onto the eastern seaboard. This process presumably allows warm-temperate reef ecosystems to support larger shoals of piscivores than the reefs themselves could sustain. It is therefore no accident that the most important reef-associated species have been those that feed on clupeoids, i.e., carpenter, silver kob, and geelbek. The extent to which these linefishes may have fertilised 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 food web. Ecosystem modelling is regarded as imperative if the impact of overfishing of teleost predators is to be fully understood.

Rock lobsters

Historically, the commercial fishery for west coast rock lobster, *Jasus lalandii*, has extended from Sylvia Hill, north of Luderitz, to Cape Hangklip in False Bay (Figure 1). The species occurs over a wide depth range, from intertidal rock pools to a few hundred metres depth, the depth of occurrence extending deeper in the southern than in the central and northern parts of the range. However, there is some uncertainty as to whether these differences have always been as extreme as they are today.

Because of their shallow habitat, rock lobsters have long been targeted by coastal dwellers. Archaeological records show that they formed a component of the diet of early indigenous inhabitants on the west coast (Grindley 1967, Parkington 1976; see above, p. 307). Exploitation of lobsters for personal use and sale continued after colonisation, especially among the poor in the community; indeed, an active fishery for rock lobsters by licenced recreational fishers persists to this day.

The first commercial processing plant was established in South Africa in 1875 to can rock lobster tails, but teething problems got in the way of the new industry and exports only developed continuity in the 1890s. At that stage, and for the next 70 yr, before traps were introduced into the fishery in the 1960s, the entire west coast rock lobster catch was made using baited hoop nets (otherwise known as ring nets or drop nets), mostly fished from wooden dinghies, with one person rowing and another setting and hauling the nets. Between 1890 and the early 1920s the industry expanded with the building of rock lobster canning factories in other towns along the west coast, as far north as Luderitz. Expansion was particularly rapid after the First World War (Figure 22), with landings in both the South African and Namibian fisheries showing rapid growth to the start of the Second World War, at which point exports declined. This downturn was short-lived, and the industry soon took off with renewed vigor to reach peak landings of over 20,000 t yr^{-1} in the early and mid-1950s (Figure 22). Since the mid-1950s, rock lobster production in both countries has declined to a small fraction of its peak. The downward spiral has tended to be by a series of steps, which in most cases has been due to adjustments to production/TAC quotas and minimum sizes. These need to treated separately for the two countries.

A production quota was first imposed on the South African west coast rock lobster fishery in 1946. Landings for the fishery derived from export production, and in later years, landed mass figures show that the quota did little to constrain catches between 1946 and the mid-1950s. Very large catches were recorded between these years, particularly in 1950 and 1951, when they exceeded 16,000 t (Melville-Smith & van Sittert, in press).

Industry production quotas were introduced in 1946. In 1970 these were superseded by individual company quotas and, from 1980 onward, by regional TACs for eight regional fishing zones (Pollock 1986). Periodic, generally downward adjustments to quotas and TACs have been a feature of the fishery since the 1950s, as it became clear that the high landings of earlier years were not sustainable. The fishery experienced significant changes in the 1990s resulting from a sudden decrease in growth increments that led to poor recruitment into the fishery. The low catches of legal-size animals led to radically revised management arrangements for the fishery in the early 1990s. Since 1993–94 the minimum size has been fixed at 75 mm carapace length (CL), which is 14 mm smaller than the 89-mm minimum size first introduced 60 yr earlier (Figure 23). The TAC has also been decreased to around half of what it was in the 1980s.

From its inception in the early 1920s, the Namibian fishery has been subject to management restrictions of one form or another. The first legislation governing rock lobster fishing in Namibia

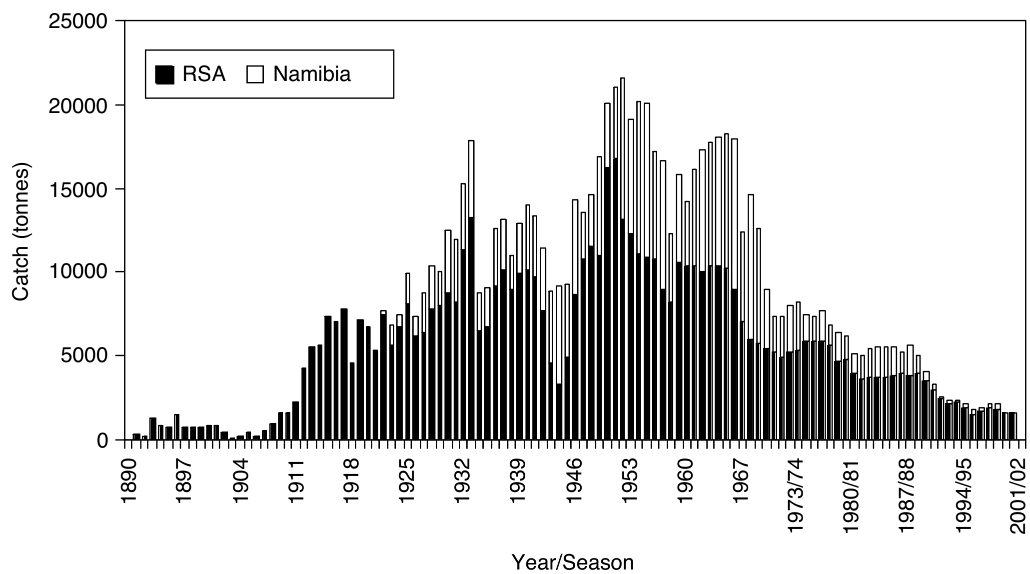

Figure 22 Annual commercial landings of west coast rock lobster (*Jasus lalandii*) in the Benguela ecoregion, 1890–2001. RSA, Republic of South Africa.

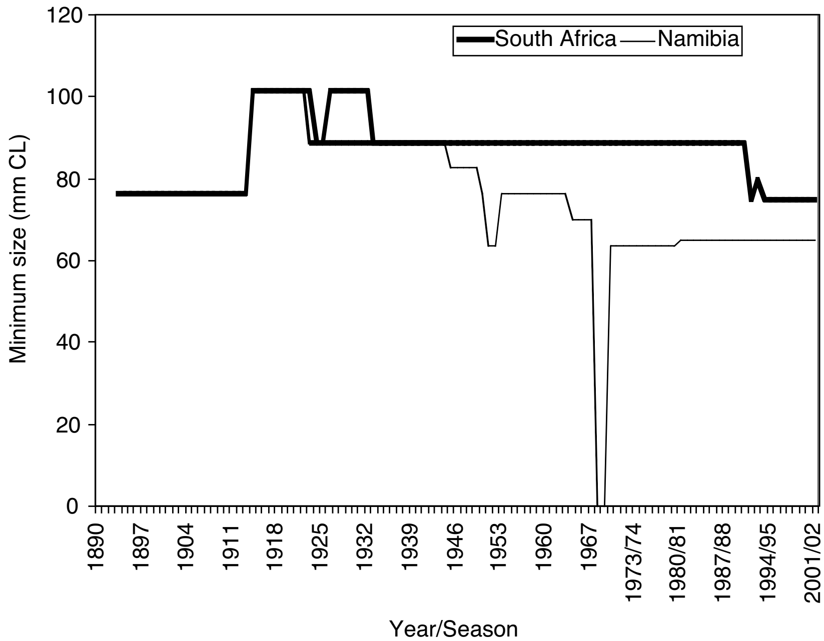

Figure 23 Legal minimum sizes for west coast rock lobsters (*Jasus lalandii*) in Namibia and South Africa, 1890–2002.

was introduced in 1922 in the form of a legal minimum size of 4 inches (101.6 mm) and a closed fishing season. Both the minimum size limit and fishing season were changed the following year, but the fact remains that conservation measures were recognised and imposed early on in the fishery. The proclamation of sanctuary areas in 1924, legislation limiting the number of factories licenced for processing lobsters, and production quotas in 1940 followed. Unlike in South Africa, where quotas limited landings, in Namibia the minimum size limit and number of processing factories appear to have had the most significant effects on historical landings, rather than production quotas (which, as shown by Beyers & Wilke 1990, were seldom attained).

The increase in landings through the early and mid-1950s followed an extra factory coming onstream in 1949, and several changes to the legal minimum size in the early 1950s (Figure 23). These included a period in the 1951 and 1952 seasons during which fishers were required to land all animals caught, the sublegal-size ones being sold on the local market, in effect eliminating the legal size limit (Beyers & Wilke 1990).

Further increases in landings in the mid- to late 1960s followed the granting of another factory licence in 1964, thereby increasing fishing effort, and a reduction in the minimum size. When catches declined further, the minimum legal size limit was abolished altogether for 1968 and 1969 (Beyers & Wilke 1990). The marketing of small tails proved problematic and further declining catches led to the implementation of a new size limit and a big reduction in the production quota in 1970. Despite these new arrangements, landings remained depressed compared to earlier years and slowly declined to the beginning of the 1990s when, over the space of two seasons, catches plummeted (Figure 22) as a result of very low catch rates. Since the 1992–93 season catches have been constrained by small TACs of under 300 t.

Over the years there has always been an unquantifiable portion of the landed catch that has gone unreported. These unreported landings, which are probably a significant portion of the reported landings, have taken diverse forms:

1. Rock lobster caught for "fishermen's bait" was exempt from regulations imposed on catches made for export until 1920.
2. Canned lobster, which dominated landing figures up to the 1950s, was (particularly in the early years) prone to high product rejection rates due to swelled or blown cans (Von Bonde & Marchand 1935).
3. Illegal fishing has been an important informal supplier of west coast rock lobster to domestic markets.
4. Illegal catches have traditionally been used by unscrupulous inshore fishers for bait or consumption at sea.
5. There have been eras when, at least in certain regions of the fishery, regulations were inadequately enforced (Pollock 1982).

Since 1983 it has been necessary for South African recreational rock lobster fishers to purchase licences, and this database has been used to conduct telephone surveys of recreational fishers (Cockcroft & Mackenzie 1997). Since the introduction of these surveys in 1991 to the 1995–96 season, the recreational catch in South Africa has been around 400 t season^{-1}. That in Namibia was at least 9 t in 1993–94 (Melville-Smith et al. 2000). The lack of continuity in these estimates has led them to be excluded from Figure 22.

Impact of fishing on the stock

West coast rock lobsters have a larval life of about 9 months (Silberbauer 1971), during which the larvae are found as far as 300–460 km offshore (Lazarus 1967, Pollock 1986). This lengthy life cycle provides ample time for mixing of the larval pool, and it is therefore reasonable to assume that Namibian and South African west coast rock lobster form a single stock.

The rise and fall of rock lobster landings in the two countries are remarkably similar (Figure 22). Both show rapid growth between the two World Wars. The decrease during the Second World War was followed by even larger landings, which peaked in the early 1950s. There is little doubt that by this time the fishery was being grossly overexploited, with catches in both South Africa (Melville-Smith & van Sittert, in press) and Namibia (Beyers & Wilke 1990) well over the respective quotas, and efforts to maintain production in Namibia were desperate enough to reduce the minimum size (Figure 23). Although CPUE in the central grounds of the South African west coast rock lobster fishery was stable between the mid-1950s to mid-1960s, catch rates had fallen from much higher levels in the early 1950s and effort had increased (Heydorn et al. 1968). A particularly dramatic decline in catches occurred in both Namibian and South African fisheries between the mid-1960s and early 1970s, although in the case of the Namibian fishery this decline was temporally masked by the abolition of a minimum size in 1968 and 1969.

There are important environmental differences between the northern (north of the Olifants River) and southern parts of the west coast rock lobster fishery. These differences are mainly due to the presence of oxygen-deficient bottom water, which at times extends close inshore, in the more northerly regions (Pollock & Shannon 1987, Grobler & Noli-Peard 1997). This results in a tendency for *Jasus lalandii* to increasingly be confined to shallower waters as one moves northward (Pollock & Beyers 1981). Pollock & Shannon (1987) have hypothesised that the lower production of the northern regions of the fishery between 1960 and 1970 was the result of environmental change over this period, resulting in a progressive expansion of oxygen-deficient shelf water. They suggest that this forced the animals into shallower water, where reduced habitat and consequent competition for food resulted in lower growth rates and higher natural mortality, which flowed through to the fishery as reduced surplus production.

Traps were introduced in the South African and Namibian fisheries in the 1960s. The introduction of this new technology affected the fishery in a number of ways. Because traps were hauled with mechanical winches, they opened up deepwater fishing grounds unavailable to hoop net fishing.

Also, because trap fishing gear is unattended, it permitted fishing at night and in weather conditions unsuitable for operating dinghies. Finally, the large volumes of lobsters taken in each trap made sorting the legal- from undersize catch difficult and probably resulted in large-scale discard mortality. Thus it would seem likely that this technology helped to maintain landings at unsustainable levels and led to a false impression of the availability of lobsters in ensuing years.

After the declines in landings during the late 1960s and to a lesser extent the 1970s, the 1980s were a period of relative stability in both the South African and Namibian fisheries, with seasonal landings of around 3500–4000 and 1000–1900 t, respectively. Both countries experienced a major decline in CPUE and landings in the 1989–90 and subsequent seasons. These changes have been direct and indirect results of a reduction in the annual growth increment (Melville-Smith et al. 1995, Goosen & Cockcroft 1995, Cockcroft 1997, Pollock et al. 1997). The 1990s have also been a period in which there have been more mass strandings of lobsters caused by hypoxic conditions associated with high biomass dinoflagellate blooms (Cockcroft 2001) than in any previous decade. Of particular concern has been the loss in the 1990s of 5–12% of the *Jasus lalandii* spawner biomass as a result of mass stranding mortalities (Cockcroft 2001).

There is no question that the size structures of the west coast rock lobster populations in South Africa and Namibia today bear no resemblance to those of the pristine stock. It is clear from the minimum sizes that were in place (Figure 23) and the quantities of lobsters that were being landed each season (Figure 22) that either there must have been a very large initial biomass of animals (>89 mm CL), which was depleted over time, or productivity in earlier years must have been considerably higher than in the modern era. Historical size composition data are scarce, but where available (e.g., Gilchrist 1913, 1914b, Anonymous 1935, 1939) they confirm that animals of the modal size classes of prewar years were of a size seldom ever recorded in modern catches.

Pollock et al. (2000) gives the harvestable biomass (i.e., biomass of animals of >75 mm CL) for the South African fishery as ~5% of pre-exploitation levels, and the spawning biomass as ~20% of pristine levels. These results, backed by model results, suggest that the depressed state of the spawning biomass may have resulted in a decline in recruitment in recent decades, and in particular since the mid-1980s, when the biomass underwent substantial change. With this fact in mind, a stock rebuilding strategy has been developed for the South African fishery in recent years (Pollock et al. 2000). The Namibian fishery is being managed by setting conservative TACs and has the stated long-term objective of rebuilding stock levels to those of the 1980s (Grobler & Noli-Peard 1997).

Wider ecological effects of Jasus lalandii *exploitation*

West coast rock lobsters are known to feed on a wide range of food items, including molluscs, polychaetes, fishes, sea urchins, other crustaceans (including their own species), algae and sponges. They thus play an important role in influencing the structure of shallow subtidal benthos on hard substrata in some (Branch & Griffiths 1988), but not all, rock lobster grounds (Tomalin 1993). The depleted stocks, and in particular removal of large size classes, which are able to consume prey that is not available to small lobsters (Pollock 1979, Griffiths & Seiderer 1980), have presumably had ecological consequences to benthic communities on the lobster grounds. However, the extent and implications of these changes remain largely unknown.

In contrast to the depleted state of the rock lobster stock on the west coast, there has been a substantial increase in the abundance of west coast rock lobsters on the South African southeast coast since the late 1980s (Tarr et al. 1996, Mayfield & Branch 2000). These increased densities of lobster, which lie outside of commercial fishing grounds, are believed to have depleted sea urchin populations and, in so doing, have disturbed the ecological balance between sea urchins and juvenile abalone, thereby contributing to the decline in the abalone fishery (see next section).

In summary, there is little doubt that whereas the primary reason for the present state of both the Namibian and South African rock lobster fisheries is overexploitation, there have been multiple factors contributing to this situation. While most of these factors have an obvious link to human impacts, for example, advancements in gear technology or illegal catches by the nonindustrial sector, there are others, such as the hypotheses of large-scale environmental change and the incompletely explained reasons for changes in lobster growth rates, for which there may be little or no relationship with human activity.

