Understanding and Mitigating Vulnerable Bycatch in southern African Trawl and Longline Fisheries

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Foreword

The litany of threats facing the world's oceans is well documented. It has been estimated by the FAO that 75% of global fish stocks are exploited unsustainably, approximately 25% of marine resources landed are dumped, ecosystems have been modified and catastrophic declines of vulnerable marine life reported, including the loss of up to 90% of the large predatory fish. Whilst some argue that these declines have been overstated, they nevertheless indicate that the world's oceans are not inexhaustible.
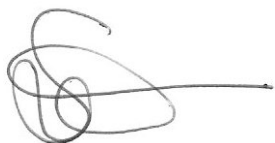
Measures taken in the latter half of the 20th century were insufficient to effectively develop and manage fisheries sustainably. During this period, fisheries were managed almost exclusively on a single species basis and largely assumed to operate in isolation from the rest of the ecosystem. As pressures on resources and ecosystems increased, the shortcomings of this single-species approach became more obvious. Ecosystems are dynamic and inherently uncertain. Through human impacts, ecosystems have been transformed, creating further uncertainty and reducing ecosystem resilience. Adaptive policies and management actions are required that not only satisfy social objectives, but also build ecosystem resilience and provide flexibility to adapt. Although it is important to understand the dynamics of individual populations, as far as practically possible, and account for them in management, it is not sufficient.

An Ecosystem Approach to Fisheries (EAF) management, which balances the diverse needs and values of all who use, enjoy or depend on the ocean now and in the future, is accepted as the preferred manner of managing fisheries (Reykjavik Declaration 2001) and there is now global resolve to reduce the ecosystem impacts of fisheries. This philosophy is firmly entrenched in various international legal instruments and policy statements. It is perhaps most aptly illustrated in the 2002 World Summit on Sustainable Development's Plan of Implementation, which urged states to apply an Ecosystem Approach to Fisheries by 2010. Locally, and in the context of this report, incorporation of ecosystem considerations underpins the South African Marine Living Resources Act (No. 18 of 1998) and the Namibian Marine Resources Act (No. 27 of 2000). Furthermore, the development of the Benguela Current Commission (BCC), which allows for greater harmonisation of the management of marine resources between the national jurisdictions of Namibia, South Africa and Angola and their commitments to the World Summit on Sustainable Development's implementation plan, bodes well for the effective implementation of an EAF in the region.

The challenge is great. Relatively little is understood about the ecosystem impacts of fisheries, and in many cases few data are available to progress our understanding. Fishing methods are not 100% selective, often catching many other species besides the target. Some of these non-target species, including threatened species of turtles, seabirds and sharks, have K-selected life history strategies making them vulnerable to even small

increases in adult mortality. Marine ecosystems are complex, dynamic systems containing inter-connected food webs which, if disrupted frequently, result in seemingly unpredictable changes. Implementing an Ecosystem Approach to Fisheries also brings with it the challenge of balancing multiple objectives and uses of marine ecosystems. Consequently individual issues cannot be addressed independently. Attempts to manage any one issue in isolation are likely to impact on other issues, and it is therefore necessary to address all objectives together as far as is practical.

This report aims to further our understanding of ecosystem impacts of longline and trawl fisheries in southern Africa, particularly the bycatch of seabirds, turtles and sharks. In the past, conservation biologists have tended to focus their research on a single taxon. We believe the strength of this work is that it addresses the issue in a holistic manner. It also acknowledges that seabirds, turtles and sharks undertake movements exceeding South Africa's national Exclusive Economic Zone (EEZ) and therefore where possible were evaluated as such. Finally, central to the ecosystem approach is the concept that people do not operate outside of natural systems and that if we hope to invoke the wise management of our oceans then the sustainability of fishers' livelihoods also needs to be kept in mind.

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Contents

Executive Summary

Over the past decade there has been concern about the bycatch of seabirds, turtles and sharks in fishing operations, in particular longline and trawl fisheries. The incidental mortality of these species has been widely held responsible for the declining populations and threatened conservation status of several species. These vulnerable, K-selected species are also top predators and as such play an important role in the functioning of marine ecosystems. They also have an economic value both in terms of non-consumptive eco-tourism activities and, at least in the case of sharks, a consumptive value. Seabirds and turtles are also indicators of the health of the ecosystem because they breed on land and their populations can therefore be accurately monitored. This report addresses the real world problem of understanding, managing and ultimately reducing the impacts of longline (targeting tunas *Thunnus* spp., Swordfish *Xiphias gladius* and Cape hakes *Merluccius* spp. and trawl (targeting Cape hakes) fishing operations.

We estimate that 21 000 seabirds (0.44/1000 hooks) are caught incidentally by South African longline and trawl fisheries, including eight endangered seabirds and one endemic species. A similar suite of seabirds is also caught in the Namibian hake longline fishery (crudely estimated to kill 20 000 birds per year). Key findings of this study confirmed that birds are most at risk in the southern region of South Africa's west coast, in the vicinity of Cape Point and on the Agulhas Bank during the winter months June–October. Full moon and daytime sets were strong predictors of mortality in the longline fishery. Evaluation of observer data

(1998–2005) confirm that Swordfish catch rates are greatest when setting takes place at dusk. Optimal line sink rates of 0.3 m.s^{-1} are a requirement of the South African longline fishery yet gear configurations to achieve this sink rate have not been established. Five gear configurations were investigated: the American longline system using no weighted swivel, 60 g and 120 g weighted swivels, the use of a wire trace and the Asian pelagic longline system. The fastest line sink rates were achieved by the addition of 120 g weighted swivel (0.35 m.s^{-1}) however the relative improvement from 60 g (0.24 m.s^{-1}) to 120 g may not warrant the additional cost and may further compromise crew safety. Despite achieving the optimal rate of 0.3 m.s^{-1} on average, no weighting regime achieved this rate consistently. Fishing effort could be limited during full moon when catch rates are known to be substantially higher. Although there is evidence elsewhere that Swordfish catch rates increase over full moon, there was no significant effect of moon phase on in catch rates in this fishery.

Mitigation measures to reduce the incidental mortality of seabirds in trawl fisheries include improved discard management to reduce the attraction of seabirds to vessels. This report explored the implications of reducing the availability of fisheries discards for seabirds by understanding foraging patterns of Black-browed Albatrosses *Thalassarche melanophrys* and White-capped Albatrosses *T. steadi* through the use of satellite transmitters. Trawl activity on the continental shelf break off South Africa provides large quantities of high quality and predictable food in the form of offal and discards for a range of species,

including non-breeding Black-browed Albatrosses and White-capped Albatrosses. This study presents evidence that Black-browed Albatrosses, in particular, forage to a large extent on natural prey, despite the high availability of discards from fishing vessels in the Benguela. Therefore, given the high incidence of albatross collisions with trawl cables, the benefit of a management decision to limit discarding as a mitigation measure is likely to far outweigh the disadvantage of reduced food in the form of fisheries waste.

Four species of Endangered turtle were caught by the South African pelagic longline fishery (0.04/1000 hooks, estimated annual mortality 165 turtles). This mortality may explain the lower than expected recovery rates of protected South African Leatherback *Dermochelys coriacea* and Loggerhead *Caretta caretta* populations since 1975. This conclusion was supported by the high degree of overlap of fishing effort and Leatherback Turtle movements recorded by satellite transmitters. Turtle catches occurred predominantly on the Walvis Ridge and the Agulhas Bank, February to June for Loggerheads and all year for Leatherbacks. Most turtles caught by the large pelagic longline fishery were alive and therefore mitigation should focus on equipping vessels with de-hooking equipment and skippers with the necessary skills to release turtles. Implementing the use of circle hooks could be considered for the swordfish directed fishery which caught 97% and 82% of Loggerhead and Leatherback Turtles. Preliminary results suggest that shark catches may be increased with the use of circle hooks, but most were hooked in the jaw which is likely to increase post-release survival should they be released.

Twenty species of pelagic sharks are caught each year by the South African pelagic longline fishery, seven of which are threatened. Blue *Prionace glauca* and Short-finned Mako *Isurus oxyrinchus* Sharks were caught at a rate of 11.7 and 1.8/1000 hooks and 1.4, and 0.9/1000 hooks by the tuna and Swordfish sectors, respectively. A decreasing length frequency and CPUE was observed for both species. Blue and Mako Sharks were predominantly caught on the west coast, south to Cape Point. The distribution of Mako Sharks catches extended to the Agulhas Bank. In general, Mako Sharks were found further inshore compared with Blue Sharks. Catches on the west coast were dominated by smaller animals and may indicate the presence of nursery grounds. The management of shark catches has been focused on incentives to release live sharks rather than on mitigation to avoid shark capture. South African legislation requires fishers to retain shark trunks in an 8% fin to trunk ratio, which is higher than the globally accepted norm of 5%. Data collected in South African harbours confirm that this higher value may be appropriate for the South African fishery. However, given the re-introduction in 2005 of Asian vessels, which process sharks differently to South African operators the validity of a fin to trunk ratio for this fleet should be investigated. A further measure to limit shark catches in South African waters is the adoption of a 10% shark bycatch limit which was found not to be practical for the Swordfish directed sector where sharks typically comprise approximately 22% of the catch.

Thirty species of demersal sharks and skates from 17 genera and eight families were caught as bycatch in the South African demersal longline fishery for hake. The four most commonly caught species, *Squalus*

mitsukurii, *Holohalaelurus regain*, *Scyliorhinus capensis* and *Raja straeleni*, were also caught in the demersal trawl fishery operating in a similar area. The overall catch rate of these four species was estimated to be 10.50, 2.19, 0.46 and 1.46 sharks per 1000 hooks and 68.3 kg.nm^{-2}, 54.3 kg.nm^{-2}, 12.6 kg.nm^{-2} and 358.1 kg.nm^{-2} for longline and trawl fisheries, respectively. There was no evidence of a decline in the biomass index, calculated from annual survey data from 1986 to 2007, for *S. mitsukurii* and *H. regani*. However, the biomass index of *S. capensis* and *R. straeleni* decreased by 44% and 69% on the west coast and by 50% and 65% on the south coast from 2000 to 2007. Most demersal sharks and skates caught by longline and trawl fisheries are discarded. Animals caught by the longline fishery are frequently alive on capture and fishers should be required to release unwanted live chondrichthyan bycatch rapidly. This thesis highlights the possibility of altering demersal longline gear configurations to reduce the catches, of at least the four most commonly caught species of demersal sharks and skates, by removing the hooks in close proximity to weights in order to reduce the numbers of hooks on or close to the seabed. Prior to the introduction of the longline fishery in 1998, untrawlable, rocky grounds were de facto protected, providing a refuge for many demersal species. Now that these refugia no longer exist, the use of closed areas should be investigated.

Time and area closures have been used conservatively as fishery management tools in the past, mainly because fisheries managers have tended to focus their attention on abundant species with high biological productivity, whereas closures are a more essential management tool for managing less abundant species with low biological productivity such as seabirds, turtles and sharks. A single closed season is unlikely to adequately protect all three groups because very little overlap exists in the seasonal distribution of their catches. There is however, considerable spatial overlap in the distribution of seabird, turtle and shark catches, particularly on the continental shelf off the west and south coasts in the vicinity of Cape Point. This thesis provides a first attempt at overlaying the spatial distribution of catches and through a conservation planning exercise using MARXAN software, identified areas where bycatch was the highest and target catches are the lowest.

Resource managers are often faced with the challenge of balancing conflicting objectives. Implementing an EAF in South African longline and trawl fisheries involves the integration of social, economic, and ecological goals. This is achievable if one does not dominate at the expense of the others. This thesis has gone some way to further our understanding and identify possible solutions taking other vulnerable species and target catches into consideration. What is now needed to save these species from extinction is action.

General Introduction

1. General Introduction

INTRODUCING THE BYCATCH PROBLEM

Since the 1990s there has been global concern about the bycatch of seabirds, turtles and sharks in fishing operations, in particular longline and trawl fisheries (Brothers 1991, Bergin 1997, Croxall & Gales 1998, Witzell 1999, Nel *et al*. 2002, Myers & Worm 2003, Lewison *et al*. 2004, Barker & Schluessel 2005, Sullivan *et al*. 2006, BirdLife International 2007). The incidental mortality of these species has been widely held responsible for the declining populations and threatened conservation status of several species (Lewison *et al*. 2004, Lack & Sant 2006, BirdLife International 2007). These three groups of animals are also top predators and as such are likely to play an important role in the ecosystem (Beddington 1984, Stevens *et al*. 2000, Baum & Myers 2004, Megalofonou *et al*. 2005). They also have an economic value both in terms of non-consumptive eco-tourism activities (Yorio *et al*. 2001, Garrod & Wilson 2003, Topelko & Dearden 2005) and, at least in the case of sharks, a consumptive value (Barker & Schluessel 2005). Because they breed on land and their populations can therefore be accurately monitored, seabirds and turtles are also indicators of the health of the ecosystem (Cherel & Weimerskrich 1995, Best *et al*. 1997).

The Benguela Upwelling System is one of the world's most productive systems, attracting millions of top predators such as seabirds, turtles and sharks (Shannon & Field 1985, Best *et al.* 1997). Many of these species travel thousands of kilometres, sometimes across oceans, to feed in its nutrient rich waters (Weimerskirch *et al*. 1999, Baker *et al*. 2007, Fretey *et al*. 2007). Not surprisingly, the Benguela Upwelling System also supports several large commercial fisheries operating within countries' Exclusive Economic Zone (EEZ) as well as on the high seas (Sauer *et al*. 2003). The spatial overlap of large numbers of top predators and large commercial fisheries in a confined area has the potential to lead to high and unsustainable catches of threatened species.