Abalone

Abalone, *Haliotis midae*, known locally as perlemoen, have been utilised by coastal fishers for at least 125,000 yr, as indicated by Middle Stone Age cave deposits and until recently, the species remained readily accessible to shore pickers in the intertidal area. There are a number of other, smaller abalone species in southern Africa, but only *Haliotis midae* has been the target of commercial fishing.

The present-day fishery covers the coastline between Quoin Point and Cape Columbine, with the most productive region being from Cape Hangklip to Quoin Point. Fishing is carried out by licenced commercial divers who operate from twin-engined ski boats launched from the shore. Except for the introduction of satellite navigation and more seaworthy fishing boats, the method of fishing is essentially the same as it has been since the inception of the commercial fishery in the 1950s. A "hookah" system is used, consisting of an onboard compressor that supplies air to the diver through a hose. The minimum size limit of 114-mm shell breadth has been unchanged since 1955. Being collected by hand, the harvest method has no secondary impact on the kelp forest ecosystem.

Records of commercial catches are available from 1953 (Figure 24) and were calculated from production figures (amount marketed, canned, or frozen annually). Initially, catches were around 500 t yr^{-1} and fluctuated due to varying market demand. However, after 1960, catches escalated dramatically and, in the absence of any limiting quota, reached a peak of around 2800 t in 1965. During this period fishing was essentially a mining operation, removing pristine accumulated beds of abalone, where up to 20,000 abalone were taken from a single bed (Newman 1965). These aggregations occurred at densities of 15–20 abalone m^{-2}, permitting individual catches as high as 3000 abalone day^{-1}. As these accumulations were mined out, catches, as well as catch rates, declined annually from 1965–70, and although production quotas were imposed for the first time in 1968 and 1969, they were not filled. Only in 1970 did the production quota first limit catches. Thereafter, catches have been quota limited, and the stabilising effect of this measure is apparent in Figure 24.

From 1983 the quota system was changed to one based on the whole mass of abalone delivered to the factories, because investigations showed that the production quota was being misused, in that abalone were being cut up into pieces and not declared as quota. It is likely that the quota was exceeded by at least 15% during the 1970s and up to 1982. Area-bound, annually revised TACs were introduced in 1985. From 1992–3 catches were sealed at the slipway, before transport to the factories, as there were allegations that abalone were being landed but not delivered to the factories. In a further refinement from 2000 the weight of the catch was recorded using mobile scales, directly at the landing sites.

Participation in the fishery has changed over the years, initially being free access to all, with around 100 divers active by the early 1960s. These divers delivered on a catch-as-catch-can basis to the four or five licenced abalone processing factories. These factories were allocated a fixed percentage of the quota from 1968. In an effort to limit the fishery to bona fide fishers, diver numbers were reduced gradually to around 50 by 1989. From 1984 the divers became legally obligated to deliver to specific factories and were also each allocated a fixed percentage of the quota, the value of which was derived from past performance. This placed divers in a position of greater financial security, giving them a stake in the fishery that was saleable and heritable. From 1998–99, the balance of power, which had previously rested with the factories, which "owned" the quota, was reallocated when divers and quota holders were amalgamated into one group called

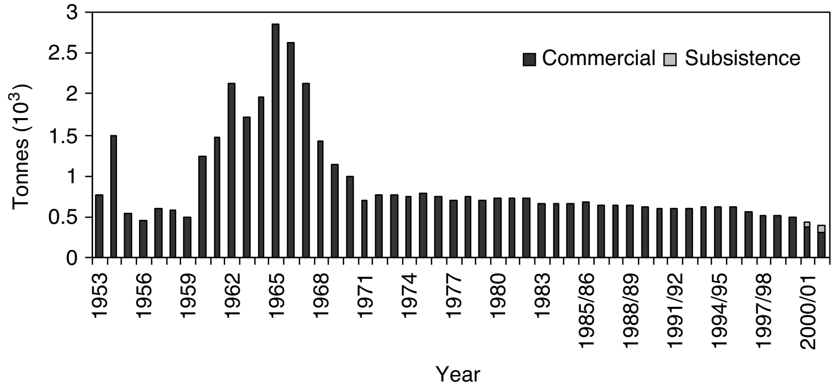

Figure 24 Commercial catches of abalone from 1953 showing the stabilising effect of quotas from 1970 and the declines evident from 1996.

right holders. These right holders could harvest or market their abalone on an equal basis, subject to certain legal constraints.

A new subsistence fisher category was created in 1997 and continued for 3 yr. This sector, however, proved virtually unmanageable due to the shore-based snorkel fishing method permitted, and the associated lack of controls, with a consequence that the amount of abalone landed for the first two seasons is unknown. In the third year, the fishing method was changed to a boat-based operation with individual allocations of 300 kg, enabling better control to be exercised over the landings and the catch for the season to be accurately recorded. This sector was then converted to a small-scale type of fishery called Limited Commercial, from the 2001–2 season, which follows normal commercial fishing practices.

Recreational fishing for abalone is also an important component of the fishery dynamics. The number of recreational fishers is not limited, but they must purchase an annual permit that limits the daily bag limit (presently) to three abalone, and they may operate from the shore only, using snorkel equipment. Monitoring of recreational fishing catches started in the late 1980s, with catches increasing to equal 89% of the commercial catch in 1993–94. A series of management measures were imposed over the years to reduce recreational catch. The most effective of these was a reduction of the length of the fishing season, from the original 270 days to only 11 days (specified weekend days) for the 2001–2 season. Nonetheless, this only resulted in a reduction of recreational take to 28% of the combined commercial sectors for 2001–2, due to greater recreational fishing effort being applied during the very limited season.

Status of the fishery

The abalone resource is presently facing a severe crisis. This is a result of two developments, poaching and ecological change, both of which coincidentally affected the resource from around 1994.

Poaching has always been a factor; however, since around 1994 it has become increasingly important, to the extent that recent data indicate that the fishery is unlikely to remain sustainable unless major improvements in compliance occur. Confiscation records obtained from the various branches of compliance services have been collected since 1994, and these reflect a great increase in activity over the last 3 yr (Figure 25), a trend that is confirmed by anecdotal information from a variety of sources. Some of the recent increases in confiscations may be attributable to increased effectiveness of compliance staff, but this increased efficiency is not believed to be equivalent to more than a factor of approximately 25–30%. This is because as compliance efficiency improves,

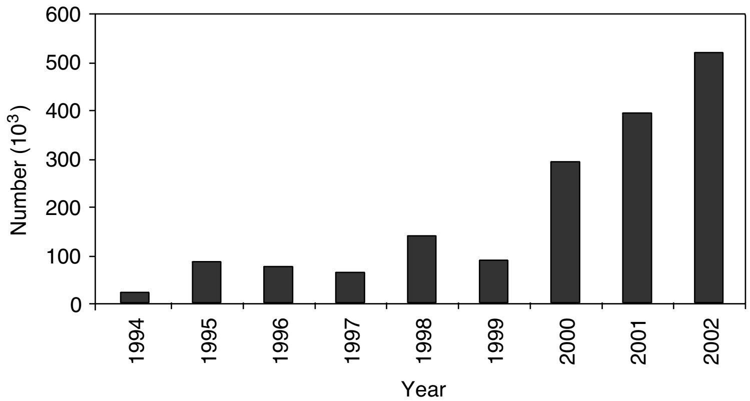

Figure 25 Records of numbers of abalone confiscated from 1994. Data for 2002 represent confiscations for the first 4.5 months only.

so do the evasive skills of the highly organised gangs involved. However, there is no doubt that the damage caused to the resource by poaching is serious. The volume of abalone removed illegally must be considerable, with most estimates assuming that confiscations represent only around 10% of actual illegal catch. Around 55% of the abalone confiscated are below minimum legal size. Given that maturity is reached about 1 yr before attaining the fishery size limit, poaching is removing large numbers of abalone from the population before they have an opportunity to reproduce.

In addition to the problems caused by overfishing, a dramatic environmental change has recently occurred in the centre of the historically most productive region of the abalone population, Cape Hangklip to Hermanus, and this has severely curtailed the reproductive effectiveness of the population. The change became noticeable from around 1994, when a large-scale incursion of rock lobsters *Jasus lalandii* into the area first became apparent. The rock lobsters, which previously occurred in low densities in this area, have consumed the majority of small invertebrates in this area, such as sea urchins *Parechinus angulosus* and gastropod molluscs. The disappearance of the sea urchin population in particular has negatively affected the reproductive efficiency of the abalone, since juvenile abalone derive important shelter from predators under the spine canopy of the sea urchins. In the absence of urchins, abalone are very vulnerable to predation, including that from lobsters. As a result, recruitment of abalone in this area is now estimated to be only 10% or less of normal. For this reason, any significant removal of abalone from the area between Cape Hangklip and Hermanus must now be seen as an essentially unsustainable practice.

The natural tendency of pristine *Haliotis midae* populations is to develop into high-density aggregations of adults in the shallow inshore zone. This adaptation facilitates reproductive efficiency in these broadcast spawners. Furthermore, abalone have developed an efficient drift-weed trapping feeding behaviour, which is essential for effective herbivory in such high-density, essentially nonmobile populations. Fishing, poaching, and ecosystem changes have resulted in severe declines in density. Extensive diving surveys in the 1980s showed that those areas that supported large populations of abalone demonstrated average densities in the shallow inshore area of around 0.8–1.3 abalone m^{-2}. These densities are similar to the those recorded in fishery-independent abalone surveys (FIAS) of the Betty's Bay marine reserve, initiated in 1995, which averaged around 1.2 abalone m^{-2} from 1995–98. In contrast, as a result primarily of poaching, the average density in three of the four main abalone commercial grounds has now declined to below 0.3 abalone m^{-2}.

It is not known whether these declines in abundance have reached the point where nearest-neighbour distances are such that fertilisation efficiency of the population is negatively affected. Recruitment studies carried out in the late 1990s indicated that abundance of juvenile abalone less than 2 yr old was not noticeably different in commercially fished grounds compared to refugia. However, at present, in certain areas that have been heavily denuded through poaching, it is likely that recruitment success has been compromised, with further negative implications for this already threatened stock.

Freshwater inflows and estuaries

Rivers and their associated estuarine systems impact on the marine environment of the Benguela in a number of important ways, most notably as nursery areas for marine fishes, as habitat for wading birds, and as sources of sediment and nutrients and other materials (including pollutants) entering coastal seas. As an arid to hyperarid region the Benguela coastline supports very few estuaries. Indeed, no perennial rivers reach the sea within the entire Namibian coastline, although the Cunene and Orange (Gariep) Rivers form its northern and southern borders, respectively. The only major river systems within the entire Benguela region are thus the Orange, Olifants, and Berg Rivers. All have been highly manipulated and, indeed, the attitude of 20th-century South African engineers to rivers may be summed up by the following quotation:

> There can be hardly a single true South African, and certainly no irrigation engineer, with soul so dead that he can contemplate our greatest river tearing down to the ocean through a vast area of country, which is thirsting for water, without feeling that some great effort should be made to design and carry out irrigation works for the Orange River, which would rival those famous works of other great rivers of the world.
>
> — Dr. A.D. Lewis, Director of Irrigation, October 12, 1928, Graaf-Reinet

This anthropocentric view of rivers, that water running down them to the sea is wasted, has led to overmanipulation for water supply of all west- and southwest-flowing rivers of the subcontinent. As a result, rivers that historically contributed to estuarine and inshore coastal processes, especially the Orange-Vaal, the Cape Olifants, the Great Berg, the Palmiet and the Breede (Figure 1), either no longer flow or flow at greatly reduced volumes and in a greatly regulated manner, due to overabstraction and the construction of water supply reservoirs and interbasin water transfer schemes (IBTs). The situation has become so serious that a number of recent Global International Water Assessment (GIWA) workshops have identified human manipulation of water resources and overutilisation as the single most important threat to the ecological functioning of the region (Prochazka et al. 2001, Davies 2002).

This part of the review focuses on two of these catchments, the Orange-Vaal and the Great Berg. However, these systems epitomise the severity of anthropogenic flow modification wrought on other systems throughout the region.

The Orange-Vaal system

The Orange-Vaal is the largest watershed in southern Africa south of the Zambezi (17˚S), draining approximately 47% of South Africa (South African Department of Water Affairs (DWA) 1986), providing between 5.5×10^9 m^3 (Keulder 1979) and 11.9×10^9 m^3 of water yr^{-1} (Cambray et al. 1986) and contributing 22.1% of the total mean annual runoff (MAR) from South Africa (Noble and Hemens 1978).

Twenty-three major dams have been constructed within the Orange-Vaal catchment since 1872 (Benade 1988), and it contains nine of South Africa's 30 largest dams. Nine of the 23 major dams were completed in the 1970s, while only one, Katse (187 m), has been constructed since 1978. Together with Mohale Dam (under construction), this dam forms Phase 1A of the massive Lesotho

Highlands Water Project (LHWP), an IBT transferring water from the headwaters of the Orange in Lesotho to the Vaal for consumption in Gauteng (Davies & Day 1998).

The flow regulation of the Orange River can be traced back to 1877, the start of irrigation schemes in the Great Fish and Sundays Valleys (Eastern Cape), as well as those in the middle Orange River. Irrigation in the middle Orange seems to have begun c. 1883, although according to Alexander (1974), surveying for Boegoeberg Dam had already started in 1872. Boegoeberg was eventually built in 1931, but in the intervening period over a dozen private weirs had already been built in the main river channel.

With the advent of the Union of South Africa in 1910, non-conservation orientated agricultural activities in the Great Fish and Sundays Valleys led to large-scale soil erosion, flash flooding, and siltation, and attention turned toward the diversion of water from the Orange River to the Eastern Cape (the Orange River Development Project (ORDP); Alexander 1974). With the announcement of the ORDP in the early 1960s, many scientists remarked that it was inappropriate to develop dams on the Orange without understanding the basic biological processes in the system. As the ORDP continued, calls for the urgent prioritisation of baseline ecological data were made, notably by Chutter (1973). Similar calls were made in 1988, with the advent of the LHWP (Petitjean and Davies 1988), but both calls were ignored until the late 1990s. Rapid population increase in Gauteng, as well as the development of the Orange Free State gold fields and the Vaalhartz Irrigation Scheme, negatively affected the Vaal and thus also the middle and lower Orange, between the late 1940s and early 1950s (Alexander 1974). With emphasis switching from irrigation supply to industrial and domestic potable use, the single-purpose diversion of Orange River water to the Great Fish and Sundays systems grew into a multipurpose scheme, comprising three reservoirs, the Gariep, van der Kloof, and Torquay dams; hydroelectric power stations; irrigation systems; and the 83-km-long Orange-Fish Tunnel IBT (Benade 1988).

Presently the Vaal catchment contains 16 of the 23 major storages of the Orange-Vaal Basin. Their existence has led to an extreme cumulative case of overabstraction and river regulation (Figure 26 and Figure 27). Indeed, the Orange-Vaal system is probably the most regulated river system in Africa (e.g., Cambray et al. 1986) and one of the top three overregulated systems in the world, ranking with the Colorado (U.S.) and the Murray-Darling (Australia).

The Orange per se had a more reliable preregulation runoff than the Vaal, but with pronounced seasonal variation and widely fluctuating interannual variation; erratic high flows transported silt and adsorbed nutrients to the coastal zone (Cambray et al. 1986; Figure 26 and Figure 27). For instance, postriver regulation, Bremner et al. (1990) reported that the floods of 1998 delivered 24.3 km^3 of water and 64.2×10^6 t of suspended sediments to the sea through the mouth at Alexander Bay. The same paper noted the source of material as bed scour and bank erosion below the major dams, as well as erosion of the estuary mouth. The resulting sediment transport led to the formation of a delta extending 1.2 km offshore with a mass of approximately 3.6 million t. Interestingly, prior to flow regulation by dams, sediments of the Orange were derived from the Middle Orange catchment.