Southern African waters are of global importance for conserving seabirds, turtles and sharks. The coastal waters are a rich foraging area for 15 of the 24 albatross and petrel species threatened with extinction, mainly as a result of longline fishing operations (Nel & Taylor 2002, BirdLife International 2007). The oceanic and inshore waters surrounding Southern Africa are also frequented by five of the seven species of endangered turtle and 36 species of threatened sharks (Hughes 1989, IUCN 2007). Nineteen of these species are threatened by fishing operations, with longlining a known threat for at least eight species (IUCN 2007).

Seabird bycatch in demersal and pelagic longline fisheries have previously been evaluated in South Africa. Ryan *et al*. (2002) estimated that between 19 000 and 30 000 seabirds were killed per year by the South African pelagic longline fishery during 1998–2000. Barnes *et al*. (1997) evaluated seabird bycatch in the South African demersal longline fishery and estimated that approximately 8 000 White-chinned Petrels were killed in 1995. In general, these studies were based on limited sample sizes (108 and 12 sets, respectively) collected

over short periods of time (Barnes *et al.* 1997, Ryan *et al.* 2002). There are no published studies of turtle or shark bycatch in South African longline fisheries, or any of the three groups in Namibian fisheries. The Worldwide Fund for Nature (WWF) and BirdLife undertook a project to evaluate the bycatch of seabirds, turtles and sharks in the Benguela, including South Africa (east of 20 °E), Namibia and Angola (Petersen *et al.* 2007). There are also no published studies evaluating seabird bycatch in trawl fisheries in the region.

For seabirds, and to a lesser extent, for turtles, effective and relatively inexpensive methods of reducing the number of animals killed in these fishing operations have been developed (Alexander *et al.* 1997, FAO 1999, Melvin & Robertson 2000, Melvin *et al.* 2004, Watson *et al.* 2005, Read 2007). Although solutions to reduce shark capture have not yet been identified, there is equal concern for the viability of their harvest given their vulnerable life history characteristics (Musick *et al.* 2000, Barker & Schluessel 2005). No mitigation experiments have previously been undertaken in the Benguela. Demonstrating the efficacy of measures under local conditions is important because conditions vary among regions, fisheries and fleets (Gilman 2006). Demonstration also helps to facilitate implementation. In some cases, mitigation measures have been adopted out of necessity to reduce the capture of vulnerable species, but have not as yet been adequately or appropriately understood. Previously mitigation measures have largely been developed on a single taxon basis. This has the shortcoming that what might solve a seabird problem, for instance, could negatively affect shark populations or fisher's livelihoods by decreasing target catches. Mitigation experiments undertaken in this report evaluate the effect on target and other non-target species.

INTRODUCTION TO THE FISHERIES EVALUATED IN THIS REPORT

Southern Africa supports demersal and large pelagic fisheries within the continental EEZ of South Africa and Namibia: a demersal longline and trawl fishery targeting Cape hakes *Merluccius* spp. and pelagic longline fisheries targeting tuna *Thunnus* spp. and Swordfish *Xiphias gladius* (Table 1). No additional data were available for the Namibian large pelagic sector, which was therefore omitted from this thesis. Available information is summarized in Petersen *et al.* (2007). There is also a trawl fishery for Cape hakes in Namibia, but no data are available on its impacts on vulnerable species.

Table 1: Summary of South African and Namibian fisheries evaluated in this report.

| | **SOUTH AFRICA** | | | **NAMIBIA** |
	Large Pelagic Longline	**Demersal Longline**	**Demersal Trawl**	**Demersal Longline**
Target species	Tunas and swordfish	Cape hakes	Cape hakes	Cape hakes
Fishery started	1960	1994	1890	1991
Trip duration	SA: 15 days; Asian: 45 days	6 days	6 days	6 days
TAC/TAE in 2005	50 permits	135 000 mt		200 000 mt
Active vessels in 2005	26	*c.* 50	79	25
Fishing effort in 2005	4 million hooks	42 million hooks	60 000 trawls	120 million hooks

South African pelagic longline fishery

The earliest record of a South African domestic pelagic longline fishery dates back to the early 1960s (Cooper & Ryan 2003, Table 1). This fishery predominantly targeted Albacore *Thunnus alalunga*, Southern Bluefin *T. maccoyii* and Bigeye *T. obesus* Tunas (Cooper & Ryan 2003). Effort in the domestic fishery waned in the mid 1960s. Thereafter, pelagic fishing effort was largely conducted by Japanese and Taiwanese vessels under bilateral agreements with South Africa. These Asian vessels set their gear relatively deeply, frequently during the day, seldom used lightsticks and primarily targeted tuna species. Their fishing effort accounted for 96% of the *c.* 12 million hooks set annually within the South African EEZ during 1998-2000 (Ryan & Boix-Hinzen 1998, Ryan *et al*. 2002). In 1995, a permit was issued to conduct a joint venture operation between a South African and Japanese vessel. This joint venture showed that tunas and Swordfish *Xiphias gladius* could be exploited profitably in South African waters and consequently 30 experimental permits were issued in 1997 to South African flagged vessels. Vessels targeting Swordfish typically use the American longline system, set their gear relatively shallow, use lightsticks and set their lines primarily at night.

A policy decision was made in 2004 to expand and "South Africanise" the South African large pelagic longline fishery (DEAT 2004, 2005a, 2007). This process commenced in 2002 when all foreign licences to target tunas and Swordfish in South African waters were revoked (DEAT 2004). This resulted in a smaller domestic fishery operating in South Africa's EEZ. The domestic fishery was developed in 2004 when 50 (20 swordfish directed and 30 tuna directed) commercial fishing rights were made available for allocation (DEAT 2004, 2005a, 2007). The motivation for this expansion was to improve South Africa's catch history and thereby motivate for larger country allocations at Regional Fisheries Management Organisations (RFMOs), such as the International Convention for the Conservation of Atlantic Tunas (ICCAT) (South Africa is a member) and Indian Ocean Tuna Commission (IOTC) (South Africa is not a member, but a co-operating party) (DEAT 2004, 2005a, 2007). Since South Africa is not traditionally a tuna fishing nation, foreign flagged vessels were once again allowed into the fishery in 2005, under joint venture agreements, on the following basis: a) South Africanisation and transformation would occur through a step-wise increase in employment of local crew, b) that skills would be transferred to South African fishers and c) that all foreign flagged vessels would re-flag to South Africa within a period of one year (DEAT 2004, 2005a, 2007).

Furthermore, the government's policy for the allocation of long term rights in the large pelagic fishery states that the targeting of pelagic sharks would be phased out as a precautionary management decision based on the likelihood of increased shark catches as bycatch in the developing large pelagic fishery. Given the susceptibility of shark species to over-exploitation due to their biology, the global under-reporting and illegal trade of sharks it was found prudent to not have two fisheries exploiting these species (DEAT 2004, 2005a, 2007).

This fishery currently operates out of Cape Town, Durban and Richards Bay (Sauer *et al*. 2003, Fig 1). South African vessels typically undertake trips of 15 days and Asian vessels of 45 days duration (Table 1). Fishing takes place predominantly on the continental shelf along the west coast and on the Agulhas Bank (Chapter 1). Average annual fishing effort in 2005 was approximately 4 million hooks (Chapter 1, Table 1).
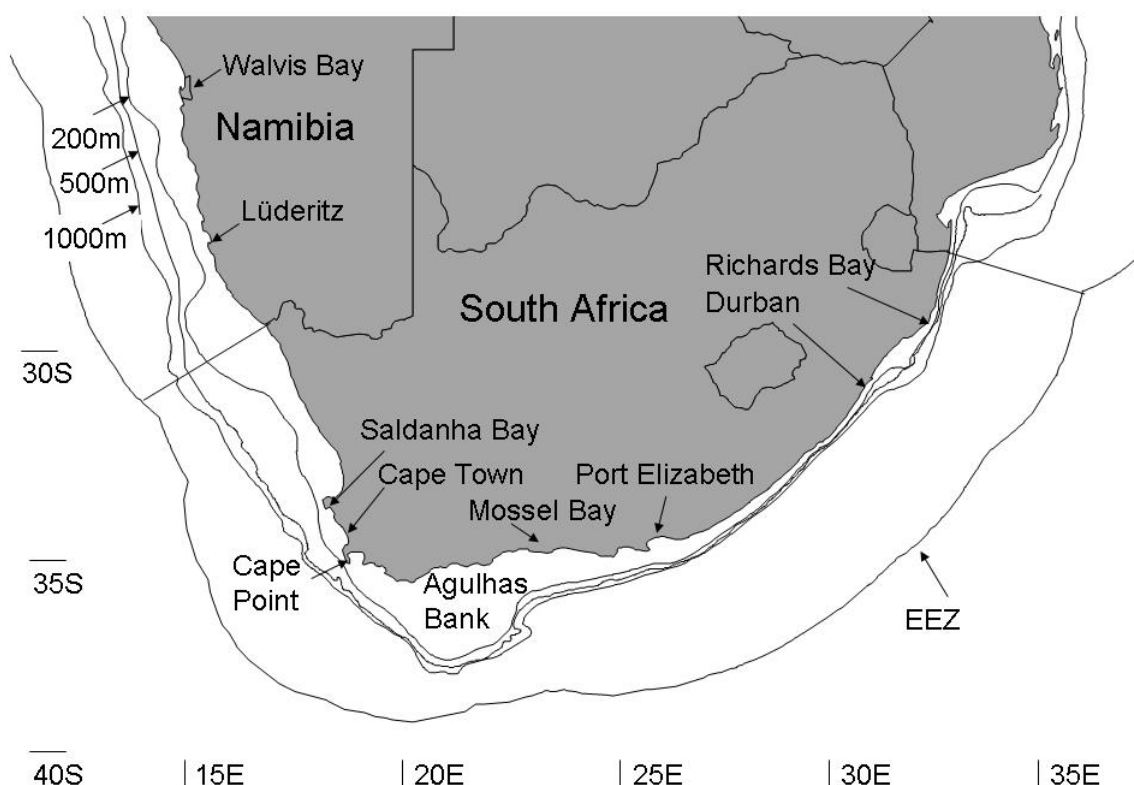

Figure 1: Map of South Africa's and southern Namibia's Exclusive Economic Zones, in relation to the 200 m, 500 m and 1 000 m isobaths.

South African demersal longline fishery

An experimental demersal longline fishery targeting Kingklip *Genypterus capensis* in the continental shelf waters around South Africa was initiated in 1983 (Japp 1993). Due to concern over the sustainability of the Kingklip resource the fishery was closed in 1990. In 1994, a five-year experimental longline fishery directed at Cape hakes *Merluccius capensis* (mainly inshore) and *M. paradoxus* (mainly offshore) was started (Table 1). During this period the number of active vessels varied between 32 and 71 (Japp 1993, Japp & Wissema 1999). This fishery operates out of Cape Town, Mossel Bay and Port Elizabeth (Fig. 1) and typically undertakes trips of approximately six days in duration (Table 1). In 1998, this fishery became commercial and has remained so until the present (Cooper & Ryan 2003). Fishing typically takes place on the continental shelf along the western and southern coasts in depths of 100–

600 m (Japp 1993, Japp & Wissema 1999). In 2007, the Total Allowable Catch (TAC) for hakes was 135 000 mt, divided between the trawl (90%), longline (6.6%) and handline (3.4%) sectors (Butterworth & Rademeyer 2005, DEAT 2005b).

Namibian demersal longline fishery

The hake longline fishery started in Namibia in 1991 (Table 1). The fleet initially comprised 11 vessels, but had grown to 25 vessels by 2007 (Voges 2005). Fishing takes place mainly between 19 °S and 30 °S, at depths of 200 to 600 m (average 330 m) (Voges 2005). This fishery operates out of Walvis Bay and Luderitz (Fig. 1) and typically undertakes trips of approximately six days in duration (Table 1). In 2007, six rights holders shared the hake longline quota. The annual quota for hake longliners is approximately 6% of the hake TAC of 200 000 mt (Butterworth & Rademayer 2005).

South African demersal trawl fishery

The demersal hake trawl fishery is the most valuable fishery in South Africa (FAO 2001). There are two sectors within this fishery: an offshore, deep-sea sector and an inshore sector (Payne 1989). Only the offshore sector was evaluated in this report because seabird bycatch is considered to be negligible in the inshore sector. The offshore trawl fishery started in the 1890s, mainly targeting Agulhas *Austroglossus pectoralis* and West Coast *A. microlepis* sole (Payne 1989). The development of the fishery can be broadly categorized into three periods (Sauer *et al*. 2003). Prior to 1977, most entrants into the fishery failed, mainly because they were reluctant to invest in processing and distribution. In the mid 1940s, annual catches were 1 000 t, which increased to 50 000 t by 1950 and had reached 115 000 t by 1955 (Sauer *et al*. 2003). During the 1960s, foreign vessels entered the fishery, escalating catches to a million tonnes per year. The International Commission for the Southeast Atlantic Fisheries (ICSEAF) was established in 1972, to investigate and control the international fisheries for hake off South Africa and Namibia (Sauer *et al*. 2003). Most hake caught were juvenile fish and in 1975 the minimum mesh size was increased from 102 to 110 mm. Between 1977 and 1992 the stocks collapsed. South Africa declared its 200 nautical mile (nm) EEZ in 1977 which reduced the number of foreign trawlers operating in South African waters by 25% (Sauer *et al*. 2003). Individual quotas were first granted in 1979, the bulk being allocated to the two major companies. In 1985 a policy was introduced to broaden access to the fishery, resulting in the number of participants increasing from seven in 1986 to 21 in 1992. Post 1992 saw major changes in quota allocations and the entry of new participants from previously disadvantaged communities (Sauer *et al*. 2003). The number of participants in the deep-water sector increased to 56 in 2000. In 2005 there were 79 vessels in the fleet which undertook approximately 60 000 trawls. The CPUE decreased fourfold from 1955 to 1997 (Sauer *et al*. 2003). Vessels operate out of Cape Town and Saldanha Bay (near Cape Columbine) and typically undertake 6 day trips.