Interannual flow variation has also been exceptionally large. The maximum Orange River flood probably reached 31,200 $m^3\ s^{-1}$, although the largest recorded flood occurred at Hopetown in 1874 (11,330 $m^3\ s^{-1}$). Palmer (1996) reports that “prior to the building of dams in the [1970s], the [Orange] river often ceased flowing in winter [1862–3, 1903, 1912, 1933, and 1949] … and was reduced to isolated pools.” At the coast, the estuary received aperiodic, large discharges, resulting in limited tidal exchange and irregular scouring, such that permanent sediment deposition was reduced. The mouth remained fresh for several months every year and according to Brown (1959), experienced extreme seasonal flooding, high silt loads and turbidity. In this context, Branch et al. (1990) report that the postregulation floods of 1988 led to abnormal dilution of coastal waters, with major effects on intertidal and shallow subtidal organisms. Mass mortalities of patellid limpets were recorded along the shore within 10 km of the mouth. The loss of these grazers led to dense growths of *Ulva*

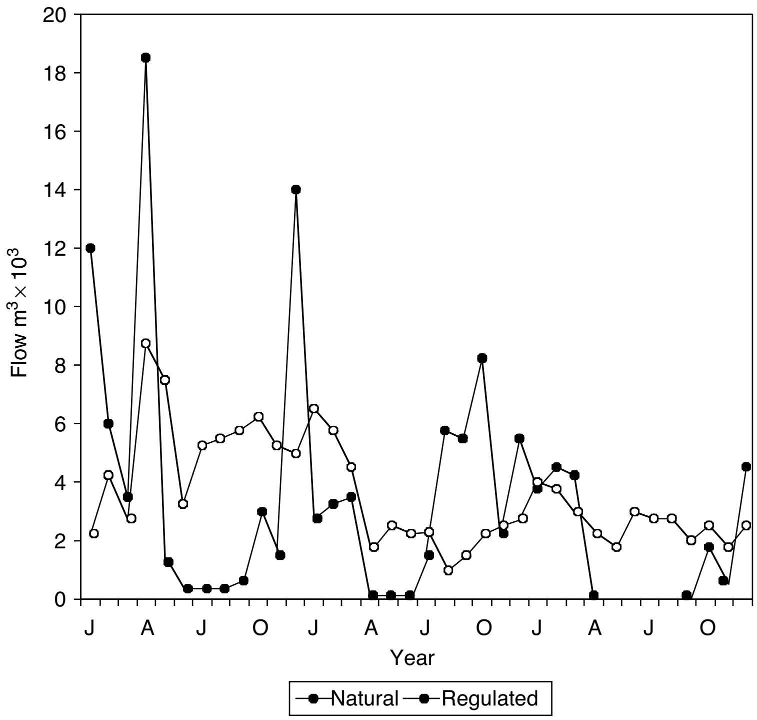

Figure 26 Natural and regulated flow regimes of the Orange/Gariep River, 1978–1980.

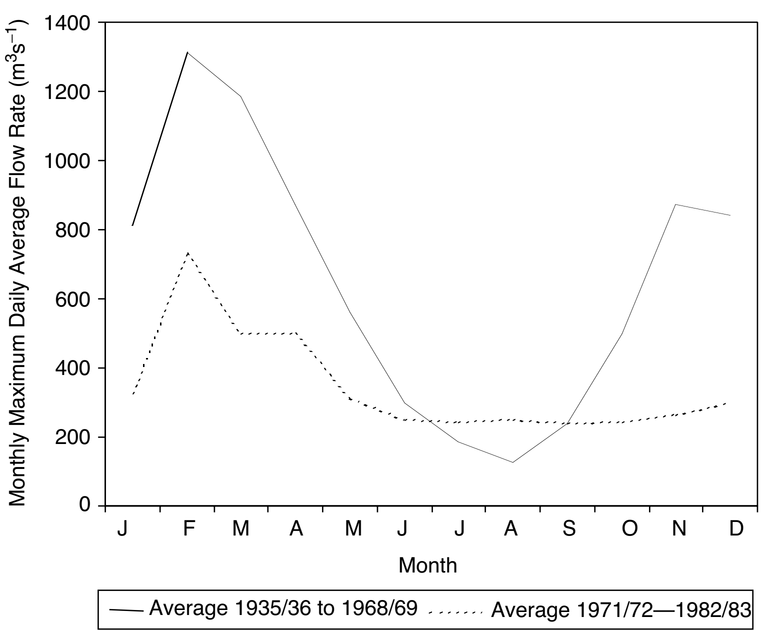

Figure 27 Seasonal variations in the monthly runoff at the Orange/Gariep River mouth under natural flow conditions (solid line, average flow for 1935–69) and after construction of the Gariep and van der Kloof Dams (dotted line, average for 1971–83). (After Cambray et al. 1986.)

and *Enteromorpha* in mid- to low-shore zones and *Porphyra* along the high shore. Clearly similar temporary shifts must have occurred along the intertidal and subtidal zones during preregulation high-flow events discharged by the historically unmodified Orange.

Lake Gariep, completed in 1970, is the main regulator, providing flood control, hydroelectric power and acting as a silt trap (Kriel 1972, 1978). The modifications of flow (Figure 26 and Figure 27) comprise changes in the (1) annual runoff, (2) interannual variation in runoff, and (3) the marked seasonality of the preregulation flow regime. Most floods from the catchment above Gariep are contained by Gariep and van der Kloof, completed in 1978, cutting the maximum flood (31,000 $m^3 s^{-1}$) by 65% (Kriel 1972). The lower van der Kloof Reservoir supplies a stable, continuous water supply for irrigation farmers and the regime is in direct conflict with the variable demand required by the aquatic environment in as much as estuarine, mouth, and inshore coastal characteristics are affected. Suspensoid loads in the Orange River are very high and constitute 0.46% of the flow volume. In perspective, a flood peak of 8500 $m^3 s^{-1}$ in 1967 was estimated to carry some 250,000 t ha^{-1} of silt (South African Department of Information 1971), while the average annual inflow of silt to Lake Gariep is c. 32×10^6 $m^3 yr^{-1}$ (Kriel 1972). This silt is now mainly trapped by Gariep.

The total natural, or virgin, annual river flow at the mouth is estimated to have been some 10.670×10^9 m^3 (Prins 1990). In 1991 the flow was measured at 5.340×10^9 m^3. This reduction may have serious implications for the rich avifauna (e.g., Frost & Johnson 1977, Siegfried & Johnson 1977), where sandbars provide ideal roosting and nesting sites for pelicans, cormorants, gulls, and terns. Courtenay-Latimer (1963) listed 172 species from Holgat to the mouth; however, the influence of the regulated river on the avifauna is complex. For instance, previous floods would have destroyed nests and inundated roosting sites (e.g., Morant 1990, reporting on the floods of 1988), while reduced seasonal flooding and altered drought conditions may make the mouth more stable for aquatic birds. On the other hand, permanent loss of silt and nutrients can only be detrimental to the system and certainly reduced runoff contributes to salinity problems due to an effective increase in concentration of available salt in the smaller volumes of water.

As a direct result of the South African National Water Act (no. 34 of 1998), environmental flows or in-stream flow requirements (IFRs) from dams are mandated, but although an IFR for the Orange River has been calculated (Venter & van Veelen 1996), it has yet to be implemented and little if any cognisance has been taken of coastal and inshore marine requirements, particularly sediment and nutrient discharges through the estuary.

The Great Berg River

The Great Berg (Berg) River rises on the northern slopes of the Franschoek Mountains in the Western Cape near Cape Town. Flowing north and west, it forms a complex floodplain and estuarine system before entering the Atlantic at St. Helena Bay. Like the Orange, it is allogenic, flowing through progressively more arid areas toward the sea where the MAP is <360 mm yr^{-1} (Ninham Shand Inc. 1992). Extensive viticulture, grain and stock agriculture are practiced, with water demand reaching a peak in the low-flow summer months. Historically the flow of the Berg (1928–88; Berg 1993) has been reduced such that flood-flow return periods of >75% are now the norm. The virgin MAR to the mouth was 903×10^6 $m^3 yr^{-1}$, while present-day annual runoff is 23–30% lower at 693×10^6 $m^3 yr^{-1}$ (Berg 1993; Figure 28).

Irrigation demands and the growing Cape Town metropolitan region have led to construction of a number of dams and IBTs. Only 1.6% of Cape Town's water supply is generated within its boundaries. Present water developments include the Riviersondereind (a tributary of the Breede)-Berg River (RSB) IBT (Snaddon & Davies 1998, 1999, Snaddon et al. 1998, 1999). Irrigation releases increase upper river flows in summer by between 560 and 4000% and deliver 15×10^6 m^3 yr^{-1}. This, coupled to overabstraction within the Breede catchment itself, has led to significant reductions in flow to the estuary of the Breede, such that changes in the flood tide delta since 1942 have annihilated the once dense *Zostera* beds at the mouth (de Villiers 1988).

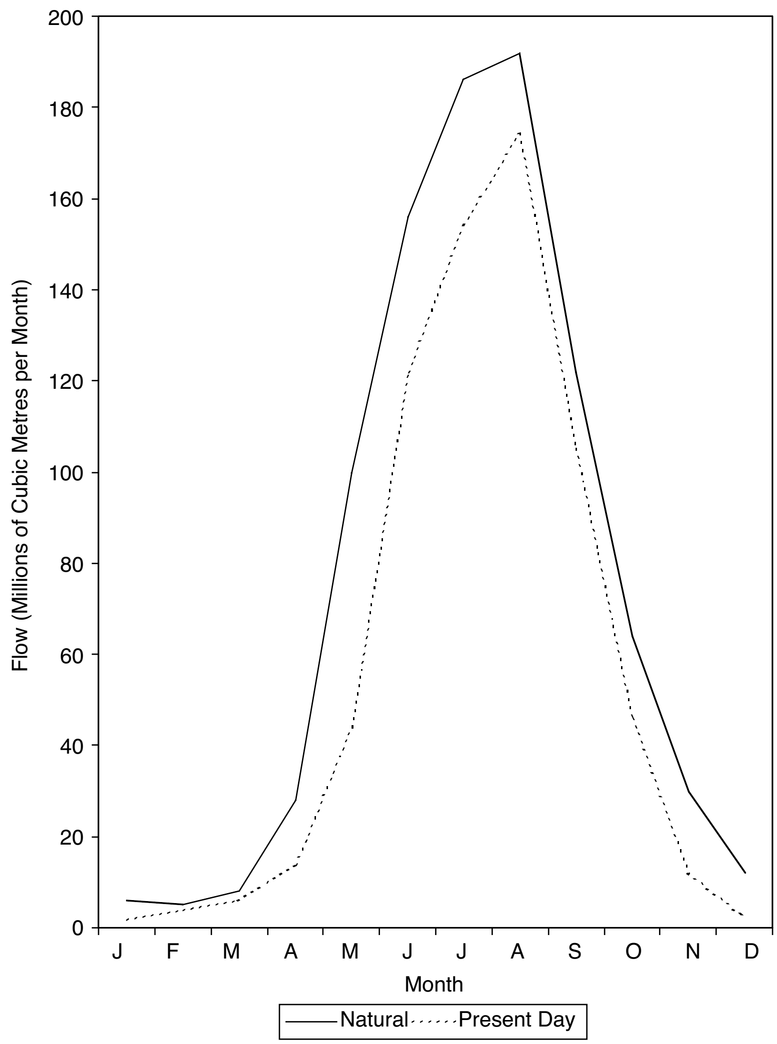

Figure 28 Virgin (natural) and regulated (present day, 1993) seasonal distribution of runoff to the Berg River estuary. (After Berg 1993.)

The uncontrolled growth of Cape Town and the development of industry at Saldanha near Langebaan Lagoon (recognised by the Ramsar Convention as a wetland of international importance) have led to the development of the controversial Skuifraam Dam in the upper Berg catchment near Franschoek (Cape Town supply). Skuifraam will harvest all upper river flows and all small to medium flood flows in the system (Berg 1993) and will decrease peak discharges in upper Berg by 40%, and in the lower system by 20%, above present abstraction rates (Basson and Beck 2001). Multiple flood flows in any year cannot be maintained for Skuifraam; return flows of 1:2 yr will be reduced to 1:3 yr (simulation data). With the construction of Skuifraam, flows in the Berg will drop from 914×10^6 m^3 MAR to 572×10^6 m^3 MAR (62% of virgin). Once additional water supply developments are executed, flows will fall further to 272×10^6 m^3 MAR (25% of virgin) (Little 1993), with severe implications for the continued functioning of the estuary, the floodplain, and the adjacent coastal zone.

To place the Berg River estuary and floodplain in context, O'Keeffe et al. (1992) note that 76% of the estuaries of the Western Cape are degraded. Of these, the Berg is undoubtedly the most important. The once mobile estuary mouth was artificially entrained in 1966, changing tidal amplitude and causing higher seawater intrusion to the floodplain (Huizinga et al. 1993). Upstream impoundments now prolong periods of low flow in rivers, increasing salinity intrusion and reducing sediment dynamics in the estuary and at the mouth. The development of Skuifraam will exacerbate these features, reducing flushing and scouring, and may ultimately lead to mouth closure (Huizinga et al. 1993), with severe consequences for inshore fisheries and coastal processes.

The nearly 6000-ha floodplain–estuarine complex supports 10 different vegetation communities, making it the most structurally complex system of its type in southern Africa (McDowell 1993). Among other features, the floodplain–estuary has the third largest area of salt marsh along the Cape coast, second only to Langebaan Lagoon, within the Benguela region. Mouth entrainment and flow regulation have significantly impacted the vegetation (McDowell 1993).

Since 1975, 127 bird species have been recorded utilising 5 of the 10 wetland communities (Hockey 1993a), and despite the fact that the system is not a Ramsar site, in terms of its avifauna it is far more significant than the adjacent Langebaan Lagoon, the Orange River Mouth, and even the St. Lucia Wetlands of KwaZulu–Natal (all of which are Ramsar sites). The system ranks second in importance only to Walvis Bay in Namibia (also a Ramsar site). In 1992, 46,000 individuals recorded represented regionally significant populations of 31 bird species and nationally significant populations of 25 species, with five Red Data species present (Hockey 1993a). In order to maintain these populations Hockey (1993a) recommended a minimum average flooding frequency of 1.9 yr^{-1}. Given the development of Skuifraam, this is unlikely to happen. Further, reduced flows over the past 40 yr, coupled to desiccation, increased salinities, and possibly pollution, have had severe impacts on the density of benthic invertebrates, thereby adding to concerns for the maintenance of the avifauna (Hockey 1993b).

The system is also vital in terms of its role as a coastal zone fish nursery. Bennett (1993) has remarked that the fishes recorded from the Berg River estuary–floodplain system represent 77% of the total coastal species, compared with between 49 and 52% for other estuaries. Indeed, the Berg has higher percentages of resident species (23% vs. between 4 and 18% for other estuaries in South Africa), dependent species (27% vs. between 9 and 25%), and partially dependent species (30% vs. between 18 and 27%) (Bennett 1993). Accordingly, flow reductions in the Berg have had far more impact on coastal fisheries than anywhere else in South Africa. Additionally, the estuary is one of only two permanently open estuarine nursery areas in the region. It is thus clear that further flow modification of the Berg will have serious implications for the maintenance of inshore fisheries. In this context, Schrauwen (1993) notes that although a local sport fishery is developing, commercial fisheries are already overexploited and that Berg River estuary salinity increases (reduced flows) render parts of the industry nonviable.

An IFR has been calculated for the Berg, but again, as in the Orange, it has not been implemented. Thus, like all other river systems flowing to the Benguela, the natural river ecosystem is at risk, with disturbing future implications for estuarine and coastal processes in the region.

Mariculture

Although certain forms of fish and shellfish farming have been practised in South Africa for many years, it is only since the 1980s that aquaculture has grown into a commercialised and economically viable industry. Freshwater aquaculture in South Africa is, however, severely limited by water availability, and the only real potential for the development of the industry in South Africa therefore lies in the sea (Cook 2000).

In the Benguela region, marine aquaculture is at present practiced mainly in Saldanha Bay, with smaller operations at various other sites. Mussels and oysters have traditionally formed the bulk of production, but seaweeds and abalone have become more important in the past few years. Although prawns are produced in brackish water ponds in KwaZulu–Natal on the east coast, they are not cultured in the Benguela region because of low water temperatures. Over the past 10 yr, total mariculture production in South Africa has remained relatively static, but some important changes have occurred with respect to individual species, the production of some species having been discontinued, while others have seen significant production increases. Total production in the Benguela region in metric tons per year (animals only) is shown in Figure 29. Some of these values were obtained directly from South African farms, while others, particularly for earlier years, were obtained from published Food and Agriculture Organisation (FAO) statistics (Grainger & Garcia 1996). Further details with respect to each of the major species listed are provided below.

Mussels

Mussel production in South Africa uses either the raft or longline system, the former proving to be the more popular. Although between 1988 and 1992 attempts were made to farm some indigenous species (e.g., *Choromytilus meridionalis* and *Perna perna*), the industry has now settled on the introduced Mediterranean mussel *Mytilus galloprovincialis* as the most economically viable species. In Saldanha Bay, the main centre of the industry, spat are collected locally on ropes. Since the first farmed mussels were produced in the early 1980s, annual production rose rapidly, reaching 2300 t in 1994. After a slight decrease in production in 1995 and 1996, production again rose in 1997 as a new small-farmer cooperative began to come onstream. Production reached a peak of 2600 t in 1998, with a total value of a little over U.S.$2.5 million. Problems associated with harmful algal blooms (HABs) in the Benguela region have severely curtailed mussel production over the past few years, and in 2000 production fell to the very low value of 160 t. It appears, however, that production is now beginning to increase again.

Oysters

As in many other countries, oyster production in South Africa and Namibia utilises the fast-growing Pacific oyster, *Crassostrea gigas*. Although an exotic species in South Africa, importation of spat is permitted because it has never been thought to show any signs of becoming invasive (although there are recent reports of wild populations in several Southern Cape estuaries). At present, the vast majority of production is from imported spat, but recently a small local hatchery started operating. Several farming methods are used and each has been developed in response to the

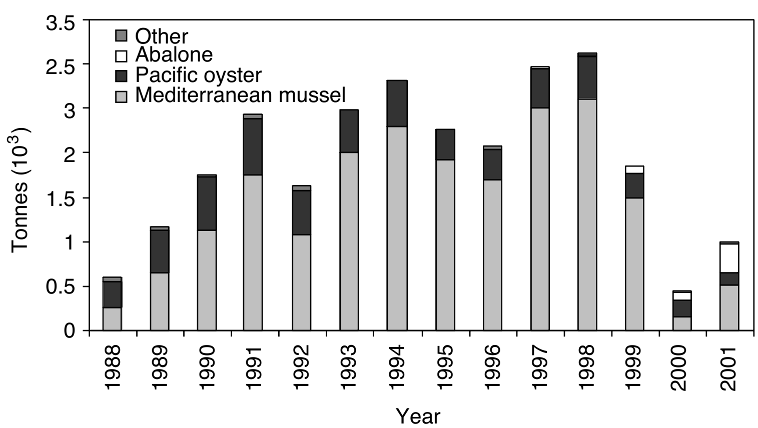

Figure 29 Mariculture production in the Benguela region from 1988–2001.

demands of local conditions. The most common method uses racks or oyster baskets in shallow coastal lagoons. Total annual production in South Africa reached over 600 t in the early 1990s but has since dropped considerably. In 1998, about 480 t was produced, the total value being about U.S.$2.26 million. By 2001, production had dropped to 147 t.

Seaweeds

Research into seaweed farming has been conducted over the past few years, concentrating mainly on *Gracilaria* and *Ulva*. *Gracilaria* is used mainly for agar extraction, whereas both *Gracilaria* and *Ulva* are used as important food items in the emerging abalone farming industry. Most of the farming effort in South Africa has been concentrated in the vicinity of Saldanha Bay, where a thriving *Gracilaria* harvesting industry existed before harbour construction altered the natural habitat of the seaweed and rendered the harvesting industry uneconomic. Both tank cultivation and suspended cultivation from rafts are being used, but at this stage, the raft method seems to be preferred for commercial production. Raft design is based on that already in use in Namibia, where *Gracilaria* farming is already an established industry, incorporating both actual culture and some limited factory processing where semirefined agar is produced. Production figures have fluctuated over the past few years as new producers have experimented with *Gracilaria* production, only a few of whom have survived to form fully commercial production units. A significant increase in 1998 occurred as abalone farmers began producing macroalgae as an abalone feed. Estimated production in South Africa in 1998 was 25 t, the total value of which was a little over U.S.$108,000. It is predicted that production in 2002 may reach 105 t.

Abalone

Perhaps the greatest mariculture potential in South Africa is for abalone farming. The local species, *Haliotis midae*, is prized among markets in the East and the market price has encouraged local companies to invest in this industry. All farms are land-based and water is pumped ashore to them. The coastline supports huge kelp beds and most farms use kelp as food, although an artificial food is being produced locally, which may replace or supplement kelp on some farms. About 10 farms are now under construction, with the most advanced almost at the full commercial production phase.

Farmed abalone from South Africa began to appear on the international market in small quantities in about 1993, but at that stage, less than 1 t yr^{-1} was being produced. By 1997, annual export production was about 10 t, and since then, production has gradually increased to about 317 t in 2001. It is predicted that this production level will continue to increase over the next few years.

Although prices have fluctuated recently, the 1998 production was valued at about U.S.$950,000. Because abalone is sold as an international commodity and is priced in U.S. dollars, the value of the industry has increased significantly over the past year or so, and it is estimated that export production is now valued at least at 95 million rands. All South African farmed abalone is exported, mainly to Japan, as it is illegal to sell farmed South African abalone on the local market. Although there is probably potential in South Africa for that production level to be increased, market demand will obviously dictate the eventual size of the industry.

Marine finfishes

The farming of marine finfishes is a relatively new activity in the Benguela region, with turbot (*Psetta maxima*) being the only species currently produced commercially. Although there are several indigenous species of marine fishes that appear to have great mariculture potential, the technology for farming them is only just being developed. It seems likely, however, that this is an area of mariculture where significant expansion can be expected in the future. Several regions of South

Africa seem to have potential for salmon or turbot farming, and several applications to start farms are being considered at present. One of the main problems is that since finfishes are exotic species to South Africa, the potential hazards of introducing them are not yet known and conservation authorities have been expressing severe reservations about possible problems. In addition, it is doubtful whether there is a sufficiently large market for these fishes in South Africa and export potential thus needs careful investigation. However, it appears to be only a matter of time before the first salmon farms will be commercialised.

Production losses

In recent years, there have been production losses in most of the major forms of mariculture. In mussel farming, the major loss was caused by adverse weather conditions leading to raft and longline destruction. In some cases, HABs have interrupted mussel harvesting, but this cannot really be classified as a loss, because mussels were again harvested after the bloom had ended. In the case of oysters, however, major permanent production losses have been caused by HABs, particularly associated with the organism *Aureococcus anophagefferens*. This organism began to affect oyster production toward the end of 1998, and its effect was severe in 1999. HABs have also affected abalone production, but only to a very small extent.

Historical and future production trends

Two trends appear in the historical mariculture production figures for the Benguela region (Figure 29). First, a general stabilisation of the industry has occurred where a number of species that were cultured on an experimental or pilot plant basis (e.g., red bait, *Mactra*, and mullet) have now been discontinued and replaced by species that are more economically viable. For example, a few years ago, three species of mussels were cultivated, but now all mussel farms have settled on the one species, *Mytilus galloprovincialis*, which produces the best return on investment.

The second trend has been where species have proved to be economically viable, leading to increased production levels. This applies to *Gracilaria*, the Mediterranean mussel, and in particular to abalone. Although the Pacific oyster has proved an economically viable mariculture species, total recent production has not increased. This, however, has resulted from environmental factors and does not reflect the viability of the species. A number of species have been identified as having good mariculture potential and numerous experiments are being conducted to assess this. Marine finfishes, particularly species of indigenous origin, appear to have very good potential for future development. A future trend is therefore likely to include development and expansion of marine finfish culture.

Environmental monitoring of the waters used for mariculture production has become extremely important in order that products conform to international standards. In the past, such monitoring has been inadequate in South Africa, and this will have to be improved if farmers wish to continue to export their products.

Because the Benguela region of South Africa has relatively few areas of sheltered water, mariculture activities are concentrated into a few prime sites. Research has been directed at establishing the carrying capacity of such sites, and the results are likely to limit the future expansion of mariculture activities (Cook & Grant 1998). In addition, the monitoring and control of HABs in the future will become increasingly important as population growth dictates that increasing numbers of people will be living close to the coast.

Mariculture in southern Africa is seen as an industry with a great deal of potential for development and expansion. The recently adopted South African Marine Living Resources Act recognises mariculture as a growth industry and the directorate of Marine and Coastal Management has been charged with the duty of developing and regulating the industry. It is therefore likely that mariculture will expand rapidly in the near future. The main areas in which short-term future growth is likely to occur are in abalone, seaweed and marine finfish farming. All these industries are relatively new to southern Africa. Although the marine finfish industry is still at an experimental level, the others have been expanding rapidly over the past few years. One of the principal objectives of management will be to ensure that expansion occurs in an environmentally acceptable and sustainable manner. Environmental monitoring of mariculture activities will be essential to ensure this sustainable development.

Marine invasive aliens

Marine organisms have intentionally, or more often accidentally, been transported across the world's oceans since the earliest human attempts at exploration, colonisation, and commerce (Carlton 1989, 1999). On an intraoceanic scale these movements date back thousands of years, but interoceanic movements only really became significant in the age of European global exploration, which began in the 1400s.

Early marine introductions took place via two main mechanisms. Semiterrestrial and littoral marine species were transported along with rocks, sand or other dry ballast placed inside the holds of ships. A much more diverse group of attached fouling species, particularly algae, barnacles, hydroids, ascidians and molluscs, were transported on the outside of the hull, or bored into the timber of the vessels. How many such species survived voyages through the tropics to southern temperate waters is unknown, but Carlton (1999) has estimated that early wooden vessels could have each carried 150 or more species per voyage. Many of these vessels sank en route or remained in their ports of destination, thus inoculating substantial and concentrated populations of potentially invasive species at these sites.

The proportion of early introductions that resulted in naturalised populations is extremely hard to estimate in retrospect, since these introductions predate any kind of ecological survey, sometimes by hundreds of years. What seems certain, however, is that many species now regarded as widespread or cosmopolitan were in reality introduced by early explorers. Such species are now referred to as cryptogenic (of unknown origin) and may include some of the most common and ecologically important species in the area of introduction. Potential examples of cryptogenic forms abound and are discussed by Carlton (1999). Several such species occur in the Benguela region, where they have been considered as naturally occurring. Examples include a variety of fouling species, including hydroids (*Obelia*), bryozoans (*Membranipora*), ascidians (*Botryllus*, *Botrylloides*) and amphipods (*Corophium*, *Jassa*). Shell and wood-boring cryptogenic species are also represented in the Benguela region, for example, the sponges *Cliona* spp. and the amphipod *Chelura terebrans* (the better-known wood-boring shipworm *Banksia carinata* and isopod *Spheroma terebrans* are found only on the Indian Ocean coast of South Africa).

Over the past 100 yr the mechanisms by which marine species have been transported have undergone considerable change. Boring species are no longer able to penetrate the steel hulls of modern vessels, and although some fouling forms still cling to the outside of vessels, antifouling paints and increased ship speeds have progressively reduced the importance of this as a transport mechanism. Other vectors, however, have increased in importance. These include the deliberate introduction of marine species for mariculture, research or aquarium display purposes (sometimes along with their parasites or pathogens). Far more important, however, is the transport of species in ballast water, which is taken on board cargo vessels to improve their stability and trim. The scale of this mechanism is unprecedented. Carlton (1999) estimates that if only 10% of the 35,000 vessels at sea at any given time are carrying ballast, and if just two unique species are carried in each vessel, more than 7000 species are "in motion" on the world's oceans on any given day. Ballast water, with its fauna, is moreover probably turning over at a rate of 5–10% d^{-1}, giving a much higher total if longer time periods are considered. The net result has been a huge increase in rates of introduction with each succeeding decade.

Because of the increased speed of vessels, ballast water is capable of transporting both adult planktonic and larval benthic forms. The trend has thus been one away from transport of sessile adult invertebrates, toward introduction of larval and planktonic forms, which can include pathogens and toxic dinoflagellate spores. The construction of new harbour areas, especially in regions of high conservation value, also establishes new foci for introduction. A prime example here has been the construction of the new harbour in Saldanha Bay, an ecologically sensitive and previously relatively pristine bay, in the 1970s. The volume of ballast water transported into this site was estimated at 6.8 million t yr^{-1} by Carter (1996).

What, then, is the situation with respect to alien marine species along the west coast of southern Africa? Griffiths (2000) lists 22 marine species introduced to South African waters, while one additional form (the alga *Schimmelmannia elegans* from Tristan da Cunha) has subsequently been reported from Table Bay docks (De Clerck et al. 2002). However, several of the species listed by Griffiths (2000) are known to have become locally extinct, are dubious records (e.g., observations of dead mollusc shells), are of uncertain introduced status, or occur only along the Indian Ocean coast. Only eight introduced species are confirmed and extant in the Benguela region. One of these, the oyster *Crassostrea gigas*, is an intentional introduction and is farmed at a variety of sites (there have been recent reports of wild populations of this species in estuaries along the Southern Cape coast, but this is just outside the region covered by this review). The remaining seven are accidental introducions, most of which are confined to harbours or lagoonal sites, where their impact is limited. Only one such species, the ascidian *Ciona intestinalis*, is known to have environmental or ecological impact, as it fouls the ropes of mussel culture rafts and smothers the mussel spat. Only two introduced species have become invasive on the open coast of the Benguela. These are the Mediterranean mussel *Mytilus galloprovincialis* and the European shore crab *Carcinus maenas*.

Carcinus maenas was first recorded from Table Bay docks in 1983 (Joska & Branch 1986) and by 1990 had spread from Camps Bay to Saldanha Bay (Le Roux et al. 1990), a distance of some 100 km. No significant subsequent expansion has been noted; indeed, no further specimens have been reported from Saldanha Bay since the single observation reported by Le Roux et al. (1990) and one dead carapace collected in 2002 (C.L. Griffiths, personal observation). Notably all established populations are from shores sheltered from waves, suggesting that *Carcinus maenas* has difficulty establishing on the exposed open coastline of the Benguela region, and hence in reaching suitable sites distant from its point of origin. There is, however, considerable concern that this species will cause severe damage if it does establish a population in the Saldanha Bay–Langebaan Lagoon complex, which has an abundance of suitable sheltered habitat and is also an important conservation and mariculture site.

Mytilus galloprovincialis was first recorded in South Africa by Grant et al. (1984). By that stage it had already established extensive populations along the entire west coast between Cape Point and Luderitz. The exact date and site of introduction are unknown, but circumstantial evidence suggests that it was recent, probably in the late 1970s, and mediated by man (Hockey & van Erkom Schurink 1992, Griffiths et al. 1992). By the early 1990s *Mytilus galloprovincialis* had spread as far east as Port Alfred, in the Southeastern Cape, and was the dominant intertidal organism along the entire west coast, with an estimated biomass of some 194 t wet mass km^{-1} of rocky coast (van Erkom Schurink & Griffiths 1990). Unpublished data suggest that standing stocks on the west coast have subsequently increased substantially and that the species continues to spread eastward at a rate of about 10–20 km yr^{-1} (N. Steffani & C.D. McQuaid, personal communication). There has also been further northward expansion into northern Namibia, although densities remain lower in this region than in South Africa.

Relative to indigenous mussel species, *Mytilus galloprovincialis* has a rapid growth rate, high fecundity, and enhanced tolerance to desiccation (van Erkom Schurink & Griffiths 1990). Its spread has therefore greatly increased the overall biomass and vertical extent of mussel beds along the entire west and (to a lesser extent) south coasts of South Africa. This has had several implications for the wider intertidal community.

Before the arrival of *Mytilus galloprovincialis*, the dominant space-occupying invertebrates in the mid- to low intertidal of the Cape west coast were the slow-growing, indigenous mussel *Aulacomya ater* and limpets, notably *Scutellastra* (formerly *Patella) granularis* and *Scutellastra argenvillei*. Much of the shore was open space, kept clear by the intense grazing activities of these and other large limpets. Because *Mytilus galloprovincialis* grows much faster and extends higher into the intertidal zone than *Aulacomya ater* (van Erkom Schurink & Griffiths 1990, 1993), it is now the dominant space-occupying intertidal species at most sites. The net result has been a massive increase in both mussel cover and biomass and a movement of both the upper limit and centre of gravity of the mussel beds upshore. This has been accompanied by a decline in the overall biomass of *Aulacomya ater*, although paradoxically this species now occurs higher up the shore than before, since it can find protection within the dense mats of *Mytilus galloprovincialis*.

As well as outcompeting indigenous mussel species, *Mytilus galloprovincialis* competes successfully for primary rock space against adult limpets (Hockey & van Erkom Schurink 1992, Griffiths et al. 1992). However, the shells of large mussels also offer a favoured settlement and recruitment site for juvenile limpets. Thus, as mussels encroach, adult limpets initially become spatially constrained and then eventually eliminated. However, at the same time, the densities of smaller limpets on the mussels increase enormously. The net result is an initial increase in limpet biomass (while the adults on the rock and the juveniles on the mussels are both present), followed by a decline, when the adults are finally eliminated. Effects on limpet fecundity differ between species. In the midshore, small *Scutellastra granularis* living on mussel shells attain sexual maturity and the overall mass of gametes released by the dense population of small individuals in areas of 100% mussel cover actually exceeds that of the few large animals previously found on the bare rock (Griffiths et al. 1992). By contrast, the larger, low-shore limpet *Scutellastra argenvillei* also recruits onto mussel shells, but it is unable to attain sexual maturity in this confined habitat. As a result, reproductive output cannot be maintained in areas invaded by mussels (N. Steffani, personal communication).