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Understanding bycatch

Chapter 1

SEABIRD BYCATCH IN THE PELAGIC LONGLINE FISHERY OFF SOUTHERN AFRICA

SEABIRD BYCATCH IN THE PELAGIC LONGLINE FISHERY OFF SOUTHERN AFRICA

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ABSTRACT

The waters surrounding South Africa provide rich foraging opportunities for pelagic seabirds. They also support a pelagic longline fleet targeting tunas *Thunnus* spp. and Swordfish *Xiphias gladius*, which set a total of 41.5 million (average 5.2 million per year) and 10.2 million hooks (average 1.3 million per year), 1998–2005, respectively. Fisheries observers collected seabird bycatch data from 2 256 sets (4.4 million hooks) and recorded a total of 1 954 birds killed during this time. Eleven species of seabird have been confirmed incidentally caught by this fishery, eight of which are considered threatened. Birds were caught at an average rate of 0.44/1000 hooks, resulting in an average of 2 900 birds killed per year decreasing from approximately 5 900 in 1998 to 1 800 in 2005. Three techniques for extrapolating total seabird mortality were investigated and little difference between estimates found. White-chinned Petrels *Procellaria aequinoctialis* were caught most commonly (68.9%) at a rate of 0.30/1000 hooks (1 650 killed each year). Albatrosses made up 30.3% of the bycatch or 0.14/1000 hooks. Three species were recorded in significant numbers: shy-type (mostly White-capped *Thalassarche steadi*) (0.09/1000 hooks, 600 per year), Black-browed *T. melanophrys* (0.02/1000 hooks, 125 per year) and Indian Yellow-nosed Albatrosses *T. carteri* (0.01/1000 hooks, 85 per year). Generalised linear models were used to explain bycatch patterns and revealed that individual vessel is the most important explanatory variable, followed by vessel flag, moon phase, season, sea state, the use of a tori line, time of set, area and bathymetry. Most birds (88%) were caught by Asian flagged tuna directed vessels (72% of albatrosses and 97% of petrels). Asian tuna directed vessels caught seabirds at a rate of 0.51/1000 hooks (0.58/1000 hooks in winter and 0.14/1000 hooks in summer) compared to South African swordfish directed vessels which caught seabirds at a rate of 0.23/1000 hooks (0.22/1000 hooks in winter and 0.24/1000 hooks in summer). More birds were caught during full moon (1.07/1000 hooks) compared to new moon (0.09/1000 hooks). Albatrosses were mainly caught on the Agulhas Bank and along the continental shelf, especially in the Atlantic Ocean. Petrels, especially White-chinned Petrels, were caught on the Agulhas Bank, but had a higher catch rate along the east coast of South Africa. Although there were subtle differences between species, all species were more likely to be caught in the austral winter and spring (June to October). Estimates of the numbers of birds killed per year are lower than previous studies. The improvement was most likely linked to the termination of the foreign bilateral agreements, as well as improved awareness among fishers linked to ongoing education campaigns. Some of the apparent decrease in catch rate could reflect reduced numbers of birds at sea, as a result of ongoing population decreases in several key species.

INTRODUCTION

Traditionally, fisheries have been managed in terms of their impact on the target species only. Since 2001, there has been a paradigm shift to an Ecosystem Approach to Fisheries (EAF) Management (Reykjavik Declaration 2001, FAO 2003) which recognizes the need for a holistic, ecological approach. This approach considers both the impacts of fisheries on target and non-target species, as well as the direct and indirect ecosystem effects of fishing operations, such as the trickle down effect of removing a top predator (Cochrane *et al.* 2004, Shannon *et al.* 2004). A major ecosystem impact of longline fishing is the incidental mortality of seabirds (Brothers 1991, Bergin 1997, Croxall & Gales 1998, Nel *et al.* 2002, BirdLife International 2007). This has lead to population decreases of many species, especially albatrosses and some large petrels (Croxall & Gales 1998, Nel *et al.* 2002, BirdLife International 2007). The fact that bycatch events are relatively rare complicates perceptions regarding the need for conservation (Robertson 1998). Fishers, who are accustomed to catching less vulnerable, r-selected species with high reproductive rates (often producing 100s to 10 000 eggs per individual per year) perceive the relatively low catch rates of seabirds as insignificant (Robertson 1998). However, given the vulnerability of K-selected species, such as albatrosses and petrels, which display conservative life history characteristics (Warham 1996), even relatively low catch rates of adults can result in population declines (Croxall & Gales 1998, Gales 1998).

South African waters are of prime importance for conserving seabirds because the Benguela Upwelling System and the Agulhas Bank provide rich foraging opportunities for a wide diversity of seabirds, including the offal discards from fishing vessels. Fifteen of the 24 species of albatross and petrel threatened with extinction, mainly as a result of fishing operations, forage in South African waters (Abrams 1983, 1985, Ryan & Moloney 1988, Nel & Taylor 2002, BirdLife International 2007). Of the species endemic to the Benguela, fishery discards make up an important component of Cape Gannet *Morus capensis* diet (especially in winter) making this species vulnerable to incidental mortality by fisheries (Berruti *et al.* 1993, Pichegru *et al.* 2007). In general, the abundance of pelagic seabirds is greatest on the continental shelf off South Africa's south west coast and decreases in a northerly direction (Crawford *et al.* 1991, Crawford *et al.* 2007).

Seabird bycatch in the pelagic fishery during 1998–2000 was estimated at 1.60/1000 hooks (Ryan *et al.* 2002) when primarily Asian vessels were operating on the western Agulhas Bank. Asian vessels caught 2.6 (range 0.1–5.4)/1000 hooks whereas South African vessels caught birds at a rate of 0.8 (0.0–4.3)/1000 hooks (Ryan *et al.* 2002). Based on these rates it was estimated that this fishery killed between 19 000 and 30 000 seabirds per year (Ryan *et al.* 2002). The termination of the bilateral agreement with foreign longliners led to a fundamental shift in the nature of the fishery, hence the need for an update on these previous estimates. Furthermore, these estimates were based on limited data (interviews and

108 sets or 143 000 hooks observed) collected over relatively short time periods (one or two years of data collection) (Ryan & Boix-Hinzen 1998, Ryan *et al.* 2002).

This study presents a comprehensive assessment of seabird bycatch in the pelagic longline fishery from 1998 to 2005 and provides recommendations for a way forward to reduce this incidental mortality. It also summarises seabird bycatch in the south eastern Atlantic and the south western Indian Oceans, which is important for regional fisheries management organizations.

METHODS

Data were collected by independent fishery observers on board pelagic longline vessels operating in the South African fishery from 1998 to 2005. This information included seabird bycatch information (species, number and status), as well as gear (e.g. number of hooks, length of mainline, etc.) and operational information (time of set, position, etc). Observers also recorded whether a tori line was used, but did not record the specifications of the tori line. These vessels carried rights to fish within South Africa's EEZ as well as on the high seas. Observers received seabird identification training and were equipped with an identification manual at sea in 2004 and 2005 (Petersen & Honig 2005). Birds killed were supposed to have been frozen and stored; however, for logistical (operational) reasons, such as lack of storage space on vessels, 39% were not brought to port. The remainder were identified to the species level, and were aged and sexed, by inspection of gonads. Post-mortem fading of bill colour prevented accurate identification of giant petrels. A sample (n=24) of shy-type albatrosses (Shy *Thalassarche cauta* and White-capped *T. steadi*) were identified using molecular markers, and 95% were found to be White-capped Albatrosses (Abbott *et al.* 2006, Baker *et al*. 2007a). However, not all samples could be identified, and are thus referred to as shy-type albatrosses throughout. Only 43% of birds returned to port could be linked to observer records due to poor labelling or lost labels enabling the verification of species identification for these birds. Where observers had incorrectly identified a bird, the identifications were corrected. These corrected identifications were used for analysis.

In cases when only heads were retained, most species could not be sexed with confidence and were excluded from the analysis of sex ratios. Albatrosses and giant petrels were aged based on plumage and bill characters; other petrels could be divided into fully grown and recently fledged birds based on moult scarring on the bill plates. Chi-square (χ^2) goodness of fit test (with Yates connection for continuity) was used to test for deviations from an assumed 50:50 sex ratio.

Catch rates are reported as numbers of birds per 1000 hooks summarised by vessel, region, season, flag, moon phase and with/without tori lines (a mandatory mitigation measure) (Cooper & Ryan 2003). The three techniques used to calculate total catch were 1) basic extrapolation of total fishing effort multiplied by an average annual catch rate, 2) building on

the basic extrapolation to incorporate the effect of vessel flag and 3) adding the effect of season and area. This was done based on the method used by Lewison and Crowder (2003):

$$\hat{C}b = \sum rsf \, (Cbrsf \, / \, Eorsf) * Edrsf$$

where $\hat{C}b$ = *Estimated total bycatch of a species, b*
$Cbrsf$ = *Observed bycatch of a species, b within region, r, season, s and flag, f*
$Edrsf$ = *Number of hooks deployed, d, within region, r, season, s and flag, f*
$Eorsf$ = *Number of hooks observed, o, within region, r, season, s and flag, f*
b = *Bycatch species or group of species*
r = *Region (1° grid cell)*
s = *Season i.e. summer = November–April, winter = May–October*
f = *Flag i.e. Asian or South African.*

Vessel flag was used as a proxy for target species, operation and gear configuration because these differed between South African flagged vessels which target Swordfish with shallow set gear (branch line are on average 20 m long) and light sticks compared to Asian flagged vessels which target tunas with deeper set gear (branch line are on average 40 m long and typically use a line setter) without light sticks.

Arcview GIS 3.3 was used to provide spatial representations of information. Counts of the numbers of seabirds (total and those birds with sufficient data to model separately, namely White-chinned Petrel *Procellaria aequinoctialis* and shy-type albatross and Black-browed Albatross *T. melanophrys*) killed by each longline set were modelled using a generalised linear model with a Poisson distribution and logarithmic link function (McCullagh & Nelder 1989). Genstat 9 (Genstat Committee 2007) was used for model fitting and Akaike's Information Criterion (AIC) was used to guide model selection (Quinn & Keough 2002). The logarithm of the parameter λ of a Poisson distribution was modelled as a linear combination of explanatory variables: e.g. for three explanatory variables, $\log \lambda = a + b_1x_1 + b_2x_2 + b_3x_3$. Explanatory variables investigated included year (January–December), season, area, vessel name, vessel flag (South African or Asian), moon phase (eight phases), branch line length (in metres), bathymetry, use of a tori line, offal discarding practices (whether offal was discarded on the opposite side to hauling or not), Beaufort scale (range 0–8) and time of set. In the case of season, four seasons i.e. summer (December–February), autumn (March–May), winter (June–August), spring (September–November) and two seasons i.e. summer (November–April) and winter (May–October) were investigated. Furthermore, the trend observed when modelling each month separately was used to investigate further combinations (see Results). The spatial distribution of seabird mortality was investigated by comparing the fit of models using various areas. Two areas i.e. Atlantic Ocean (west of 20 °E) and Indian Ocean (east of 20 °E), five areas i.e. West coast (North of Cape