Mussel beds are also structurally complex habitats that provide refuge to a diverse community of associated organisms. Griffiths et al. (1992) showed that the infaunal communities colonising *Mytilus galloprovincialis* and *Aulacomya ater* beds were similar in both species richness and composition (69 and 68 species, respectively, with 70% shared). However, because *Mytilus galloprovincialis* grow to a larger size, attain a greater biomass, and develop thicker, more structurally complex beds, they support a much denser invertebrate fauna than *Aulacomya ater* beds (76,600 vs. 34,000 individuals m^{-2}). They also tend to provide refuge for larger infaunal individuals.

A third implication of the increase in overall mussel biomass is that mussels form an important component in the diets of a wide range of predatory species, including both aquatic forms, such as fishes, rock lobsters, starfishes, predatory whelks and octopuses, and terrestrial ones, including shorebirds and humans (Griffiths & Hockey 1987). The introduction of *Mytilus galloprovincalis* has led to a massive increase in mussel standing stock, and hence enhanced food availability to natural predators. The benefits to terrestrial species may be particularly marked because they are now able to gain access to mussel stocks in the upper shore, where none occurred previously. Hockey & van Erkom Schurink (1992) and Griffiths et al. (1992) both compared the contents of oystercatcher middens in Saldanha Bay before and after the *Mytilus galloprovincialis* invasion. They show a dramatic switch in oystercatcher diet away from limpets (31 to 19%) and *Aulacomya ater* (36 to 3%) in favour of *Mytilus galloprovincialis*, which comprised over 66% of oystercatcher diet between 1987 and 1991. The increased availability and accessibility of food to oystercatchers resulting from the *Mytilus galloprovin-*

cialis invasion have simultaneously reduced the length of time oystercatchers have to forage on each tide (and hence increased their resilience to storms, human disturbance, etc.). They have also increased the proportion of pairs successfully raising two chicks (Hockey & van Erkom Schurink 1992).

As shown above, the *Mytilus galloprovincialis* invasion has had profound effects on the appearance and ecological processes occurring within rocky shores over much of the South African coastline. While the ecological costs of this invasion may be significant and are negative in the case of the limpet *Scutellastra argenvillei*, which is characteristic of seashores in this region, the economic implications are not necessarily negative. The only costs (so far undetermined) may be additional fouling of ships' hulls, seawater intake pipes, and other marine structures. However, since *Mytilus galloprovincialis* is essentially an intertidal species, these costs are not excessive. On the positive side, *Mytilus galloprovincialis* has formed the basis of a substantial mariculture industry, based almost entirely within Saldanha Bay (see mariculture section). The small-scale commercial exploitation of wild intertidal stocks is also being considered and may provide the basis for a small-scale industry in the economically depressed coastal settlements of the region. The benefits so obtained seem likely to outweigh the costs of removal from areas in which this species is currently problematic.

Marine construction and mining

The Benguela region is an area of low population density and the only significant engineering activities that occur along this coast are marine diamond mining and construction of a small number of harbours.

Diamond mining

The first diamonds discovered on the west coast of southern Africa were found near Lüderitz, Namibia, in 1908 (Clark et al. 1999) and were exploited by terrestrial and beach mining. In 1926 the first South African discoveries were made near Port Nolloth. Drs. Merensky and Reuning recognised the link between old marine terraces and diamond deposits, which led to prospectors examining the areas immediately north and south of the Orange River mouth. Today, concessions spanning the entire coastline, from the Kunene to the Olifants River, are set apart for beach mining, but less than 1% of this area is currently being mined (Clark et al. 1999).

Once a deposit is located, the overburden is stripped by a variety of earth-moving machines (Clark et al. 1999). Excavation is generally undertaken on a block-by-block basis and the overburden used to build a seawall extending the shoreline a few hundred metres out to sea. The bulk of the diamond-bearing gravel is then removed and transported to treatment plants, where the diamonds are extracted. The mine tailings are then disposed of on tailing dumps or pumped directly into the sea. This has a significant effect on the beaches, since the tailings are coarser than the beach sediment. In Elizabeth Bay, Namibia, this habitat change has resulted in a change from a mussel- to a crustacean-dominated community, with a loss of diversity. The sediment pumped into the sea is also known to settle out on reefs or rocky shores and suffocate the organisms living there. Fishes, however, appear to benefit from the sediment plume, as it provides shelter from predators; thus, their species richness and abundance increase. It is thought that, provided seawalls were constructed from sediment of the same grain size as the beach, the communities should return to their original state, as the walls are eventually eroded by wave action.

The first offshore mining took place in 1961 (Clark et al. 1999). During the late 1960s the diamond market slumped and offshore mining operations ceased in 1971. Smaller-scale operators continued to mine from converted fishing vessels, and these shallow-water operations increased over the years. During the early 1990s deepwater offshore mining resumed. Offshore mining is centred around Oranjemund, at the mouth of the Orange River, but concessions spanning the entire Namibian coast and west coast of South Africa have been divided between mining companies (Figure 30). Less than 1% of this area is currently being mined. There are three main categories of offshore mining: shallow water, which consists of divers operating from shore or from small boats to a depth of 30 m; midwater, remotely operated tools, mostly airlift dredges, utilised at 30–75 m; and deep water, customised mining vessels and specially designed remotely operated mining tools for operations deeper than 75 m.

Most mining activities take place at depths of 110–135 m (Van der Merwe 1996). Two methods are employed to extract the sediment, underwater crawlers and large rotating drills. These are thought to be equivalent in their severity of disturbance. High-powered airlift suction is used to deliver the sediment to the anchored mining vessel, and once the sediment is screened for diamonds,

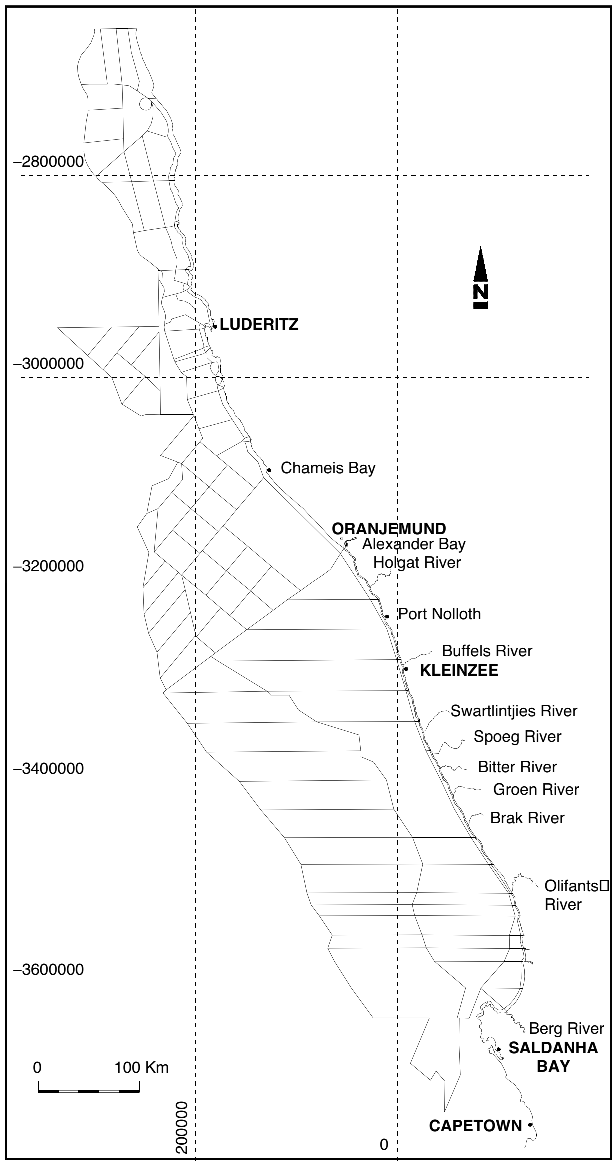

Figure 30 Map of the west coast of South Africa and Namibia showing how the shelf is divided into mining concessions.

it is released overboard. The heavier gravel sinks to the seafloor almost immediately, but the fine sediment remains suspended in the water column and can travel great distances before finally being deposited. This alters the sediment patterns in the area. Unmined sites are characterised by layers of gravel overlain by fine to very fine sand, whereas at mined sites, the sediment is a mixture of sediment sizes and types. This alteration, however, is temporary because the water circulation patterns responsible for the sediment patterns are still functioning unhindered. A similar method is employed in midwater mining (Clark et al. 1999). The ships range from 50–140 m in size and operate on a 24-h basis for 11 months of the year.

Mining activity has both direct and indirect effects on benthic macrofaunal communities. The direct effect is the mortality of organisms as a result of gravel extraction. The indirect effect is a result of the resuspension of fine sediments and the altering of sediment granulometry. This change in the habitat causes a shift in benthic community structure. Van der Merwe (1996) assessed the rates of recovery of the benthic macrofauna after mining. Grab samples from unmined areas were compared with samples from areas mined at different times in the past, providing a quasi-time series of recovery after mining. The change in community composition and species diversity was immediate (within 1 month) due to mortality from gravel extraction (Figure 31). This was followed by a period of deterioration (up to 19 months) due to the slow reaction time of the benthos to disturbance. After 43–51 months the benthos samples from mined areas seemed to have recovered in terms of diversity and were no longer characteristic of disturbed areas. However, the species composition had not returned to that of the unmined state. It was interesting to note that the sediment granulometry returned to the initial state within 19 months, indicating the slow reaction time of the benthos. Thus, although areas disturbed by mining activities can recover to their initial state, it may take up to a decade to do so.

Shore-based shallow-water mining typically consists of two or three divers, their assistants, and a tractor (Clark et al. 1999). The divers operate on surface-supplied diving equipment and suck up gravel through a hose. The gravel goes to a rotary classifier, the gravel concentrate being collected and sorted onshore. Boat-based mining typically consists of a 10- to 15-m vessel with a five- to eight-man crew, of which two or three are divers. The vessel is generally fitted with one or two hoses and activities are limited to daylight hours for 3–10 diving days month^{-1}. Some larger vessels (20–22 m) are also operational on a 24-h basis for up to 21 days month^{-1}. Pulfrich & Penney (1998) investigated the effect of this method on the benthos and concluded that mining resulted in a change in the benthic community composition, but that the benthos should return to the unmined state in a few years.

Harbour developments

The major industrial harbours in the Benguela region are located only in Cape Town, Saldanha Bay, and Walvis Bay (and on a smaller scale at Lüderitz), whereas fishing harbours are present at most other coastal settlements.

Since the early 1600s, sheltered bays such as Table Bay, Saldanha Bay, and Walvis Bay have been used as natural harbours, but it was only in 1890 that the first international harbour in the Benguela region was constructed in Table Bay. Table Bay docks have subsequently been subject to a series of major extensions and modifications (Figure 32A) that have profoundly changed the nature of the shoreline in the bay. A new foreshore and development scheme, proposed in 1937, commenced in 1939 with the reclamation of land from the sea and eventually pushed back the shoreline approximately 1.5 km. Between 1975 and 1985, two deep-sea container berths and two coaster berths were added (Figure 32A). In contrast, Saldanha Bay harbour, situated about 150 km north of Cape Town, was only built in the 1970s, as an export vehicle for the steel and ore industry. Between 1971 and 1984, the breakwater linking Marcus Island to the mainland, the ore jetty, the oil jetty, the multipurpose terminal, the general maintenance quay, and the small-craft harbour (Figure 32B) were all constructed and the export of iron ore commenced. Industrial development

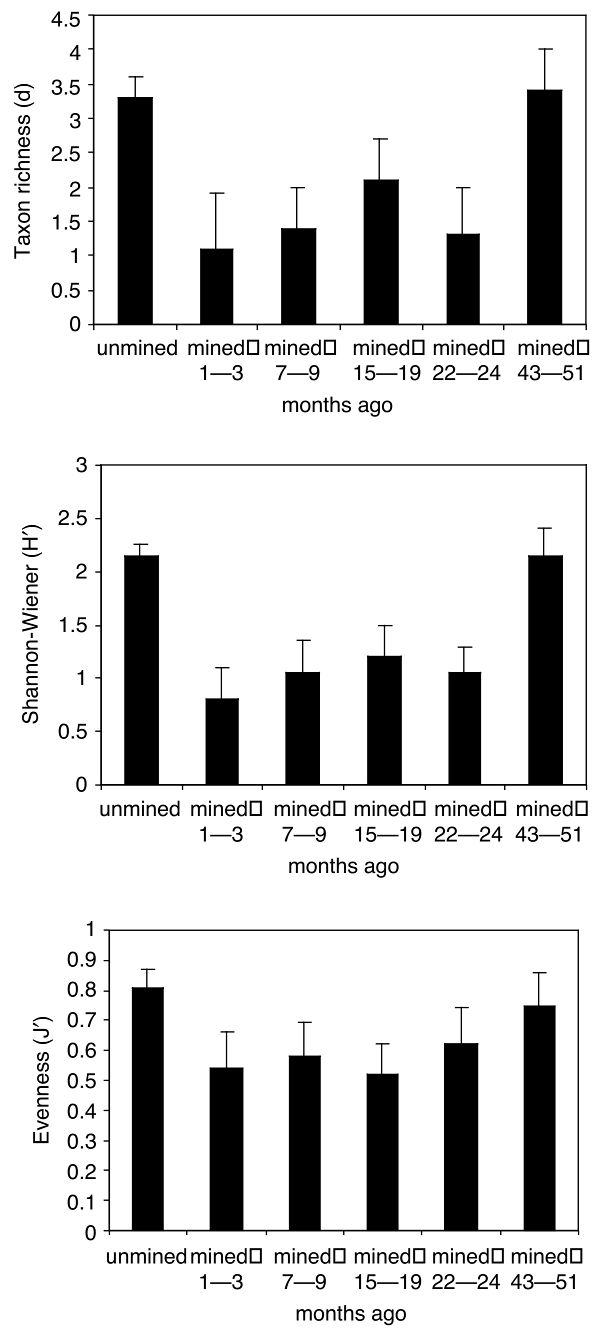

Figure 31 Diversity indices indicating the recovery of the benthos following deep-sea mining. (After Van der Merwe 1996.)

in Saldanha in the early 1990s led to the extension of the multipurpose terminal. All of these changes took place over a period of only 30 yr.