Columbine), Cape Point (Cape Columbine to Cape Point), Agulhas Bank, south coast (21–25 °E) and east Coast (25 °E to the eastern boundary of the EEZ) (Fig 1). Bathymetry was investigated as a continuous variable (to the nearest metre) and in these categories: 0–500 m, 500–1 000 m, 1 000–1 500 m. Time of set was investigated by calculating nautical dawn and dusk from almanac tables. Sets commencing and finishing during daylight (i.e. after nautical dawn), were classified as 'light' and those commencing and finishing during the night (i.e. after nautical dusk), were classified as 'dark'. Sets either commencing during daylight and finishing during the night or vice versa were classified as twilight.
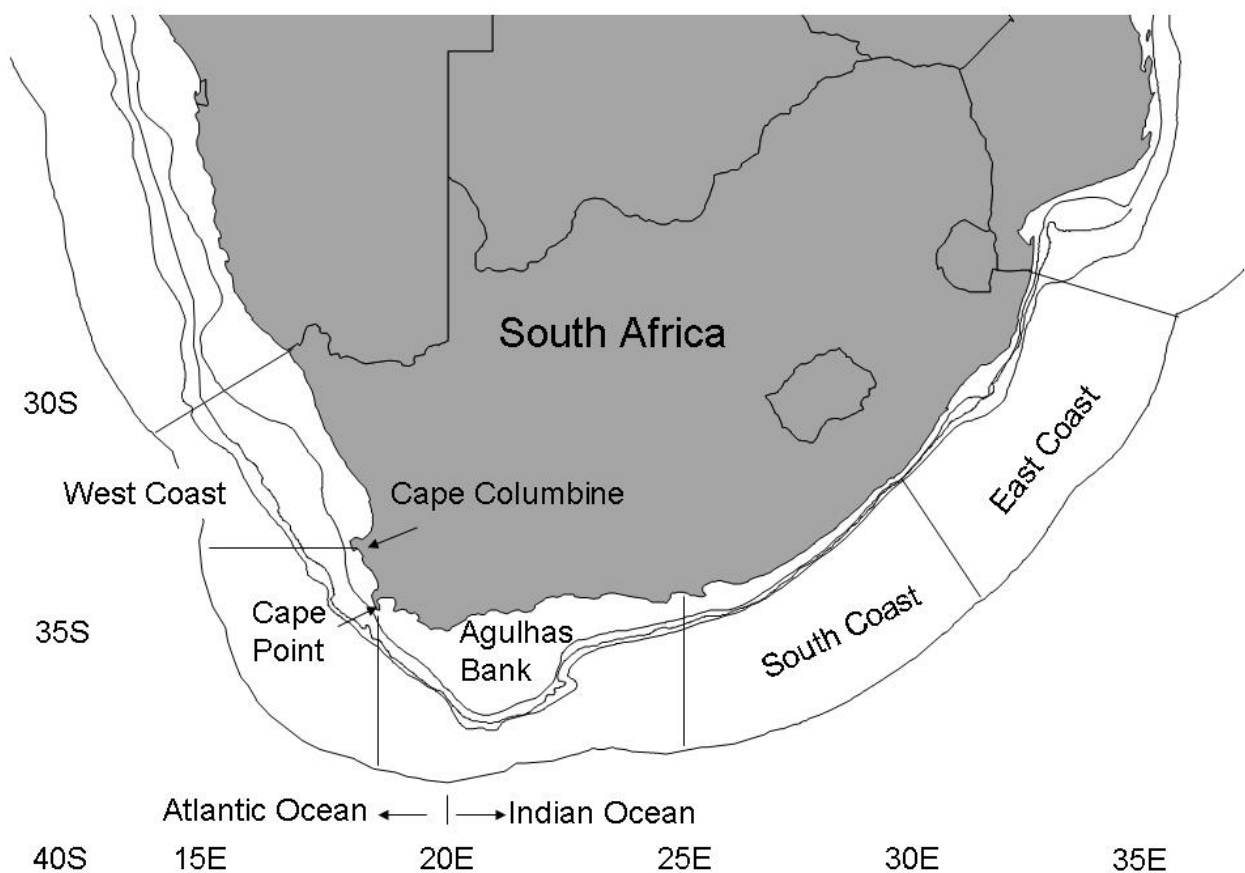

Figure 1: Map of South Africa's Exclusive Economic Zone, depicting 100 m, 500 m and 1 000 m isobath and areas used in GLM analysis.

RESULTS

Bycatch data and 13 explanatory variables were available for 2 256 longline sets. A total of 4.4 million hooks set by 50 vessels (32 South African vessels targeting Swordfish *Xiphias gladius*, 20% of total effort, 18 Asian flagged vessels targeting tunas *Thunnus* spp., 80% of total effort) were observed over the eight-year period from 1998 to 2005. The average number of hooks per set was 1 960 (range 500–3 800, SD 627). South African vessels tended to set fewer hooks per set (average 1 400, SD 346, range 500–2 000) and predominantly set their gear in the early evening compared with Asian vessels (average 2 500, SD 487, range 1 000–3 800) which typically set their gear in the early hours of the morning (Fig. 2).

a) South African vessels

b) Asian vessels

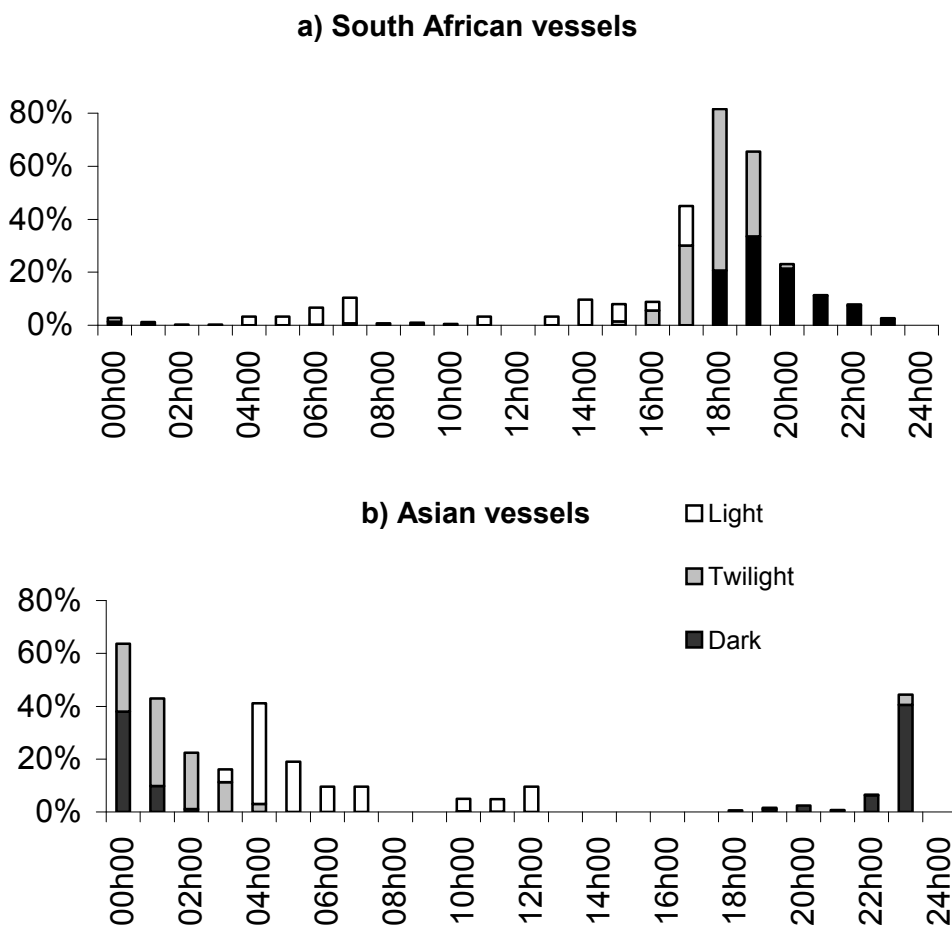Figure 2: Time of commencement of setting for a) South African and b) Asian flagged vessels based on nautical dawn and dusk. Sets were defined as 'light' if the entire setting period took place between nautical dawn and dusk (white bars), 'dark' if the entire setting period took place between nautical dusk and dawn (dark bars) and 'twilight' if it started in the dark and ended in the light or vice versa (shaded bars).
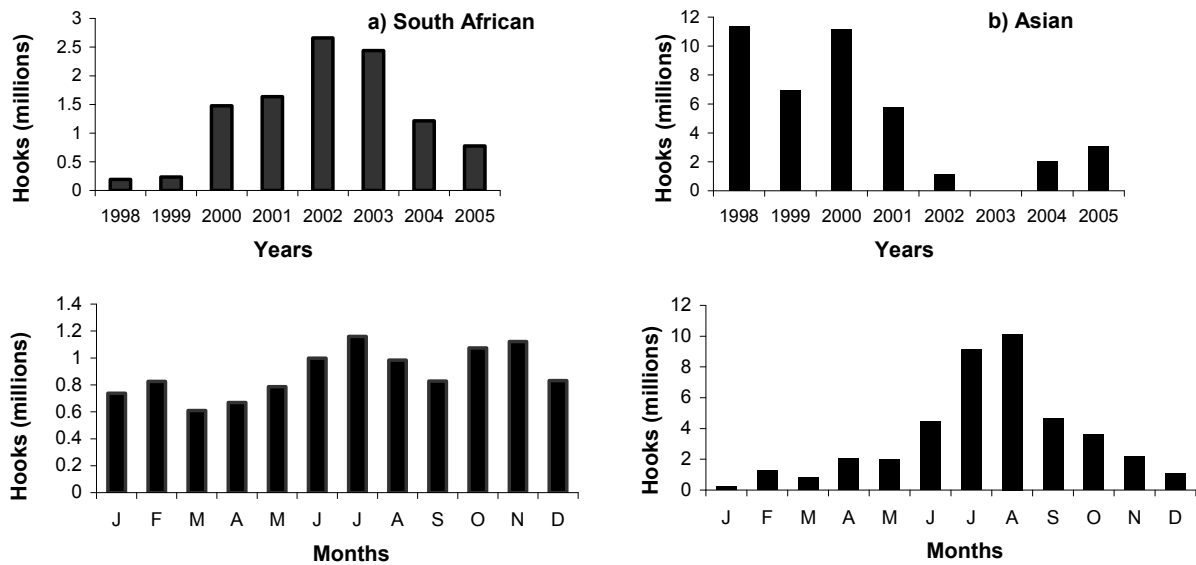
Figure 3: Annual and monthly trends in fishing effort for the a) South African and b) Asian fishing fleet operating in the South African large pelagic longline fishery, 1998–2005.

South African vessels set 10.2 million hooks from 1998 to 2005, an average of 1.3 million hooks per year. Fishing effort peaked at 2.6 million hooks in 2002, then decreased to 0.8 million hooks in 2005 (Fig. 3a). Fishing effort was relatively uniform throughout the year, with slight peaks in July and November (Fig. 3a). South African vessels tended to operate on South Africa's west coast and off Richards Bay on the east coast (Fig. 4). Observer data were collected from 827 sets or 1.0 million hooks (9.8% of total effort; Fig. 5). Although these vessels were licensed to target Swordfish, the catch was a combination of Swordfish (20.4%), tunas (35.4%), Blue Sharks *Prionace glauca* (23.9%) and Mako Sharks *Isurus oxyrinchus* (2.9%).

Asian vessels set 41.5 million hooks from 1998 to 2005, an average of 5.2 million hooks per year. Fishing effort averaged 10 million hooks per year until 2000, then decreased to 1 million hooks in 2002, as a result of the termination of the bilateral agreements between South Africa and Japan and Taiwan (Fig. 3b). In 2004, foreign vessels were allowed back into the fishery through joint venture agreements with South African companies. Fishing effort was highest during winter (June–September) (Fig. 3b). Asian fishing effort was more dispersed than the South African fishing effort and took place throughout the EEZ and beyond; the greatest proportion (21.0%) took place on the Agulhas Bank (Fig. 4). Observer data were collected from 1 414 sets or 3.4 million hooks (8.2% of the total); 91% of these were from 2005, when improved observer coverage was the result of a condition placed on foreign vessels fishing in South African waters. Observer data were not spatially representative. However, areas of highest fishing effort corresponded with sampling areas and are considered representative of those areas (Fig. 6). Asian vessels targeted tunas which comprised 76.6% of their catch, with the remainder comprising of Swordfish (2.4%), Blue Sharks (5.3%), Mako Sharks (2.6%) and other sharks and fish (1.0%).
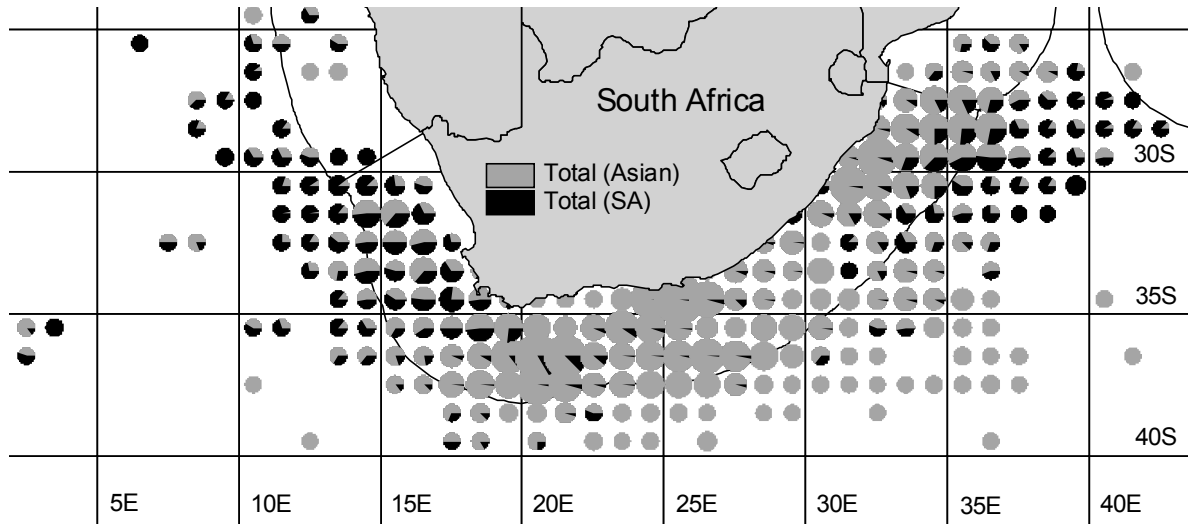
Figure 4: The distribution of Asian (shaded) and South African (black) fishing effort by 1° grid cell, 1998–2005. Note that effort extended outside the South African EEZ, both into international waters and into the EEZs of Namibia and Mozambique. Symbol size is a relative measure of the proportion of effort per 1° grid cell.
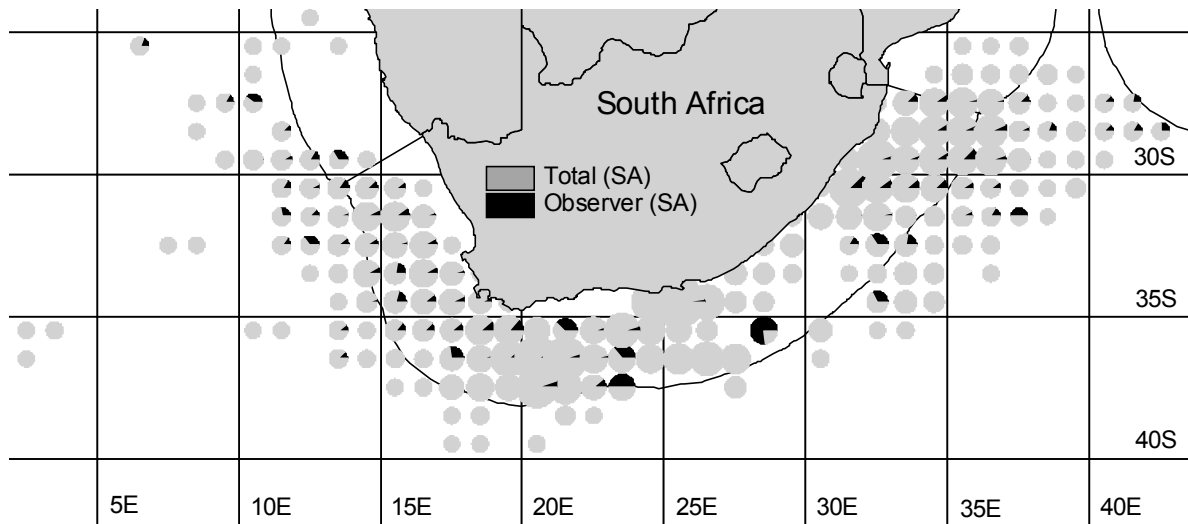
Figure 5: Distribution of observer effort (black) in the South African fleet by 1° grid cell, 1998–2005. Symbol size is a relative measure of the proportion of effort per 1° grid cell.
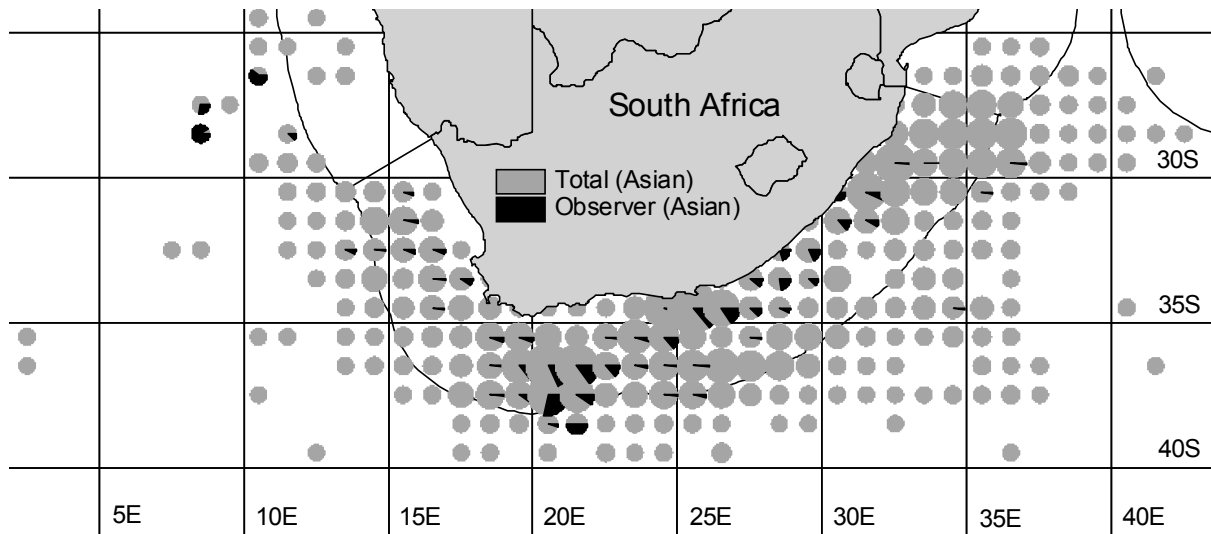
Figure 6: Distribution of observer effort (black) on Asian flagged vessels by 1° grid cell, 1998–2005. Symbol size is a relative measure of the proportion of effort per 1° grid cell.

Seabird bycatch

A total of 1 954 seabird mortalities was recorded by observers on pelagic longliners from 1998 to 2005, 1 191 of which were returned to port. Species identification could be verified for 43.0% of observed mortalities, of which 60.4% were identified correctly to species and 70.6% were correct to the grouping 'albatross', 'petrel' or 'shearwater'. 94.0% of albatrosses were identified correctly as an 'albatross' (the remainder being simply labelled 'seabird'). White-chinned Petrels *Procellaria aequinoctialis* were most commonly misidentified by observers and labelled as 'giant petrels' *Macronectes* spp, 'Great-winged Petrel' *Pterodroma macroptera* and 'Flesh-footed Shearwaters' *Puffinus carneipes*. In fact, no Flesh-footed Shearwater or Great-winged Petrel specimens were returned to port. A total of 16 species was recorded by fisheries observers, 10 of which were in threat categories (BirdLife International 2007). Tristan *Diomedea dabbenena* and Southern Royal Albatrosses *D. epomophora* were frequently misidentified as Wandering Albatrosses *D. exulans* (Table 1). Cape Gannets were reported by observers, but none were returned to port.

The identification of 11 species was confirmed on shore (Table 1). White-chinned Petrels made up 68.9% of the seabird bycatch, caught at a rate of 0.25/1000 hooks. Albatrosses made up approximately 30.3% of the bycatch, at a rate of 0.14/1000 hooks. Three albatross species were recorded in significant numbers: namely shy-type (23.9%, at a rate of 0.09/1000 hooks, Black-browed (4.1%, at a rate of 0.02/1000 hooks) and Indian Yellow-nosed albatrosses *T. carteri* (1.5%, at a rate of 0.01/1000 hooks). Atlantic Yellow-nosed Albatrosses *T. chlororynchos,* Southern Royal Albatross, Tristan Albatross, Northern Giant Petrel *Macronectes halli*, Southern Giant Petrel *M. giganteus,* Great Shearwater *Puffinus*

gravis and Subantarctic Skuas *Catharacta antarctica* were also confirmed caught (Table 1). Overall, birds were caught at a catch rate of 0.44/1000 hooks.

Table 1: Seabird bycatch composition reported by fishery observers and those confirmed on shore by experts in the pelagic longline fishery off South Africa, 1998–2005.

Common Name	Scientific Name	IUCN Status	Observers n	Observers %	Confirmed n	Confirmed %
Shy/White-capped Albatross	*Thalassarche cauta/steadi*	Near Threatened	411	21.0%	285	23.9%
Black browed Albatross	*Thalassarche melanophrys*	Endangered	89	4.6%	49	4.1%
Indian Yellow-nosed Albatross	*Thalassarche carteri*	Endangered	44	2.3%	18	1.5%
Atlantic Yellow-nosed Albatross	*Thalassarche chlororhynchos*	Endangered			3	0.3%
Grey-headed Albatross	*Thalassarche chrysostoma*	Vulnerable	2	0.1%	-	-
Southern Royal Albatross	*Diomedea epomophora*	Vulnerable	-	-	4	0.3%
Northern Royal Albatross	*Diomedea sanfordi*	Endangered	4	0.2%	-	-
Tristan Albatross	*Diomedea dabbenena*	Endangered	-	-	3	0.3%
Wandering Albatross	*Diomedea exulans*	Vulnerable	3	0.2%	-	-
Unidentified albatrosses			78	4.0%	-	-
Northern Giant Petrel	*Macronectes halli*	Near Threatened	15	4.0%	2	0.2%
Southern Giant Petrel	*Macronectes giganteus*	Vulnerable			2	0.2%
White-chinned Petrel	*Procellaria aequinoctialis*	Vulnerable	1099	56.2%	823	68.9%
Spectacled Petrel	*Procellaria conspicillata*	Critically Endangered	2	0.1%	-	-
Cape Petrel	*Daption capense*	Near Threatened	7	0.4%	-	-
Great-winged Petrel	*Pterodroma macroptera*	Least Concern	2	0.1%	-	-
Unidentified petrels			7	0.4%	-	-
Great shearwater	*Puffinus gravis*	Least Concern	-	-	4	0.3%
Flesh-footed shearwater	*Puffinus carneipes*	Least Concern	3	0.2%	-	-
Sub-Antarctic Skua	*Catharacta antarctica*	Least Concern	1	0.1%	1	0.1%
Cape Gannet	*Morus capensis*	Vulnerable	20	1.0%	-	-
Other unidentified seabirds			167	8.5%	-	-
Total			1954	100.0%	1194	100.0%

Significantly more male than female White-chinned Petrels were killed, with most being immature birds (59.0%) (Table 2). 15.3% of adult birds had brood patches and 5.9% had enlarged gonads. These birds, which were likely to be breeding, were all recorded south of Cape Point and predominantly east of 20 °E.

There was no other evidence of sex-biased mortality (Table 2). Shy-type and Black-browed albatrosses were mostly juvenile/immature (73.4% and 61.2%, respectively) with a smaller proportions of sub-adult (16.0% and 16.3%) and adult (10.6% and 22.5%) birds, respectively (Table 2). By comparison, most yellow-nosed albatrosses were adult (83% and 67% of Indian and Atlantic Yellow-nosed Albatrosses, respectively). Four of 12 adult Indian and 2 of 3 adult Atlantic Yellow-nosed Albatrosses either had a brood patch or enlarged gonads indicating they were breeding or recently abandoned a breeding attempt.

Table 2: Summary of age and sex distribution of seabirds caught by the South African pelagic longline fishery and returned to port for autopsy and identification, 1998–2005.

Common Name	n	% F	p-value
White-chinned Petrel	694	41%	<0.001
shy-type albatross	183	53%	0.9
Black-browed Albatross	41	59%	0.87
Indian Yellow-nosed Albatross	17	55%	0.99
Tristan Albatross	3	33%	-
Southern Royal Albatross	1	-	-
giant petrels	4	50%	-
Great Shearwater	4	0%	0.06
Subantarctic Skua	1	100%	-
Total	948	44%	-

Table 3: Summary of the percentage variance explained from GLM analysis of White-chinned Petrels, shy-type albatrosses, Black-browed Albatrosses and seabirds in general.

Parameter	White-chinned Petrels	shy-type albatrosses	Black-browed Albatrosses	All Birds
Vessel Name	21.9	19.3	24.2	13.5
Flag	*	0.7	0.5	8
Moonphase	8.8	3.6	1.4	7.2
Daylight	0.8	0	0.1	0.4
Season	7	2.5	5.5	4.7
Area		} 1.2**		0.3
Bathymetry	0.8			0.2
Ocean	2.1		3.4***	
Tori line	0.5			0.5
Beaufort scale	0.9	0.1	0.8	0.7
Discarding		0.4		
Branch line			0.1	
Total	42.8	27.6	36.0	35.5

*   *Although vessel flag on its own explained 9.7% of the variance, vessel name only produced a better fit.*

**   *The model produced the same fit whether bathymetry or area was included, but no additional deviance was explained by adding both.*

***   *Of the three measures of location i.e. area, ocean basin and bathymetry, ocean basin gave the best fit.*

The effect of season and area

In general, catch rates were greatest on the Agulhas Bank and off Cape Point where they reached a maximum of 3.9/1000 hooks (Fig. 7) and on the continental shelf edge in water depths of between 500 m and 2 000 m and reached a maximum of 0.6/1000 hooks between 1 000 and 2 000 m (Fig. 8). Adult White-chinned Petrels were caught from April to December and reached a maximum in June when catch rates peaked at 1.4 White-chinned Petrels per 1000 hooks. Whereas immature White-chinned Petrels were predominantly caught from June to September and reached a maximum in September. Significantly more White-chinned Petrels were caught in the Indian Ocean (p=<0.001) and along the continental shelf edge (p<0.001) (Table 3). Catch rates reached a peak of 3.9/1000 hooks on the Agulhas Bank (21 °E grid cell) (Fig. 7d).

Although mortality in all age groups of shy-type albatrosses occurred throughout the year, it peaked in winter months (June–October) (t=5.74, p<0.001). Shy-type albatrosses were predominantly caught on the Agulhas Bank east of 20 °E and along the west coast (Fig. 7a). Catch rates reached a maximum of 3.4/1000 hooks just south of Cape Agulhas.