Harbour construction has both direct and indirect effects on the fauna. This has been best documented for Saldanha Bay (Moldan 1978, Monteiro et al. 1999). The direct effects of harbour construction include dredging, which causes mortality to organisms being removed with the sediment and to organisms buried under sediment. However, recolonisation of the dredged areas

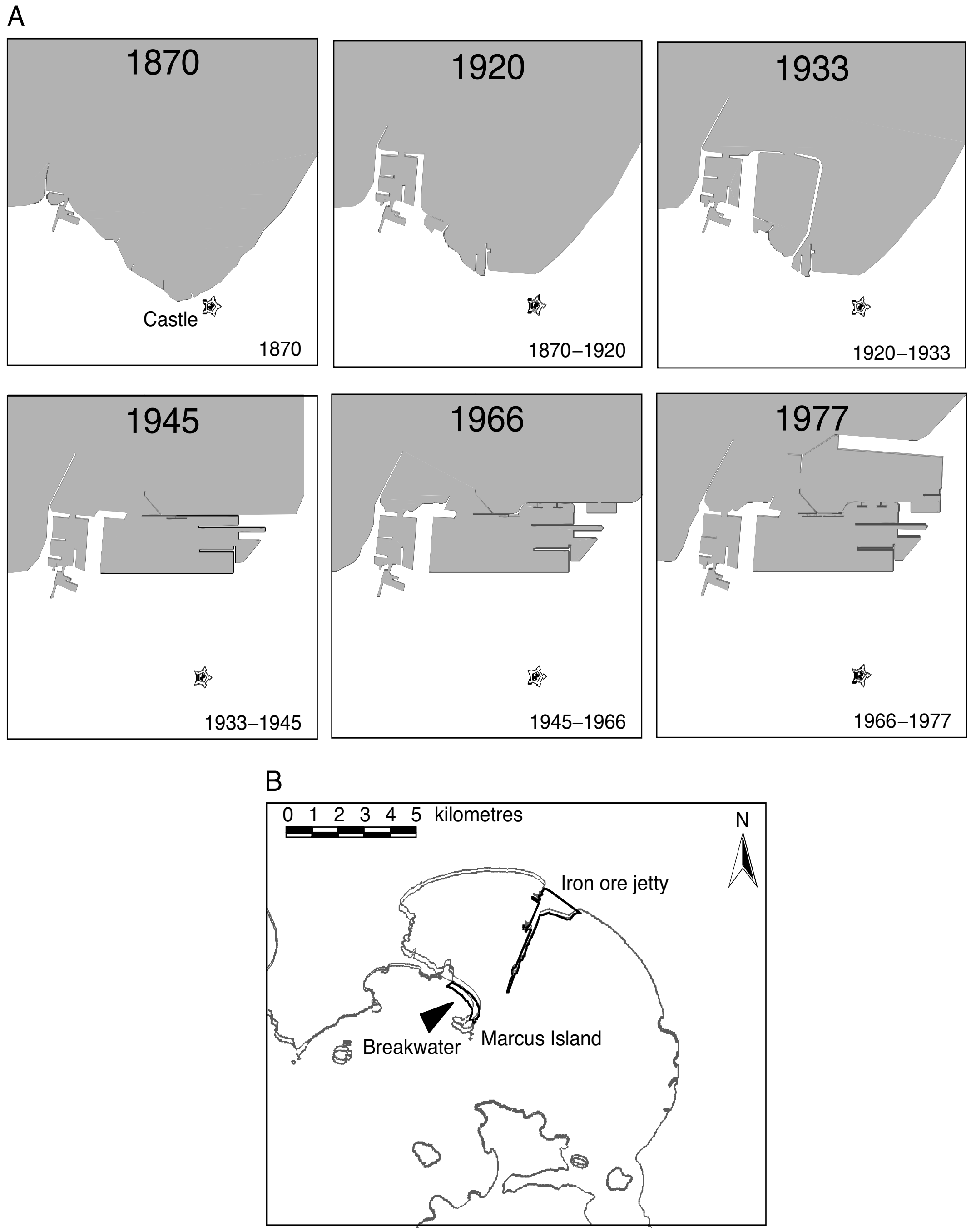

Figure 32 (A) Historical sequence of developments of Table Bay Harbour, 1870–1977. (B) Saldanha Bay Harbour, showing major developments since 1970.

in Saldanha occurred within 2 yr (Moldan 1978). The breakwater, loading terminal, and jetty are also physical barriers to water movement within the bay and result in decreased wave exposure within the bay, which in turn has altered the sediment distribution pattern (Monteiro et al. 1999). This habitat change is permanent and has resulted in significant changes in the macrobenthic community structure. A comparison of the benthic communities before and after harbour construction (Kruger, unpublished) revealed no change in overall species diversity or species richness, but a species mix very different from that before harbour construction. In fact, 50% of the species present before harbour construction have been lost and replaced by species preferring a more sheltered habitat.

In conclusion, both harbour construction and marine diamond mining have been shown to have some serious direct negative environmental effects, but these are believed to be localised and thus unlikely to have a significant effect on the ecological functioning of the Benguela as a whole. The indirect effects of the establishment of harbours, in that they may provide foci for the introduction of alien invasive species (see above) and both open up new areas for fishing and result in increased levels of exploitation, probably have far more profound environmental effects than the construction of the harbour itself, but they remain unquantified.

Pollution

The earliest reference to marine pollution in the Benguela region, and indeed in southern Africa, dates from 1811, when a British soldier stationed at the Cape of Good Hope recorded that all kinds of waste were conveyed daily to the shore of Table Bay and deposited in the surf, polluting it for lengthy periods, and that whales were cut up on the beach, contaminating both sea and air (Ewart 1970). There is virtually no further reference to marine pollution in the region for another 100 yr, despite increases in population and the growth of industry. The latter began after the turn of the 20th century, initially focused in Table Bay, but later spreading to other bays along the coast — False Bay, Walker Bay, Gansbaai, Saldanha Bay, St. Helena Bay and Lambert's Bay, and eventually Walvis Bay (Namibia). It continued to be thought that anything could be dumped into the sea with impunity, the only criteria being aesthetic, particularly with regard to sewage disposal to sea.

During the Second World War, several major oil slicks in the region caused concern because of the wholesale deaths of marine organisms, especially birds. Catastrophic oil pollution has been an intermittent problem ever since. By 1977, 650 million t of petroleum yr^{-1} were being transported through the region, resulting in up to 450,000 t polluting the sea (CSIR 1979). In recent decades oil pollution has diminished, due to improved legislation and cleanup procedures, although oil spills continue to be a hazard to birds such as penguins.

The establishment of the CSIR (South African Council for Scientific and Industrial Research) in 1945 marked a turning point in pollution studies. A major appointment was that of Dr. G.J. Stander, who designed and implemented the world's first plant for the total recycling of sewage onshore. This was built in Windhoek in 1969 and Namibia can today claim to be the only coastal country in Africa to have no raw sewage discharged to sea. In 1973 the CSIR initiated a programme to study and combat marine pollution around the South African coast. Work sponsored by the programme indicated that southern African coastal waters presented very low levels of pollution by international standards; only specific impact areas (the bays already mentioned) have potentially damaging levels. This was confirmed by an international workshop, which considered the transfer of pollutants between the Agulhas and Benguela systems (CSIR 1979). The CSIR initiative also resulted in the first bibliography of marine pollution for South Africa (Darracott & Cloete 1976).

By 1984 there were 61 pipelines discharging effluent to sea along the South African coast, most of them commissioned since the end of the Second World War and about a third in the Benguela region. To these must be added numerous surf zone discharges, pollution from harbours, and a host of small industries discharging into storm water drains, as well as runoff polluted by agriculture (Brown 1987). Attempts to consolidate available information on impact areas began to bear fruit (e.g., Quick & Roberts 1993), and this was facilitated by programmes such as the CSIR's studies of the impact of fish factory effluents on the benthic fauna (e.g., Bickerton et al. 1997). The reports show this impact to be extremely local, there being no significant impact outside the bays in which the factories are situated.

Although studies have continued to focus on major impact sites, a more general monitoring programme was initiated by the Department of Marine and Coastal Management in its Mussel Watch Programme, begun in 1985 and ongoing. The regular assessment of metal concentrations in the tissues of *Mytilus galloprovincialis* from 42 sites around the Cape Peninsula demonstrates that levels of lead have declined over the years, although concentrations in parts of False Bay continue to be unsatisfactory. Levels of zinc and cadmium have been consistently elevated in west coast waters, even in nonimpact areas. This is possibly due to natural causes. Pollution by plastics has also caused concern since the early 1980s. A wide variety of plastic materials pollute the region, but industrial pellets account for most of the mass (Ryan 1988). The pollution particularly impacts seals, seabirds and fishes, as a result of both ingestion of small particles and entanglement with larger pieces of plastic.

Bailey (1996) reviewed all major outfalls, monitoring procedures, and flow volumes around the South African coast (Table 2). It remains apparent that, although we have some insight into the extent of pollution in impact areas and its effect on benthic invertebrates, our knowledge of impacts on commercially exploited animals remains scanty. Little attention has been paid to pollution in the Benguela region as a whole, and the possible effects of the discharge of nutrients, particularly nitrogen and phosphorus, have been neglected. Excessive nitrogen enrichment, from sewage, storm water, agriculture and industry, is recognised as a worldwide problem (World Resources Institute 1998), resulting in plankton blooms and consequent perturbation of marine ecosystems. The increasingly common onshore migrations and subsequent death of rock lobsters (*Jasus lalandii*) on South Africa's west coast have been attributed to plankton blooms depriving the biota of oxygen; nearly a million kilograms of lobsters perished in this way at Elands Bay in March 2002. Elevated nitrogen levels may well play a role in this phenomenon.

Despite the justifiable concern with marine pollution, it may be concluded that thus far pollution in the Benguela region has had a negligible impact on the biota as a whole, certainly as compared to the exploitation of resources. It is also apparent that pollution levels are low compared with most other regions, especially of industrialised countries.

Climate change

The International Panel on Climate Change (IPCC) states unequivocally that the Earth's climate is changing. Global average surface temperature has been increasing at a rate of about 0.15˚C decade^{-1} since the late 19th century, the total increase since that time being about 0.6˚C (IPCC 2001). Rates of warming are believed to have been greater on the land than the sea, the increase in sea surface temperature (SST) in the period 1950–93 being about half that on land. Observed increases in temperature are believed to be a function of anthropogenic production of greenhouse gases, notably CO_2, methane, nitrous oxide and chlorofluorocarbons (CFCs), that have the ability to trap long-wave radiation emitted by the Earth. Concentrations of CO_2 in the atmosphere have increased by 31% since 1750, methane by 150%, and NO_2 by 16% (IPCC 2001). Annual land precipitation in the middle and high latitudes, and tropical regions of the Northern Hemisphere, has increased (0.5–1% decade^{-1} in the former and 0.2–0.3% decade^{-1} in the latter regions) over the 20th century, but has decreased (0.2–0.3% decade^{-1}) in the subtropics (10–30˚N). No comparable systematic changes in rainfall have been detected in the Southern Hemisphere. Global mean sea level is rising at a rate of 1.0–2.0 mm yr^{-1}, attributable to thermal expansion of the oceans (the major component) and the melting of the polar ice caps. The behaviour of the El Niño Southern Oscillation (ENSO) has been unusual since the mid-1970s, compared with the previous 100 yr, with the warm phases becoming more frequent, persistent, and intense than the cool phases. World climates have also

Table 2 Main marine outfalls along the west coast of South Africa (partly after Bailey 1996, Lees 1969 and the South African Fishing Industry Handbook 1994)

Place/company	Type	Average flow (m^3 d^{-1})	Notes
Gansbaai: Marine Products	Fish	28,500	Surf zone outfall into high-energy bay: established in 1962
Hermanus	Domestic fish	796	Several small surf zone outfalls into Walker Bay since 1945
Gordons Bay	Domestic	1000	Surf zone outfall of treated effluent; fully treated surf zone outfall
Strandfontein (False Bay)	Domestic	70,000	North coast of False Bay, since 1970
Simonstown: Marine Oil	Fish oil	140	Commissioned in 1960s, decommissioned in 2000; mainly glycerol
Hout Bay	Domestic fish	15,000	Pipeline discharge, commissioned in the 1980s; untreated sewage
Hout Bay: Da Gama	Fish	1130	Surf zone discharge, operating since 1959
Llundudno	Domestic	160	Surf zone discharge, treated sewage
Camps Bay	Domestic	34,000	1-km pipeline, commissioned 1977; untreated sewage
Green Point	Domestic	29,000	1-km pipeline, commissioned c. 1920, twice upgraded; untreated sewage
Milnerton: Caltex	Oil refinery	3000	500-m pipeline, maximum flow of 9000 m^3 d^{-1}; operating since 1966
Milnerton: Kynock	Fertiliser	1000	500-m pipeline, commissioned in 1969, decommissioned in 2002; mainly ammonium nitrate
Saldanha Bay: Sea Harvest	Fish	3350	Surf zone outfall discharging into sheltered bay; established in 1965
Saldanha Bay: Southern Seas	Fish	4000	Surf zone outfall discharging into sheltered bay; established in 1948
St. Helena Bay:Fish Company	Fish	3048	Surf zone outfall, established in 1946
St. Helena Bay: Suid Oranje	Fish	1095	Surf zone outfall, established in 1965
St. Helena Bay: West Point	Fish	2100	Surf zone outfall, established in 1950
Lamberts Bay: Canning Company	Fish	4322	Surf zone outfall into high-energy bay; established in 1918
Doring Bay: Canning Company	Fish	25	Surf zone outfall

become more variable and the intensity and frequency of extreme weather events appear to have increased (IPCC 2001). Loss of ozone from the stratosphere has been noted in many areas of the world, linked to the production of CFCs. The effects are particularly pronounced over Antarctica, where an ozone hole, characterised by the depletion of 60% or more of ozone, opens up over an area the size of Canada each spring (Smith et al. 1992). The decreases in stratospheric ozone have been accompanied by increases in UV-B radiation and inhibition of photosynthesis in Antarctic waters (Smith et al. 1992).

Changes recorded in southern Africa mirror these global patterns. Recent temperature trends over the Southern Hemisphere (1950–85) indicate a warming trend of 0.1–0.5°C decade^{-1} in the lower troposphere, rising to 0.2–0.8°C decade^{-1} in the latter part of this period (1966–85) (Tyson 1990, Karoly 1988). Warming in the Benguela region (west coast of South Africa) in this period was about 0.6°C (Tyson 1990). A slight warming trend has also been noted in SST data for the southeast Atlantic, corresponding to an increase of about 1°C in the period 1920–88 (Taunton-Clark & Shannon 1988). No large-scale systematic linear trends are evident in rainfall patterns during the 20th century (Tyson et al. 1975, Tyson 1986), but some evidence is available to suggest that variability and extremes are increasing in the south, particularly the drier western parts (i.e., the Benguela region; Tyson 1986, Nicholson 1986, 1993, Mason et al. 1999). A trend of increasing upwelling intensity has been observed in the Benguela over the last four decades (Shannon et al. 1992), mirrored by similar trends in most of the other major coastal ocean upwelling centres of the world (Bakun 1990). Bakun (1990) believes that these changes are a function of the buildup of CO_2 and other greenhouse gases in the atmosphere. He argues that the CO_2 buildup has enhanced daytime heating and reduced night time cooling, and has led to an intensification of continental lows adjacent to upwelling regions. This in turn, he argues, has increased onshore–offshore pressure gradients, intensified alongshore winds, and hence accelerated coastal upwelling.

With intensified upwelling one would expect an increase in primary productivity, but data from the Benguela indicate that, if anything, chlorophyll *a* concentrations have declined in recent decades (Brown & Cochrane 1991). Abundances of zooplankton, on the other hand, have increased over a similar period (Verheye et al. 1998). Tide gauge records for the Benguela region indicate that sea levels have risen by approximately 1.2 mm yr^{-1} over the last three decades and are in close agreement with the international estimates (Brundrit 1995).

Future projected changes in climate for the Benguela region

Numerical models generally referred to as global climate models (GCMs) provide the only quantitative estimates of future climate change. These models are based on physical laws represented by mathematical equations that are solved using a three-dimensional grid over the globe. A large number of GCM experiments have been completed recently, employing a variety of different models. Most of the predictions reported here are derived from these experiments. It must be acknowledged, however, that the ability of these models to provide accurate predictions is still questionable, particularly with respect to regional level prediction (Michell & Hulme 1999 and references cited therein).

Ragab & Prudhomme (2002) provide predictions of changes in land surface temperature and precipitation for southern Africa, including the countries bordering the Benguela region (Angola, Namibia, and South Africa), generated by the U.K. Hadley Centre's global climate model using the IS92a forcing scenario (this assumes an increase in atmospheric CO_2 of 1% yr^{-1}). They predict that by 2050 annual average temperatures will have increased by 1.0–2.75°C, with winter increases projected to be slightly greater than those in summer. Predicted changes in average annual rainfall in 2050 over the Benguela region varies widely, ranging from –25 to +25%. Average rainfall over the South African west coast is expected to decrease by 0–15% (slightly worse in summer than winter), to increase on average by 5–25% on the southern, central, and extreme northern parts of Namibia (summer and winter being similar), and to decrease on average (0–10%) in the lower northern parts of Namibia (summer worse than winter).

Schulze et al. (2001) provide predictions of changes in annual rainfall and river runoff over southern and eastern Africa for 2050 from the UKTR95 GCM and ACRU agrohydrological modelling system. They predict that both rainfall and annual runoff will decrease by 0–30% across the entire Namibian and South African west coast, the hardest hit areas being the extreme northern and southern parts of Namibia and the northern half of South Africa. Arnell (1999) also used data from the U.K. Hadley Centre's global climate model (HadCM2 and HadCM3), together with a macroscale hydrological model, to simulate river flow across the globe at a spatial resolution of 0.5° latitude × 0.5° longitude. On this basis he predicts that average annual runoff to the Benguela would decrease by 0–50 mm yr^{-1} (from an average of 0–200 mm yr^{-1}), making the percentage runoff change in southern Africa among the highest in the world. These projections correspond closely with those reported by Clark et al. (2000), who estimated reduction in runoff from four rivers on the South African west coast to be in the region of 35–84% under a 2 × CO_2 scenario, using the HadCM2 GCM coupled to the ACRU modelling system.