Adult and sub-adult Black-browed Albatrosses were caught from May to September whereas juvenile and immature birds were caught for a slightly longer period (May–October) (t=5.01, p<0.001) (Table 4). Significantly more Black-browed Albatrosses were caught in the Atlantic Ocean, west of 20 °E (3.4% of variance, t=–5.23, p<0.001) (Fig. 7b). Catch rates peaked at 3.3/1000 hooks just south of Cape Agulhas. Although there were insufficient data to model yellow-nosed albatross bycatch, the catch rate reached a peak of 0.10/1000 hooks off the west coast of South Africa (Fig. 7c).
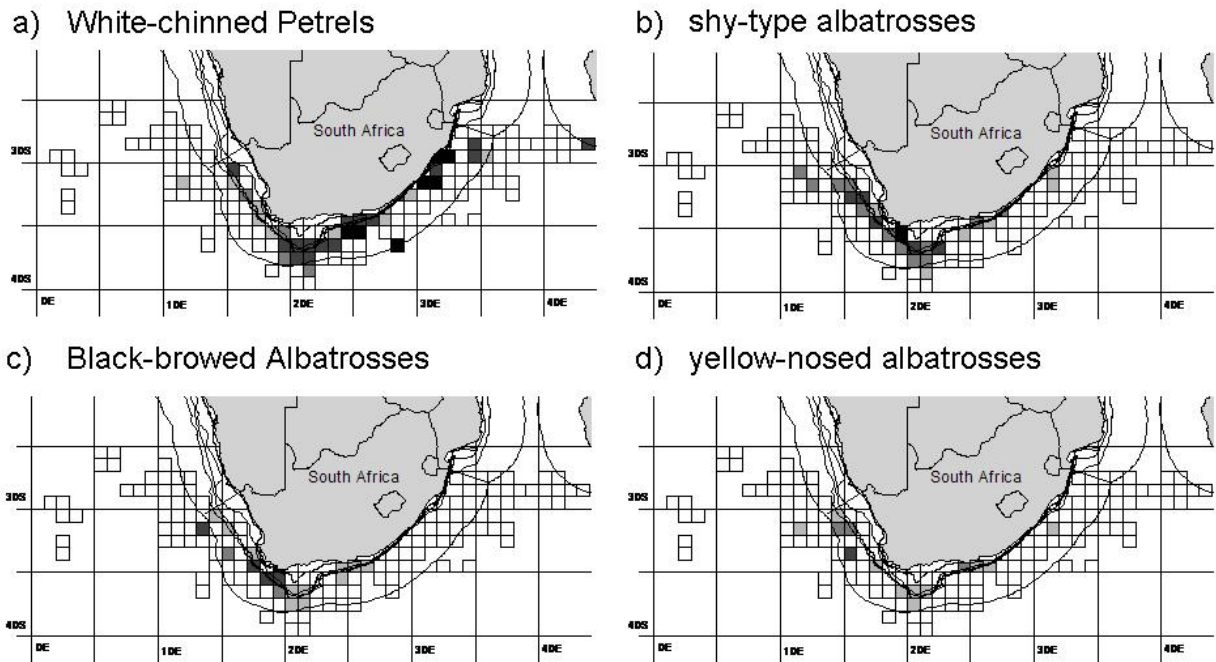
Figure 7/…

Figure 7: Catch rates per 1° grid cell of White-chinned Petrels and three species of albatross caught by the South African large pelagic longline fishery, 1998–2005.
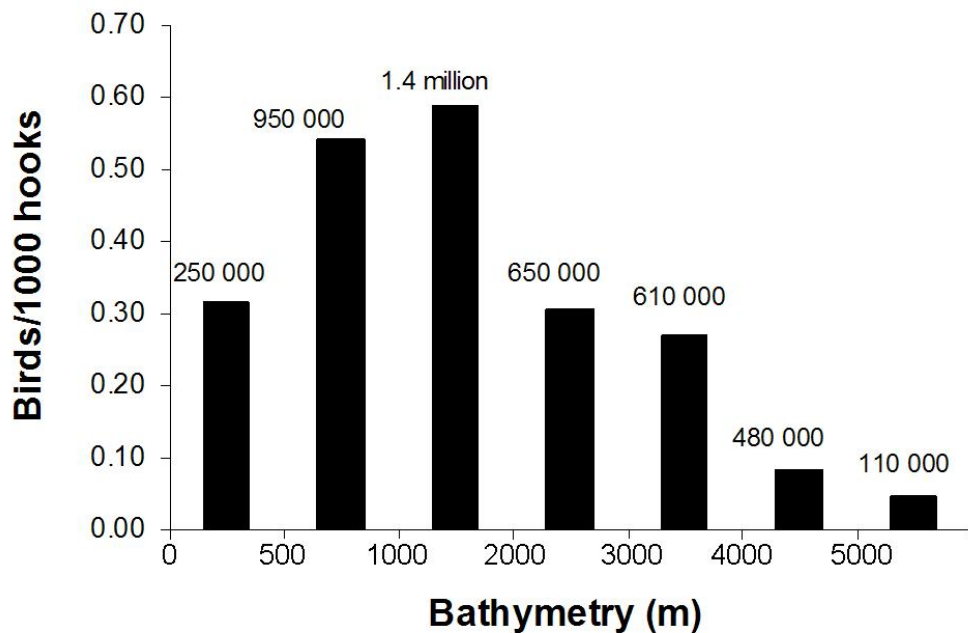

Figure 8: The effect of bathymetry on seabird catch rates by the South African pelagic longline fishery, 1998–2005.

The effect of vessel flag and vessel name

Most variance in seabird bycatch was accounted for by individual vessel (Table 3). The term vessel includes properties of skipper behaviour (such as preference for bait type, area, time of settings etc) and fishing operations (gear configuration, deck lighting, line setter, bait caster etc). It is likely that a number of factors affecting mortality are confounded in this term. Ten of 50 vessels (20%) caught birds at a rate higher than the average of 0.44/1000 hooks i.e. 0.90/1000 hooks and were responsible for 74% of the mortality from 44.2% of the hooks. Four were South African flagged and six were Asian flagged under joint-venture agreements. A single vessel (Asian) caught 22.7% of observed birds at a rate of 2.3/1000 hooks. Vessel flag was a significant predictor of seabird bycatch in all models, but a high degree of co-linearity occurred because vessels which were either flagged to South Africa or Asia for the duration of the study.

Most birds (88%) were caught by Asian flagged vessels (72% of albatrosses and 97% of petrels). Asian tuna directed vessels caught seabirds at a rate of 0.51/1000 hooks (0.58/1000 hooks in winter and 0.14/1000 hooks in summer) compared to South African swordfish directed vessels which caught seabirds at a rate of 0.23/1000 hooks (0.22/1000 hooks in winter and 0.24/1000 hooks in summer) (Fig. 9, Table 4).

Table 4: Summary of catch rates (birds killed per 1000 hooks set) and extrapolations stratified by 5° grid cell, season and flag.

Species	White-chinned Petrels	shy-type albatrosses	Black-browed Albatrosses	yellow-nosed albatrosses
Asian tuna winter	0.36	0.11	0.02	0.01
Asian tuna summer	0.09	0.03	0.00	0.02
SA Swordfish winter	0.05	0.07	0.05	0.03
SA Swordfish summer	0.02	0.09	0.02	0.01
Total catch rate	0.25	0.09	0.02	0.01
Estimated total capture (1998-2005)	13 185	4 768	988	679
Expected capture per year	1 650	600	125	85
Estimated capture of 30 tuna vessels*	4 960	1 529	245	179
Estimated capture of 20 Swordfish vessels*	137	314	137	83
Total (20+30)	5 100	1 840	380	260

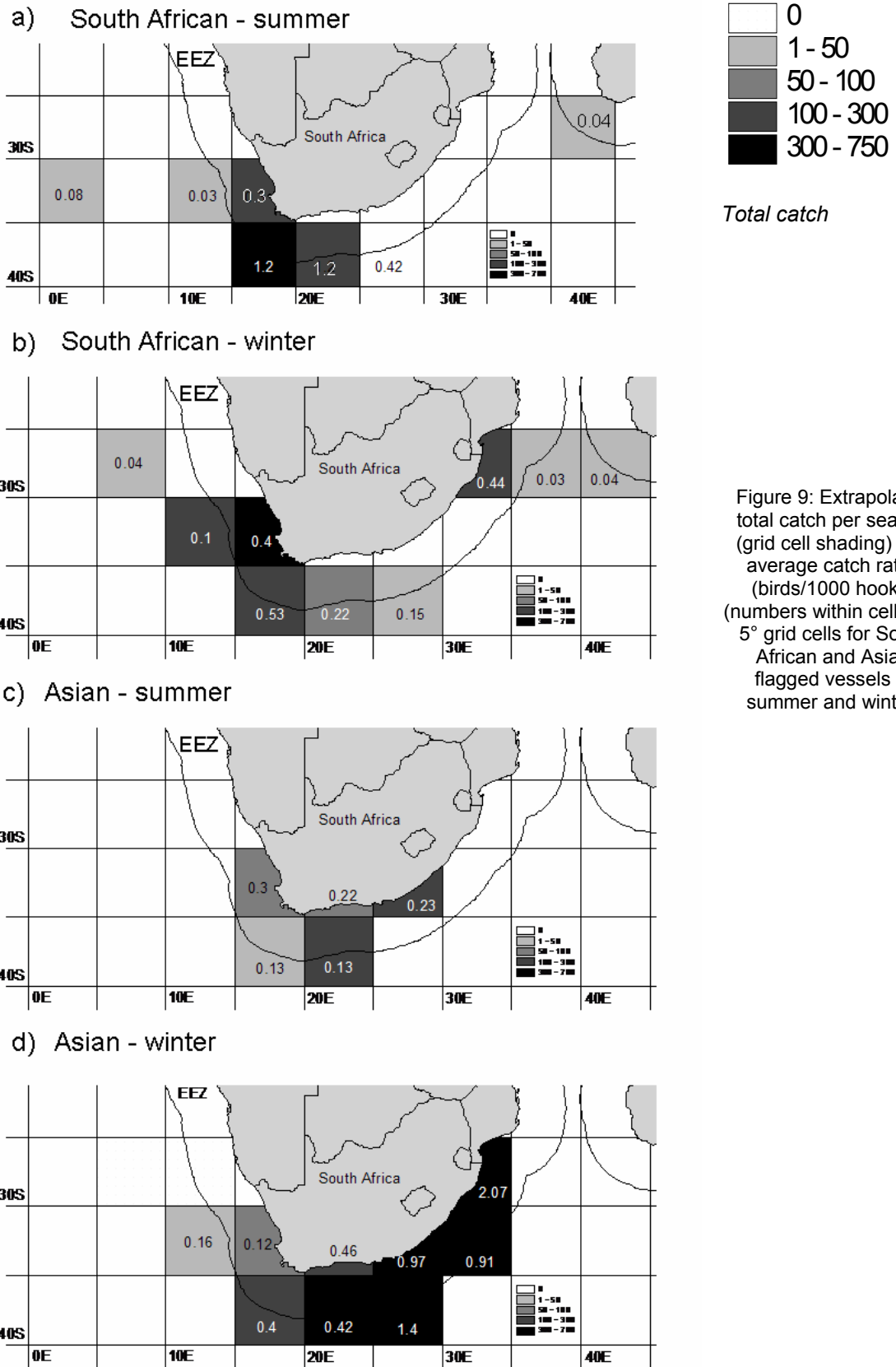*average of 2000 hooks per set and 200 sets per year*

Figure 9: Extrapolated total catch per season (grid cell shading) and average catch rates (birds/1000 hooks) (numbers within cells) for 5° grid cells for South African and Asian flagged vessels in summer and winter.

The effect of daylight and moon phase

Light levels affected bycatch rates both in terms of day versus night sets and moonlight. Catch rates were highest when sets occurred during the day (p<0.001) and over full moon periods when lines were set at night (p<0.001) (Table 3, Fig. 10). The average catch rate was 1.07/1000 hooks during full moon compared to 0.09/1000 hooks at new moon (Fig. 10).

Time of set was not included in the model for shy-type albatrosses because it was not significant (p=0.54) and the increased number of parameters increased the AIC. Interestingly, light conditions were particularly important in explaining White-chinned Petrel mortality, with more birds caught at night than during the day (t=–4.61, p<0.001).
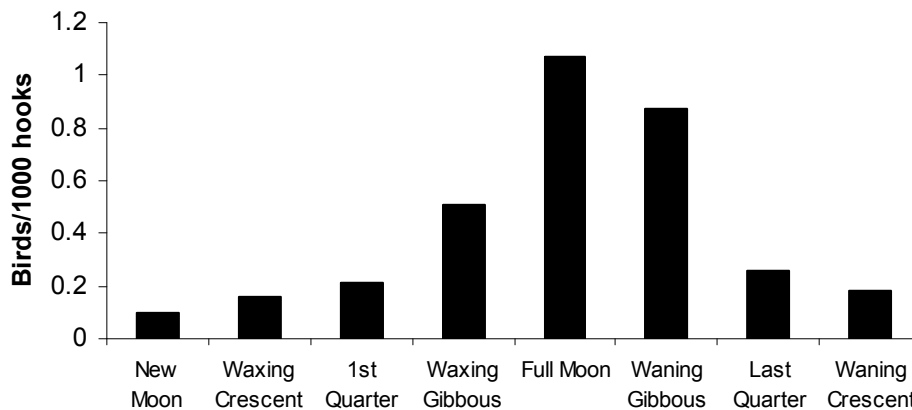
Figure 10: The effect of moon phase on seabird mortality, 1998–2005.

Effect of sea state

In all models, wind strength played a significant (p<0.001) role in determining seabird bycatch, with more birds caught as wind strength increased (t= –6.76, 1,7 and 2.1 for White-chinned Petrel, shy-type albatross and Black-browed albatross respectively) (Table 3).

Effect of mitigation measures

The use of a tori line decreased seabird mortality from 0.6 per 1000 hooks to 0.1 per 1000 hooks (t= –3.76, p<0.001) (Table 3). Tori lines were only used on 51% of sets. This increased to 72.6% of sets in 2005. In the case of shy-type albatrosses, discarding offal on the same side as hauling explained 0.4% of the variance (t= –2.97, p<0.001). Capture rate decreased when discarding occurred on the opposite side to hauling (Table 3). In the case of Black-browed Albatrosses, branch line length explained 0.1% of variance (t= –2.07, p=0.04), with increasing mortality linked to increasing branch line length (Table 3).

Total mortality

Simply extrapolating the average catch rate (0.44/1000 hooks) to the total fishing effort predicted that approximately 2 870 birds were killed per year (range 1 100–5 600) (Table 5). When the extrapolation took vessel flag into account the annual average estimate increased to 2 955 birds per year (range 560–6 050) (Table 5). Extrapolation based on stratification by 5° grid cell, by season and by flag suggested that 2 890 birds were caught per year (1 650 White-chinned Petrels, 600 shy-type albatrosses, 125 Black-browed Albatrosses and 85 Indian Yellow-nosed Albatrosses, 430 other seabirds) (range 550–6 200) (Table 4 and 5). In all cases a decreasing trend was observed from approximately 5 900 in 1998 to 1 800 in 2005. Although, generalised linear modelling revealed no significant annual trend in catch rate (t=0.99, p=0.323).

Table 5: Summary of total bird mortality in the South African large pelagic longline fishery using a basic extrapolation of a single catch rate applied to the total effort, an extrapolation by flag and an extrapolation stratified by 5° grid cell, season and flag, 1998–2005.

Year	Basic	Extrapolation by flag			Extraplation by flag, area and season		
	Total	Asian	SA	Total	Asian	SA	Total
1998	5 095	5 807	44	5 852	5 673	44	5 716
1999	3 148	3 529	54	3 583	3 324	54	3 378
2000	5 574	5 708	340	6 047	5 885	336	6 221
2001	3 273	2,959	377	3 335	2 672	369	3 041
2002	1 688	601	611	1 212	569	597	1 166
2003	1 073	0	561	561	0	547	547
2004	1 417	1 024	279	1 303	994	274	1 267
2005	1 694	1 569	178	1 747	1 610	173	1 783
min	1 073	0	44	561	0	44	547
max	5 574	5 807	611	6 047	5 885	597	6 221
Average	**2 870**	2 649	306	**2 955**	2 591	299	**2 890**

DISCUSSION

Magnitude of seabird bycatch

Seabird catch rates reported in this study were lower than previous estimates for the South African pelagic longline fishery which was estimated to kill birds at a rate of 1.60/1000 hooks (Ryan *et al.* 2002) during 1998–2000 when primarily Asian vessels were operating on the western Agulhas Bank. At this time a total of between 19 000 and 30 000 seabirds was estimated to be killed each year. The present study, based on a much larger sample size, reported a catch rate of 0.44/1000 hooks or an estimated total catch of 2 890 birds per year (1998–2005). The improvement was most likely linked to the termination of the foreign bilateral agreements, as well as increased awareness among fishers linked to ongoing education campaigns. However, it is worth bearing in mind that some of the apparent decrease in catch rate could reflect reduced numbers of birds at sea, as a result of ongoing population decreases in several key species (BirdLife International 2007).

There is no standard method of estimating total bycatch from observer data (Lewison & Crowder 2003). This study investigated three techniques for extrapolating total seabird bycatch which ranged from the most basic to including variables which were significant predictors of bycatch i.e. vessel flag (also proxy for target species and gear configuration), area (5° grid cell) and season (winter versus summer) and found little difference between total estimates (2 870, 2 950, 2 890 respectively) (Table 5). Interestingly, the smallest difference was between the most basic extrapolation and the most complex. A similar method of extrapolation by strata was used by Klaer & Polacheck (1997), Stehn *et al.* (2001), NPFMC (2003), and Lewison & Crowder (2003). A large proportion of sets where no birds are caught frequently complicates extrapolations of seabird bycatch (Lewison & Crowder 2003). Pennington (1983) overcame this problem by only using sets where bycatch took place. Categorical and Regression Tree Analysis (CART) also has been used to estimate seabird bycatch (NMFS 2000) and uses continuous predictor variables to identify non-overlapping groups (Breiman *et al*. 1984). The Pennington method was not used because there were adequate samples of sets with bycatch. The CART method was not used because the data were so complex that the algorithm failed to converge and results were not readily interpretable. Generalised linear models performed adequately for all analyses and were the preferred method that was finally used in this analysis. Despite uncertainty and estimation error, estimates are most likely conservative. Estimates are based on observed sets (accounting for 9% of total fishing effort, 1998–2005), where compliance to regulations is known to be higher than unobserved sets with the implication that compliance is lower and hence seabird bycatch higher on unobserved sets (Gales *et al.* 1998). Furthermore, fisheries observers were not required to specifically record seabird bycatch data until 2002 and only received formal training on seabird identification from 2004. Birds dislodged from the line and therefore not hauled onboard may account for up to 30% of birds hooked (Brothers 1991); were also not included in this estimate.

Similar catch rates have been reported in the Japanese longline fleet operating in the Southern Ocean (Brothers 1991). Catch rates higher than that reported in this study include: 1.35/1000 hooks reported in the Brazilian pelagic longline fishery (Olmos *et al.* 2000), 0.92/1000 hooks in the Australian domestic fishery (1994/5) (Brothers & Foster 1997), 4.7/1000 hooks in the Uruguayan tuna and swordfish fishery 1993/94 (Sagi *et al.* 1998). Similarly, catch rates lower than that reported in this study have also been reported for pelagic fisheries in many regions including off the east coast of Australia (0.28/1000 hooks in the eastern tuna and billfish fishery) (Baker & Wise 2005), the Southern Ocean (Japanese Southern Bluefin *Thunnus maccoyii* fleet report to catch birds at a rate of 0.14–0.18 (2001/02) (Kiyota & Takeuchi 2004), south east Pacific (Spanish longline fleet report 0.09/1000 hooks) (Mejuto *et al.* 2003), Southern Indian Ocean (Spanish fleet report rates of 0.002/1000 hooks) (García-Cortés & Mejuto 2005) and Uruguay (0.26/1000 hooks) (Jimenez 2005).

Factors affecting bycatch

Area and season were good predictors of seabird mortality. Sub-Antarctic birds, such as albatrosses and petrels migrate to the productive waters off southern Africa in their non-breeding months (Crawford *et al.* 1991). This coincides with the period of highest mortality during winter, when those species breeding farther south on Southern Ocean islands, are not breeding. Large numbers of pelagic seabirds aggregate on the continental shelf and shelf break (Ryan & Moloney 1988, Ryan & Rose 1995), which explains the higher mortality recorded in that region. The present study suggested that the area south west of Cape Point and the Agulhas Bank are especially important for albatrosses, whereas the continental shelf off the east coast seems to be more important for White-chinned Petrels. It may also be that White-chinned Petrels are not more abundant on the east coast in absolute numbers, but rather that their proportion relative to the number of albatrosses is highest in that region. Where the proportion of White-chinned Petrels is lower, the larger albatrosses may out-compete the smaller petrels (Camphuysen *et al.* 1995, Wanless 1998).

Light conditions also played an important role in explaining the observed bycatch rates. Similar findings have been found elsewhere (Ashford *et al.* 1995, Klaer & Polacheck 1995, 1998, Brothers *et al.* 1999a). Klaer and Polacheck (1998) report catch rates to be five times higher for sets during the day compared with those at night. Albatrosses, which tend not to feed at night except in bright moon conditions (Harrison *et al.* 1991), were more likely to be caught during the day. Shy Albatrosses feed on prey that are found at or near the surface during the day (Hedd *et al.* 2001). Weimerskirch and Guionnet (2002) report Black-browed Albatrosses to be particularly active in the first part of the night i.e. from dusk till mid-night. White-chinned Petrels on the other hand, which do feed at night (Cherel *et al.* 1996), are more likely to be caught at night when competition is also at its lowest and thus it is not surprising that time of set accounted for more of the variance in the White-chinned Petrel model than for the other species models. Consistent with other studies (Klaer & Polacheck 1998), moon phase was a key factor in all the models. Hedd *et al.* (2001) reported that

nocturnal activity of Shy Albatrosses was strongly influenced by moon phase, when birds spent more time flying at increased flight speeds. This activity at full moon highlights the fact that setting lines at night will reduce mortality, but not eliminate it and thus it is essential to use tori lines especially on bright moon nights. Most vessels discarded offal on the opposite side to hauling, helping to decrease seabird mortality. The fact that more birds were killed with longer branch lines is likely to be the result of the slower sink rates of long branch lines.

Impact on key species

The most common species killed, the White-chinned Petrel, is classified as Vulnerable (BirdLife International 2007). The sexual bias (more males killed) evident in White-chinned Petrel mortality in the region is likely to further exacerbate the effect on this species which displays monogamous breeding and often long-lasting pair bonds (Ryan & Boix-Hinzen 1999, Ryan 1999). A male bias in longline fisheries mortality has been recorded elsewhere (Ryan & Boix-Hinzen 1999, Ryan 1999, Nel *et al.* 2002). This species breeds at islands throughout the sub-Antarctic and disperses widely during its non-breeding season. As a result it is killed by many fisheries throughout its range (Cherel *et al.* 1996, Barnes *et al.* 1997, CCAMLR 1997, Olmos 1997, Weimerskirch *et al.* 1999). Few reliable historical population estimates exist for this species. White-chinned Petrels nest in burrows and thus present a challenge to assess accurately. A study conducted at Bird Island (South Georgia) estimated a decrease of 28% over 20 years (Berrow *et al.* 2000). White-chinned Petrels caught in South African waters include both breeding and non-breeding birds (Wiemerskirch *et al.* 1999, this study). This is in contrast to most albatrosses found in South African waters which were mostly non-breeding birds, with the exception of some yellow-nosed albatrosses. Weimerskirch *et al.* (1999) reported that satellite tracked White-chinned Petrels breeding on the Crozet Islands frequently entered the South African EEZ (Wiemerskirch *et al.* 1999).

Shy-type albatrosses are the most commonly caught species of albatross in South African waters, which are important foraging areas for juvenile and non-breeding birds (Abrams 1983, 1985, Ryan & Moloney 1988). When competing for food, larger albatrosses generally out-compete smaller albatrosses (Phillips *et al.* 2003). Shy-type albatrosses are larger and may be more aggressive, dominating Black-browed and yellow-nosed albatrosses, explaining why shy-type albatrosses make up the larger proportion of the catch independent of their relatively even abundance in South African waters (Crawford *et al*. 1991, Chapter 7). Most (95%) shy-type albatrosses foraging in southern African waters are White-capped Albatrosses *T. steadi* from the New Zealand population (Abbott *et al.* 2006, Baker *et al*. 2007a). This species is impacted by fishery interactions across much of its foraging and breeding range, with all age classes at risk (Baker *et al.* 2007a). In a recent global review, Baker *et al.* (2007a) estimated that 8 000 White-capped Albatrosses were killed each year by trawl and longline fisheries interactions in the Southern Ocean. However, little is known about their population status, breeding biology, life history and at-sea distribution (Robertson *et al*. 2003). While attempts have been made to estimate the population, these estimates have not been published (Baker *et al.* 2007b). A recent assessment indicates that global

population is approximately 117 000 annual breeding pairs, which is larger than previously thought (Baker *et al.* 2007b). The reported level of mortality highlights the need to continue to acquire accurate population estimates and trends for White-capped Albatross populations to assess the impact of fisheries operations on this species. Because of the limited estimation of population size for this species, there can be no reliable assessment of population status or trend (Gales 1998, Baker *et al.* 2007a, b). In the absence of this information, it is not possible to assess accurately whether this level of mortality is sustainable.

In contrast, accurate population status and trend information are available for Black-browed Albatrosses feeding in the Benguela, which breed on South Georgia in the south west Atlantic and Crozet and Kerguelen Islands in the southern Indian Ocean (Croxall *et al.* 1998, Weimerskirch & Jouventin 1998, Poncet *et al.* 2006). This species is globally listed as Endangered (BirdLife International 2007). At South Georgia, numbers have decreased at 4.8% per year since the mid 1970s (Croxall *et al.* 1998). Black-browed Albatross populations breeding at Bird Island (15% of the South Georgia total) have declined by 35% from 1989/1990 till 1997 (Croxall *et al.* 1998). Similarly, Black-browed Albatross populations breeding at Kerguelen have declined by 17% between 1978/1979 and 1994/1995 (Weimerskirch & Jouventin 1998). Incidental fisheries mortality has been identified as the main cause of the observed decline (BirdLife International 2007), and mortalities in South African waters contribute to this trend. The productive waters of the southern Benguela are an important foraging area for Black-browed Albatrosses, especially juveniles and non-breeding adults. The larger number of immature birds killed (73%) suggests that they are more at risk. In addition, most adults leave each summer, returning to their breeding islands. During this time the birds caught were almost exclusively juveniles.