Clark et al. (2000) also provide projections of changes in pressure systems and wind fields over southern Africa for spring and summer under a double CO_2 scenario, using data from the National Centre for Atmospheric Research (NCAR) Climate System. This period was chosen because it corresponds to the period of most intense upwelling and the spawning period for pelagic fishes in the Benguela. The results of this analysis suggest that the South Atlantic High Pressure System will intensify, especially in the late summer months, and will ridge farther south and east of the subcontinent than it does at present. Southerly and easterly winds are expected to increase over the Benguela region as a result, generating more intense upwelling.

The current trend of rising sea level is expected to accelerate in the future, with recent estimates (based on the HadCM2 and HadCM3 models) indicating a 12.3-cm rise by 2020, a 24.5-cm rise by 2050 and a 40.7-cm rise by 2080 (Nicholls et al. 1999).

Projected impacts of climate change on marine biota

Relationships between biological and physical environmental processes are not well understood for the Benguela. Even greater uncertainty must thus be attached to projections regarding effects of climate change on marine biota than to the changes themselves. However, most authors are of the opinion that change in wind stress in the Benguela region is likely to have more pronounced consequences for marine biota than other effects, such as increasing temperature, sea level rise, changing rainfall, and river runoff, because of its influence on large-scale oceanographic processes (Siegfried et al. 1990, Brown & Cochrane 1991, Clark et al. 2000, Lutjeharms et al. 2001). Increases in wind stress over the Benguela region (considered to be the most likely outcome of climate change) are expected to result in an intensification of upwelling, increased nutrient availability, enhanced primary production, increased advection of cold upwelled water offshore and reduced rainfall over the adjacent subcontinent, all of which could affect pelagic and demersal food webs and fish production. Pelagic fish recruitment is dependent on a balance between food supply and losses across the open-ocean boundary, both of which are a function of wind stress. Best recruitment appears to occur under intermediate conditions and hence may be negatively affected if upwelling intensifies or diminishes.

Another phenomenon of the Benguela system that may be affected by changes in wind dynamics is the irregular occurrence of Benguela Niños (Shannon et al. 1986, Crawford et al. 1990, Siegfried et al. 1990, Lutjeharms et al. 2001). These events generally coincide with periods of low or sharply reduced zonal wind stress in the western equatorial Atlantic and are characterised by the sudden collapse of the Angola–Benguela front and a poleward flow of warm water along the coast from Angola into Namibia. They are usually accompanied by a southward penetration of tropical species such as *Sardinella aurita* and certain copepod species normally only found from Angola northward, a decrease in primary production off Namibia, southward displacement of local (Namibian) fish stocks, an influx of low oxygen water from the north and associated mortalities of fishes and other organisms. It is believed that changes in the equator–pole temperature gradient and poleward shifts in oceanic and atmospheric systems (considered to be a likely consequence of climate change) may lead to an increase in the frequency and intensity of these events, with immediate consequences for the upwelling system, sea surface temperatures in the region, and biota of the coastal zone (Siegfried et al. 1990, Lutjeharms et al. 2001).

Changes in the influence of the Agulhas Current on the Benguela system, brought on by changes in wind stress, may also be important in the future. The Agulhas Current flows down the east coast of South Africa and terminates in a tight loop south of the African subcontinent, the Agulhas retroflection. The current normally follows an extremely stable trajectory but is periodically (four to six times yr^{-1}) interrupted by a solitary meander, the Natal Pulse, which causes the current to shed a ring of warm water when it reaches the retroflection area. These rings then drift off into the South Atlantic, or up the west coast (Lutjeharms & van Ballegooyen 1988, Lutjeharms & Gordon 1987, Gordon & Haxby 1990). These rings have been observed to interact with upwelling plumes and can contribute to the failure of anchovy recruitment in the Southern Benguela and to a tendency for winter depressions moving past the Southwestern Cape to intensify (Duncombe Rae et al. 1992, Brundrit & Shannon 1989). Increases in wind stress over the South Indian Ocean (also a projected consequence of climate change) may lead to an increase in frequency of the Natal Pulse and, consequently, to an increased flux of Agulhas rings into the South Atlantic, with concomitant effects on the biota (Lutjeharms & de Ruiter 1996, Lutjeharms et al. 2001).

Temperature is generally considered to be one of the most important physical variables controlling the life of all aquatic organisms. Changing global temperatures could thus have far-reaching consequences for marine organisms in the Benguela. The most obvious changes that can be expected are that individual species, or species assemblages, will shift their distribution patterns. This is likely to be most pronounced in those species that are most temperature sensitive, or whose distribution patterns are strictly governed by temperature. Cold-tolerant species typically found only on the cool temperate west coast are thus likely to become more restricted in their distribution. They may retreat to greater depths or become restricted to the immediate vicinity of the stronger upwelling cells. Some of the warm-tolerant species from the east and south coasts may also expand their ranges southward and westward, possibly even extending around Cape Point onto the west coast.

Projected changes in stream flow (a function of changing rainfall patterns) are also likely to have serious consequences for estuaries of the Benguela region. Reductions in the frequency or intensity of flooding in particular have major consequences for estuaries (Reddering & Rust 1990). These include changes in the erosional capacity and other sedimentary processes, depth profiles, mouth configuration, duration of the open phases and tidal prism within the estuary. Sand shoals situated in the mouths and lower reaches of estuaries will grow larger, constricting the channel and reducing tidal exchange with the sea. Ultimately this will have the effect of increasing the frequency and length of time for which the mouth will close. A change in flow may also be accompanied by changes in nutrient levels, suspended particulate matter, temperature, conductivity, dissolved oxygen and turbidity (Drinkwater & Frank 1994), all of which play a role in structuring biological communities in estuaries. Many estuaries will simply remain closed for much of the year, or for several years at a time, thereby excluding many marine species. Many marine fishes in southern Africa make use of estuaries as nursery and breeding grounds (Wallace et al. 1984), estuaries on the west coast of South Africa being disproportionately more important than in the rest of the country, due to the paucity of sheltered embayments along this coast (Bennett 1994). These fishes have adapted their breeding habits to take advantage of the seasonal opening and closure of river mouths. Seasonal changes in river flow are likely to alter the timing of the open and closed phases and will negatively impact recruitment into these systems. A reduction in freshwater runoff is also likely to result in a reduction in the extent to which wastewater discharges are diluted before reaching estuaries. Thus the concentration of pollutants in estuarine waters will increase, while levels of dissolved oxygen will decrease, reducing the capacity of these environments to support biological communities.

The potential impacts of sea level rise on the coastal environment of the Benguela include increased coastal erosion, inundation, increased saltwater intrusion, raised groundwater tables and increased vulnerability to extreme storms (Klein & Nicholls 1999). Several major towns and cities, such as Cape Town, Walvis Bay, and Swakopmund, are situated at sea level and are thus at risk from some or all of these sources. Lutjeharms et al. (2001) are of the opinion that the impact of sea level rise on the ecological functioning of the Benguela system is likely to be insignificant, except in shallow coastal lagoons and estuaries, where much of the marine production in linked to salt marsh ecosystems. In areas where sea levels are rising and a strong supply of sediment is absent, marshes rapidly become waterlogged or completely inundated, and species unable to tolerate these conditions or the increased salinity from marine waters die back and expose the underlying sediments to further erosion (Beeftink 1979).

Certain minor responses can also be expected of marine plants and algae as a result of elevated CO_2 levels in the atmosphere. Some plants (e.g., sea grasses) are expected to show enhanced photosynthetic rates and growth, whereas others (e.g., intertidal macroalgae) are already CO_2 saturated and may not show any response (Beardall et al. 1998). Some response can also be expected from increases in ultraviolet radiation reaching the Earth's surface, related to losses in ozone from the upper atmosphere due to human production of CFCs. However, effects of increasing UV radiation are likely to be minor in comparison to other effects of climate change. Enhanced UV-B fluxes are likely to favour species with UV tolerance or repair mechanisms (Beardall et al. 1998). Intertidal species, for example, generally show less inhibition of photosynthesis by UV-B radiation than their subtidal counterparts. Increases in UV-B fluxes may thus exert some sort of control over species' distribution patterns (Larkum & Wood 1993, Beardall et al. 1998). UV-B radiation can also cause damage to early developmental stages of fish, shrimp, crab, and other species (Häder et al. 1995), and may thus disproportionately affect those species with planktonic larval stages.

Synthesis

The history of human impacts on the Benguela can usefully be subdivided into four broad epochs, each with its own distinctive pattern of resource usage. These epochs are termed the aboriginal (c. 10,000 BP–c. 1652), preindustrial (c. 1652–c. 1910), industrial (c. 1910–c. 1975) and postindustrial (c. 1975–c. 2002) periods.

The aboriginal epoch is reviewed separately above (p. 306) and marks a long period of low-level, opportunistic exploitation of mainly intertidal or stranded organisms. It appears very unlikely that these low levels of exploitation had any significant impacts on the stocks in question. This is both because of the limited absolute biomass removed from the system and because of the lack of technology required to hunt successfully for the larger predatory species in the system. This is a notable contrast to the situation in terrestrial systems, where low levels of aboriginal activity are thought to have resulted in the decline, or even extinction, of many large mammalian species.

In fact, subsistence and recreational exploitation of a type similar to that reported from the aboriginal epoch continues to take place in South Africa to this day. However, exploitation levels have remained at low levels of intensity in the Benguela, since this region still hosts a sparse human population. This is in marked contrast to the situation on the east coast of South Africa, where rapid human population growth and high levels of poverty have resulted in very intense levels of subsistence exploitation of intertidal species. These activities have been shown to have devastating consequences both on the populations of targeted species and on the structure of the intertidal community as a whole (for review, see Branch & Griffiths 1988).

The remaining three epochs are reviewed in chronological order below.

The Preindustrial Epoch (c. 1652–c. 1910)

The preindustrial epoch divides naturally into two: the Dutch (c. 1652–c. 1795) and the British (c. 1806–c. 1910) periods. The Dutch East India Company viewed the Cape primarily as a resupply station for its vessels trading between Europe and Asia. The demand on marine resources thus emanated almost entirely from small resident and transient maritime populations (Muller 1938). The

DEIC also discouraged private enterprise for the local market, except under company licence (Roux 1975). The effect was thus to minimise impacts on marine resources and inadvertently to afford them protection. The main marine resources exploited during this period were readily accessible inshore resources, such as whales, seals, guano and inshore linefish stocks, particularly snoek.

In the last quarter of the 18th century the DEIC faced a new challenge to its monopoly from an itinerant international whaling fleet. The subsequent unchecked "bay whaling" boom decimated the Benguela's southern right whale population. Thereafter, the new British authorities at the Cape opened the fishery to settler entrepreneurs, who rushed into shore-based whaling and, together with foreign whalers, precluded any recovery in the right whale population (see whale section, p. 308).

The British action signalled an important change in economic policy away from monopoly and toward *laissez faire*. Settler maritime enterprise was now encouraged and protected through the granting of property rights to coastal land. However, local demand remained small, and transport infrastructure too primitive to stimulate significant commercial fishing. This impetus came instead from the importation of indentured Indian labour onto the sugar plantations of the Southwest Indian Ocean, following the abolition of slavery in 1838. Cape Town merchants responded to the new regional demand for dried fish by leasing crown land between False Bay and St. Helena Bay for export snoek fisheries in the 1840s (Wardlaw Thompson 1913). The civil commissioner of Malmesbury went further, inaugurating a system of "fishing leases" in 1856, whereby lots of the crown land reserve above high water mark were let to all comers at £1 yr^{-1} (Cape of Good Hope 1882).

The African guano rush of 1843–45 opened up the coast north of St. Helena Bay to colonial enterprise. The Royal Navy finally imposed order and private property on the two epicentres of the guano rush, Ichabo and Malagas Island (Craig 1964). The guano islands were subsequently leased to local merchants, the imperial government annexing those in the no-man's-land north of the Orange River in 1861 to protect its property rights against foreign interlopers (Cape of Good Hope 1861). The British conversion of loan farms and wasteland into freehold tenure also catalysed a copper mining boom in Namaqualand (Smalberger 1975) and commercial grain production in the southern west coast interior (Marincowitz 1985). Both relied on the Benguela sea corridor to reach their markets. As a result, ports developed in the mouth of the Berg River, at Elands Bay, Lambert's Bay, Hondeklip Bay, and Port Nolloth (Cape of Good Hope 1893, Smalberger 1975). North of the Orange River the colony's merchant proconsuls negotiated trading, fishing, and mineral rights with indigenous peoples on the mainland from their guano island redoubts, integrating the Namibian coast into Cape Town's emerging west coast maritime hinterland (Kinahan 1991).

The preindustrial Benguela was controlled by Cape Town merchants through their tenure of key coastal production sites. Stephan Brothers dominated the snoek export fishery through its monopoly over St. Helena Bay and, by 1892, produced two thirds of the colony's dried-fish exports (Cape of Good Hope 1882, 1892). They also controlled the grain trade through their acquisition of shipping sites south of Lambert's Bay and extension of credit to farmers (Cape of Good Hope 1893, 1903, Dooling 1999). The Cape Town–London partnership of De Pass, Spence and Company similarly dominated guano production through its long leases to the Namibian islands and Malagas, paying a paltry rent for the former in return for guano (Van Sittert & Crawford 2003). A host of other, smaller Cape Town merchants traded on the coast in the interstices of these monopolies, particularly in the no-man's-land north of the Orange River.

The late pre-industrial Benguela was thus the product of a *laissez faire* British colonial state, which stimulated commercial production by freeing up natural resources for private enterprise. The state's interest was limited to the revenue from its alienation and leasing of strategically sited coastal land, leaving private merchants to impose their rule over large stretches of the Benguela. For their power, these merchant empires depended on the absence of the state and continued isolation of the region. The mineral revolution in the last quarter of the 19th century was to eventually erode the conditions sustaining this preindustrial Benguela, by stimulating both railway construction and fisheries industrialisation.

The Industrial Epoch (c. 1910–c. 1975)

Around the turn of the 20th century railway construction eventually undermined the merchant monopoly over the southern west coast, by drawing off both labour and grain (Cape of Good Hope 1892, 1903). Stephan Brothers responded by abolishing the fishing lease system, introducing steam shipping and opposing the extension of the railway to Saldanha Bay, but to no avail (Cape of Good Hope 1882, 1903). The German annexation of Namibia in 1884 similarly disrupted Cape Town merchant enterprise north of the Orange River (Angra Pequena and West Coast Claims Joint Commission 1885, Cape of Good Hope 1895). At much the same time, the Cape colonial state refused to renew De Pass, Spence and Company's lease to the Namibian islands, converting guano production into a state enterprise, which was used to sustain flagging wheat yields in the Southwestern Cape with infusions of subsidised manure (Cape of Good Hope 1895). The railway revolution also created a significant local market for fish.

In 1896 the Cape government appointed a marine biologist, John D. Gilchrist, and provided him with a steam trawler, the *Pieter Faure*, with the aim of investigating the fishing potential of the waters around the Cape colony (Brown 1997). Over the next decade Gilchrist successfully prospected trawling grounds between Cape Town and East London and commercial trawling commenced as early as 1900 (Brown 1997). Cape Town's harbour was initially the main base for the new industry, which by 1909 was "exporting" demersal fishes to the neighbouring colonies of the Free State and Transvaal (Cape of Good Hope 1909).

Industrialisation of the Benguela, however, stalled during the interwar decades. The national market remained small, impoverished, and unreceptive to fish, making monopoly and export imperative to protect narrow profit margins in trawling (Union of South Africa 1927, 1934, 1940). Lack of suitable technology also limited the ability of fishers to exploit open-ocean stocks and held overall catches at relatively low levels, at least compared to what was to follow after the Second World War.

The Second World War belatedly transformed inshore fishes from consumer product into industrial raw material, through government contracts and import substitution industrialisation (Van Sittert 1992). Technological advances and a new demand for canned fish also provided an impetus for the development of the fledgling purse-seine fishery, which rapidly expanded to become the largest fishery in the region on a tonnage basis.

Industrialisation required the creation of property rights in marine resources as a basis for private capital investment. This was justified in terms of a need to restrict access to avoid a "tragedy of the commons" and took the form of annually renewable quotas or operating licences vested in the processors, rather than the producers of the fish (Von Bonde 1931, Union of South Africa 1953). On the basis of these rights, first allocated for rock lobster in 1946 and pelagic fishes in 1950, processors floated public companies in which the state, through the Fisheries Development Corporation (FDC), held substantial shares (Van Sittert 1992). The capital invested in inshore fishing more than tripled from £1.1 million in 1944 to £3.7 million by 1947 and was used to modernise vessels, plants and machinery (Skaife 1948). The FDC provided separate capital infusions for housing and boat loans, the former to secure cheap black labour and the latter to create a white boat-owning class in the inshore fisheries (Van Sittert 1992, 2002).

Until the 1950s almost all the catch from the Benguela was taken by locally based fisheries, but in the early 1960s word of the vast hake resources of the region quickly spread and a fleet of Soviet trawlers began to operate off Namibia, soon to be followed by Japanese and Spanish vessels operating farther to the south. As a result, catches quickly rose to unsustainable levels and the stock collapsed. The same fate was soon followed by the pelagic fishery off Namibia, which was decimated in the late 1960s and early 1970s by factory ships operating outside the then 12-mile territorial limit.

The Postindustrial Epoch (c. 1975–c. 2002)

The unrestrained international exploitation of the Southern Benguela that characterised the industrial era was phased out following the establishment of the exclusive economic zone off South Africa in 1977, which returned control of fisheries to local government. However, international exploitation continued in the Northern Benguela until Namibia established her EEZ in 1990. This postindustrial epoch was also marked by the realisation that marine resources were not unlimited and, indeed, require careful management if they are to survive the rapidly increasing capture power of technological fisheries.

More recently, the emphasis has changed from not merely managing the biological stocks in the region in a sustainable manner, but to using marine resources as economic and social tools. This is well articulated within the Marine Living Resources Act of 1988, which sets out, *inter alia*, to "utilise living marine resources to achieve economic growth, human resource development, capacity building within fisheries and mariculture branches, employment creation … "

The result of these processes and policy changes has been a stabilisation of offshore catches and, in some cases, a rebuilding of stocks, at least in the South African sector. However, off Namibia the pelagic fishery has collapsed, although this may be partly a result of poor environmental conditions, as well as poor management. There has also been a dramatic restructuring of the fishing industry, particularly in terms of ownership, in order to "address historical imbalances and to achieve equity within all branches of the fishing industry" (Marine Living Resources Act 1998). Unfortunately, inshore fisheries, which are the most accessible sectors to new entrants to the industry, have proven far more difficult to manage than the capital-intensive offshore sectors. Thus, abalone, rock lobster and linefish sectors in particular remain severely overexploited and require decisive intervention if they are to survive as viable fisheries.

In parallel with developments within conventional fisheries, the postindustrial era has also been characterised by the development of perturbations of the ecosystem that go beyond the conventional extraction of living marine resources. The most obvious of these are the rapid development of mariculture (particularly to replace dwindling wild abalone stocks), the introduction of marine invasive species (which have significantly impacted the structure and function of rocky intertidal communities throughout the region), the development of marine mining, and the increasing manipulation and reduction of freshwater inflows into the system. Also of concern in the longer term is the impact of global climate change, which may ultimately have major implications for the distribution patterns and ecological functioning of the system.

Removal of biomass from the system

Given the data presented above, it is possible to make approximate calculations of the total marine animal biomass removed from the Benguela system over the past 200 yr. In making such calculations, numerical catches have to be converted to wet weights. This has been done using the following conversion factors: each humpback whale has a mass of 24 t and each right whale a mass of 32 t (P. Best, personal communication), while the mass of seal pups is taken as 22.7 kg and of bulls as 150 kg (see seal section, p. 312). A conversion ratio of 1:3 was used to convert the weight of dried fishes back to live weight (Van Sittert, unpublished data). Note that discarded by-catches, particularly from the demersal fishery, which may dump substantial catches of invertebrates and trash fishes, could not be included in these calculations. This may have resulted in significant underestimates of mortality, but not necessarily of actual removal of biomass from the food chain, since trash fishes may be consumed by seabirds, seals, and other predators. In addition, the retained demersal catch is processed at sea and the offal (heads and guts) is discarded at sea, thereby returning an estimated 30,000–46,000 t to the system in 1997 (Walmsley 2004). Illegal and recreational takes are also not included, since no reliable historical data exist for these sectors.

The resulting estimated rate of removal of biomass by decade is depicted in Figure 33 and amounts to a total of over 56 million t of wet animal biomass removed from the system over the past 200 yr. The first few decades on record were marked by intensive and focused exploitation of whale stocks. Perhaps surprisingly the biomass removed from the Benguela at this stage was relatively very small by modern fishery standards, although of course the impact on the targeted stocks themselves was severe. From 1800–1920 the overall mass of catches removed fluctuated from lows of just over 1200 t (in the 1810s and 1820s) to highs of about 28,000 t (in the 1870s and 1880s) — a range of more than 20-fold. These relatively modest catches were certainly not restrained by lack of resources, since stocks of major fish and invertebrate species remained relatively pristine throughout this period. Rather, the fishery was limited by technological constraints, which prevented large-scale exploitation of the major pelagic and demersal species, and also by the restricted market, which remained essentially local.

From 1910 there was a progressive increase in catches in each decade through to the 1960s. This was moderate at first, but became extremely rapid and dramatic in the 1940s–1960s, when the process was driven by rapidly developing fishing technologies and a simultaneous rapid globalisation of markets. This resulted in offtakes well in excess of 1 million t yr^{-1} through the 1960s and 1970s. Such levels were biologically unsustainable and have subsequently declined, as a result of both stock collapses and more conservative and stricter management intervention, to levels of approximately 0.66 million t yr^{-1} in the 1990s (Figure 33).

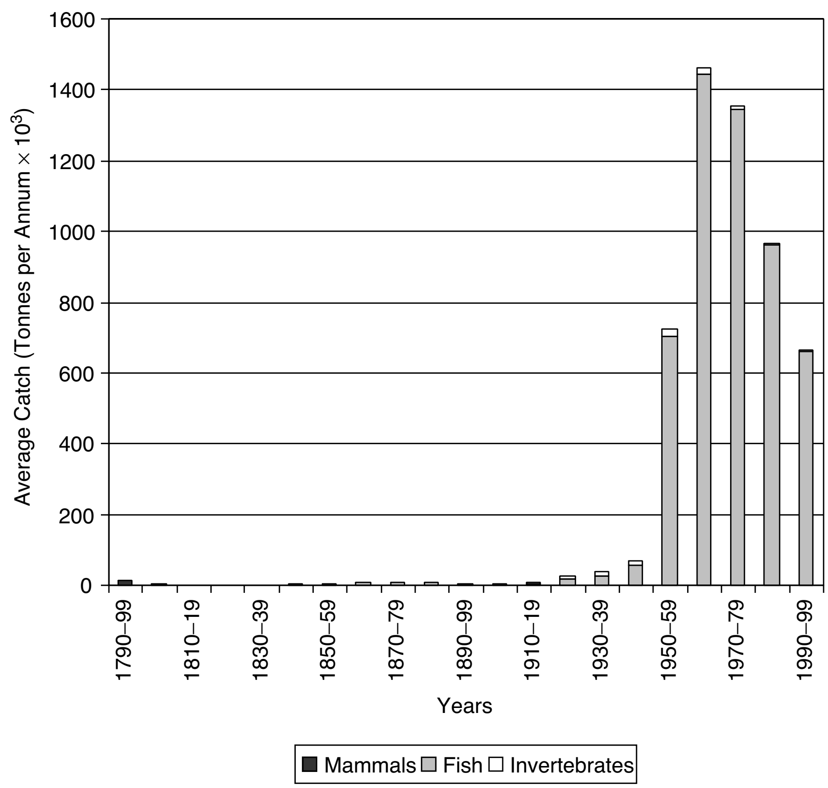

Figure 33 Average annual animal biomass removed from the Benguela system each decade since 1790 (metric tons × 1000); compiled by summing the figures for all the sections covered in this review.

Changes in trophic level

Another way in which temporal catch trends can be analysed is by plotting changes in the trophic level at which the system is fished. Previous analyses of this type have generally shown that the weighted mean trophic level of fishes landed has been in decline in most areas for which sufficient data are available (Pauly et al. 2000). This phenomenon, known as "fishing down of marine food webs," results from the progressive removal of large, slow-growing predators from the system and their replacement in catches by smaller, faster-growing forage fishes and invertebrates.

Figure 34 presents the results of an analysis of this type for the Benguela system. Catches for each stock were taken from earlier sections of this review. The trophic levels of the various species were extracted from Table 4.6 of Shannon (2001), with additional values for invertebrate species taken from Scott (2001).

The results (Figure 34), plotted on a decadal timescale, do indeed show a general declining trend in trophic level fished. From 1790–1900 this decline was only very gradual, as the main target species throughout this period remained large predatory species (e.g., whales, seals and snoek), all of which occupy similar trophic levels of 4.5–4.7. The development and expansion of the trawl, linefish and rock lobster industries in the early 20th century, however, resulted in a more rapid decline in trophic level, to about 4.1. This became precipitous in the period after the Second World War, when enormous catches of pelagic fish (trophic levels of 3–3.6) came to dominate the fishery

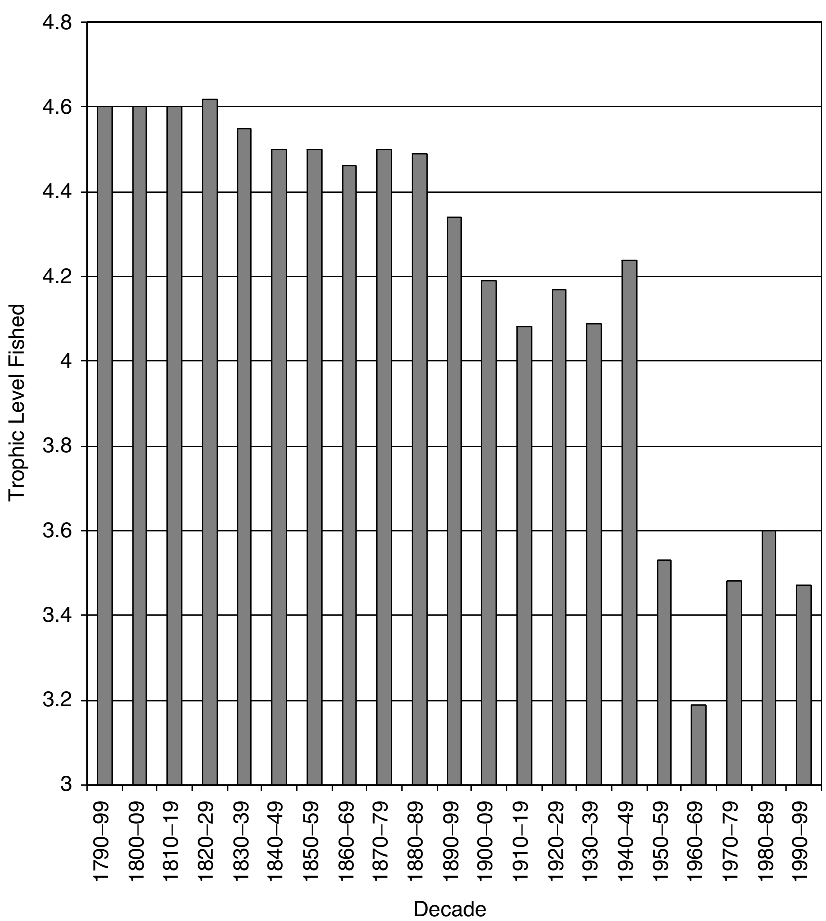

Figure 34 Changes in mean trophic level of the catch removed from the Benguela region since 1800.

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in terms of biomass. Interestingly, the last three decades have seen a gradual increase in trophic level fished, as the proportion of pelagic fishes in the catches have shown a relative decline. However, in the light of recent recoveries in pelagic stocks off South Africa, this trend can be expected to reverse in the decade 2000–10.

Of course, fishing down the food web is not of itself problematic. Indeed, because there is approximately an order of magnitude more energy to be intercepted as one moves each step down the food chain, good justification could be found for deliberately fishing at lower trophic levels, because this potentially increases the yield of animal protein to feed the ever-expanding human population. Rather, there should only be concern where there is a decline in trophic level fished, but catches do not increase as expected (Pauly et al. 2000). These relationships can be visualised by plotting catch landed against trophic level fished (Figure 35). This plot shows that there was a gradual decline in trophic level fished up to 1940, but that this was not accompanied by an equivalent increase in biomass landed. In the 1950s and 1960s an increase of more than an order of magnitude in landed catch was achieved by an equally large decline of almost one unit in trophic level fished, as pelagic stocks came onstream. As these resources collapsed and catches declined during the 1980s, there was little recovery in tropic level fished, indicating that the system was overexploited. The stabilisation in catches and slight decline in trophic level over the past decade indicate a level of exploitation at what are hopefully now sustainable levels.

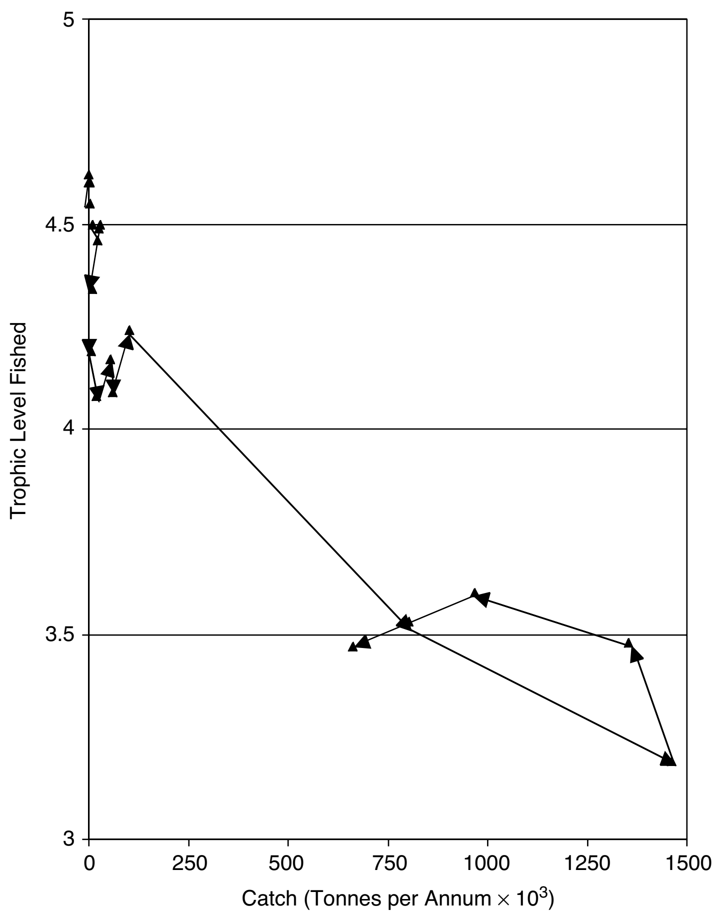

Figure 35 Relationship between trophic level fished and biomass removed from the Benguela system, 1790–2000.

Conclusions

The above review shows the Benguela to have a relatively short history of significant human perturbation, given that low population pressure and primitive technology in the precolonial era probably had minimal effects on the marine ecosystem. This has the fortunate by-product that the history of human impact is relatively well documented and that the original state of the system could be relatively easily reconstructed.

In the early colonial or preindustrial eras the main targets of exploitation were large mammals and line-caught fish, particularly snoek. Being so vulnerable to capture, the mammal populations were quickly depleted, whereas late development of a significant finfishery meant that these resources remained relatively pristine until at least the early 20th century.

The modern industrial finfishery underwent an explosive expansion in the decades after the Second World War, when exploitation clearly exceeded sustainable levels. However, the expansion of territorial waters to include the fishing grounds, together with a more conservative management policy, has subsequently stabilised catches and major stocks now appear to be exploited at sustainable levels. By contrast, management of many inshore resources remain problematic, mainly because of the large number of dispersed users and lack of enforcement of regulations.

Recently, concern is also shifting from management of individual stocks to impacts acting at the ecosystem level, e.g., climate change, invasive aliens and ecosystem effects of fishing, which have received remarkably little attention to date, but could have major effects on future health of the system.

Acknowledgements

Financial support for this review was provided through a grant from the HMAP project, which in turn is funded by the Alfred P. Sloan Foundation. Additional financial support was provided through a research grant to C.L. Griffiths from “The Sea and the Coast II” programme, a joint venture of the South African National Research Foundation and Department of Environmental Affairs and Tourism.

Our thanks to Mr. Barry Rose (Irvin & Johnson Ltd.) for information on the operational history of the trawl fishery and Dr. Hannes Holtzhausen (Namibian Ministry of Fisheries and Marine Resources) for annual handline catches of silver kob off Namibia.

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