Implications for management

The rate at which seabirds were caught in this fishery is almost an order of magnitude greater than the international target of 0.05/1000 hooks (Environment Australia 1998, FAO 1999). Decreases in levels of mortality have been achieved by the introduction of effective and relatively inexpensive measures (Alexander *et al.* 1997, Brothers *et al.* 1999b, FAO 1999, Melvin & Robertson 2000, Gilman 2001, Melvin *et al.* 2004). Mitigation measures such as the use of a tori line, night setting, optimal line sink rates and discarding practices are conditions of longline fishing permits in South Africa, but compliance is low. Compliance improved in 2005 when improved observer coverage was the result of a condition placed on joint-venture vessels operating in the fleet. Similar improvements in compliance with improved observer coverage have been reported elsewhere (Gales *et al.* 1998). Management actions should therefore focus on addressing low compliance. The use of a tori line has become the primary and most commonly prescribed seabird mortality mitigation measure in longline fisheries globally (Brothers *et al.* 1999a, Melvin *et al.* 2004). A tori line consists of a line with a number of streamers attached to it. The streamers are designed to cover the point where the bait enters the water and distracts foraging birds from taking the baited hooks (Brothers *et al.* 1999a). The system works well for surface feeding birds, but

diving birds can still dive to the bait outside of the effective area of the streamers. Still, this method has been demonstrated to reduce bycatch rates by 70–96% (Brothers *et al*. 1999a, Mc Namara *et al*. 1999, Boggs 2001, Løkkeborg 2003). Melvin *et al*. (2001) reported a reduced number of attacks on baits when using paired, compared to single tori line.

Controlling for all other factors, tori lines significantly reduced the number of birds caught by the South African pelagic longline fishery from 0.64/1000 hooks to 0.10/1000 hooks (0.04% variance, p<0.001). Although this represents a significant reduction, it is still substantially higher than the accepted target of 0.05/1000 hooks (Environment Australia 1998, FAO 1999, Cooper & Ryan 2003). Tori line efficacy varies with the design, deployment, the sinking rate of gear, the assemblage of seabirds present, their diving ability, the season and weather conditions (Klaer & Polacheck 1995, Brothers *et al*. 1999a, Melvin & Robertson 2000, Melvin *et al*. 2004). A number of studies undertaken in other southern Hemisphere pelagic longline fisheries have also reported reduced efficacy of tori lines (Murray *et al*. 1993, Duckworth 1995, Klaer & Polacheck 1995, Brothers *et al*. 1999a). Melvin *et al*. (2004) suggested the reason for these studies not being able to show consistent and clear benefits from using tori lines was due to tori line design. In South Africa, tori line specifications were seldom recorded by observers. Where they were recorded, they frequently did not meet the regulations which stipulate that a tori line should have at least 28 paired streamers ranging from 4 m in length immediately astern the vessels to 1 m further astern, spaced 5 m apart (starting 10 m astern the vessel), attached 7 m above the water and have sufficient drag (e.g. buoy, road cone or sea-anchor) to ensure at least 100 m aerial coverage (DEAT 2005). Tori lines used by Asian-flagged vessels, in particular, are considered inadequate and typically had no drag at the end of the line and streamers were approximately 50 cm in length.

Recent work comparing the use of a single versus paired tori lines in demersal longline fisheries strongly suggests that paired lines are more effective at reducing seabird attacks (Melvin *et al*. 2001, Sullivan & Reid 2002). It is recommended that the effectiveness of paired tori lines is investigated in the South African pelagic longline fishery given the high levels of seabird bycatch currently experienced in this fishery. Furthermore, the poor compliance to this measure should be addressed and observer protocols should be amended to include information on tori line design to facilitate accurate assessment of tori line efficacy.

The proposed increase in fishing effort to 50 rights holders is estimated to potentially increase the fishing effort threefold from an average of 6.4 million hooks per year (1998–2005) to 20 million hooks per year (average of 200 sea days and 2 000 hooks per set; Table 4). If the distribution of fishing effort remains constant and fishing effort increases threefold per 5° grid cell, seabird bycatch could increase to approximately 7 580 birds per year (5 100 White-chinned Petrels, 1 840 shy-type albatrosses, 380 Black-browed Albatrosses and 260 yellow-nosed albatrosses). This further highlights the need to address low compliance and may warrant the introduction of further measures to curb this mortality (Chapter 10).

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Chapter 2

TURTLE BYCATCH IN THE PELAGIC
LONGLINE FISHERY
OFF SOUTHERN AFRICA

TURTLE BYCATCH IN THE PELAGIC LONGLINE FISHERY OFF SOUTHERN AFRICA

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ABSTRACT

Capture by pelagic longline fisheries has been identified as a key threat to turtle populations. This study represents the first assessment of turtle bycatch in the South African pelagic longline fishery for tunas and Swordfish. A total of 181 turtles were caught on observed sets, 1998–2005, at a rate of 0.04/1000 hooks (0–15.5/1000 hooks, SD 1.28). Loggerhead Turtles *Caretta caretta* comprised 60.0% of the total turtle capture and were caught at rate of 0.02/1000 hooks. The second most commonly caught species was the Leatherback Turtle *Dermochelys coriacea* (33.8%) which were caught at rate of 0.01/1000 hooks. Five Hawksbill Turtles *Eretmochelys imbricata* were reported caught at a rate of 0.001/1000 hooks and three Green Turtles *Chelonia mydas* at a rate of 0.001/1000 hooks. Catches were clustered, with 70% of turtles caught on 1% of sets. Apart from one set occurring on the Agulhas Bank, all sets that caught three or more turtles occurred on the Walvis Ridge and on the shelf edge north of the Orange River (25–31 ºS and 0–15 ºE). Most of the variance in turtle bycatch was accounted for by 'vessel'. Five of 50 vessels were responsible for catching 65% of turtles at a rate of 0.4/1000 hooks. The target species (Swordfish *Xiphias gladius* or tunas *Thunnus* spp) was the second most important explanatory variable; 89.5% of turtles were caught by swordfish directed vessels at a rate of 0.15/1000 hooks. Season was the third most important explanatory variable, with more turtles caught between January and June (0.13/1000 hooks) than in the remainder of the year (0.03/1000 hooks), although Leatherback Turtles tended to be caught throughout the year. Extrapolations based on stratification by 5° grid cell, by season and by target species estimated that a total of 190 turtles caught per year (approximately 100 Loggerheads and 50 Leatherbacks). Three techniques were investigated and extrapolations varied between 190 and 560 turtles per year depending on technique. An increase in fishing effort to 50 rights holders is likely to increase turtle bycatch to approximately 770 turtles per year. Leatherback Turtles caught by the South African pelagic longline fisheries are likely to be from the South African nesting population. This population has been protected at its nesting beaches but has not recovered as expected. The overlap of turtle tracks and fishing effort suggest that this fishery could be partially responsible.

INTRODUCTION

Six of the world's seven species of turtle are Endangered (IUCN 2007) due to impacts both on land and at sea (Lutcavage *et al.* 1997, Frazier 2003). Many breeding sites are now protected, but some populations have failed to recover. This has, at least in part, been attributed to their incidental capture at sea (Hillestad *et al.* 1982, Lutcavage *et al.* 1997, Watson *et al.* 2005). Turtles are caught in a variety of fishing gear, including trawl nets (Magnuson *et al.* 1990, Fennessy *et al.* 1994), drift nets (Silvani *et al.* 1999) and gill nets (Julian & Beeson 1998). Considerable effort has been made to reduce turtle bycatch in trawl fisheries with Turtle Excluder Devices (TEDs) (Brewer *et al.* 1998, Fennessy & Isaksen 2007), but until recently relatively little attention has focused on the impact of longline fisheries on turtles (Witzell 1999, Lewison *et al.* 2004, Carranza *et al.* 2006).

Pelagic longline fisheries for tunas *Thunnus* spp and Swordfish *Xiphias gladius*, incidentally catch turtles (Lewison *et al*. 2004). Limited data are available to evaluate turtle bycatch worldwide (Lewison *et al.* 2004). Unlike landing records for target species, bycatch monitoring relies on onboard observers or on fisher logbooks because turtles are seldom returned to port. Several nations deploy observers on their fishing fleets, but total observer effort is low (Lewison *et al.* 2004). Regional Fisheries Management Organisations, such as the International Commission for the Conservation of Atlantic Tunas (ICCAT) and the Indian Ocean Tuna Commission (IOTC), recently have passed resolutions and/or recommendations on turtle bycatch and established bycatch working groups. However, their work has been hampered by limited data, especially on high seas (ICCAT 2003, IOTC 2005).

South Africa provides a good opportunity to investigate turtle bycatch because it has both a swordfish and tuna directed pelagic longline fleet allowing for comparisons between these two techniques. There has been an observer programme in these fisheries since 1998. South Africa is also an important region for turtle conservation. Five species occur within South African waters (Hughes 1989), all of which are Endangered or Critically Endangered (IUCN 2007). South African waters are important developmental areas for Green *Chelonia mydas* and Hawksbill *Eretmochelys imbricata* Turtles, whereas Olive Ridley Turtle *Lepidochelys olivacea* are occasional migrants to South African waters (Hughes 1974). Loggerhead *Caretta caretta* and Leatherback *Dermochelys coriacea* Turtles nest along the South African northern coast of KwaZulu-Natal where they have been protected since 1965 (Hughes 1974). The shore-based monitoring of these populations is one of the longest quantitative Loggerhead and Leatherback Turtle monitoring programmes, with ongoing studies since protection (Hughes 1974, Hughes 2001, Wright 2004). Initial population increases for both species were attributed to the protection of nesting sites. Numbers of turtles breeding in the monitoring area increased from the 1960s to the 1970s, but subsequent recovery of the Leatherback population suggests that mortality may be occurring elsewhere (Wright 2004). This study investigates the extent to which incidental mortality by

South African pelagic longline fisheries is contributing to these observed trends, and presents the first complete assessment of turtle bycatch in this fishery.

METHODS

Data were collected by fisheries observers on board South African and Asian flagged pelagic longline vessels from 1998 to 2005. This information included turtle bycatch (species, number and status), as well as gear (number of hooks, length of mainline, etc.) and operational (time of set, position, etc.) information. These vessels were licensed to fish within South Africa's Exclusive Economic Zone (EEZ) as well as on the high seas.

Identification of species was made by observers at sea (Petersen & Honig 2005). Observers received limited training in species identification prior to 2005. The identification of hard-shelled turtles is therefore unreliable, but because Leatherback Turtles are unique in their soft-shell their identification is likely to be more robust. Catch rates are reported as numbers of turtles per 1000 hooks, summarised by vessel, region, season and flag. The three techniques we used to calculate total catch were 1) basic extrapolation of total fishing effort multiplied by an average annual catch rate, 2) building on the basic extrapolation to incorporate the effect of vessel flag and 3) adding the effect of season and area. This was done based on the method used by Lewison and Crowder (2003) and Camiñas *et al.* (2006):

$$\hat{C}b = \sum rsf \, (Cbrsf / Eorsf) * Edrsf$$

where $\hat{C}b$ = Estimated total bycatch of a species, b
Cbrsf = Observed bycatch of a species, b within region, r, season, s and flag, f
Edrsf = Number of hooks deployed, d, within region, r, season, s and flag, f
Eorsf = Number of hooks observed, o, within region, r, season, s and flag, f
b = Bycatch species or group of species
r = Region (one degree grid cell)
s = Season i.e. summer = November–April, winter = May–October
f = Flag i.e. Asian or South African.

Numbers of turtles caught on each set were modelled using a generalised linear model with a Poisson distribution and logarithmic link function using Genstat 9 (McCullagh & Nelder 1989, Genstat Committee 2007). Model selection was made using change of deviance considerations using Akaike's Information Criterion (AIC) (Quinn & Keough 2002). The logarithm of the parameter λ of a Poisson distribution was modelled as a linear combination of explanatory variables: e.g. for three explanatory variables, log $\lambda = a + b_1x_1 + b_2x_2 + b_3x_3$. Explanatory variables investigated included year, season, area, vessel name, target species, moon phase, branch line, bathymetry, bait type, Beaufort scale and time of set. The number of hooks per set was used as an offset variable.

Satellite transmitters (Platform Transmitter Terminals (PTTs)) were deployed on 10 post-breeding Leatherback Turtles during nesting on the northern KwaZulu-Natal coast during 1999–2006. Seven were deployed from late January to early February, at the end of the nesting season, and three were deployed in early December, in the early nesting season. The devices were attached using a harness made of elastic cord (Hughes *et al.* 1998). Their movement and diving behaviour has been published in relation to oceanography (Luschi *et al*. 2003, 2006). In this paper we investigate Leatherback distribution in relation to fishing effort and bathymetry. Fishing effort data were obtained from vessel logbooks submitted to Marine and Coastal Management, Department of Environmental Affairs and Tourism, the government department responsible for managing fisheries in South Africa. Etopo-2 bathymetry data (Sandwell 1990) were obtained from the National Geophysical Data Center (NGDC).

Tracking data were filtered through speed (maximum velocity was set at 10 km.h^{-1}) and ARGOS quality (all positions with a quality code of Z were excluded) filters. The data were further filtered to remove inter-nesting activity to investigate post-breeding activity. Spatial analyses (including kernel home range and density distribution maps) were made using Arcview 3.2. The overlap of Leatherback tracks and longline fishing effort was investigated using kernel home range analysis which calculates a utilization distribution (based on Worton 1989) as a grid coverage, using a least squares cross validation (Silverman 1986). Kernel density plots have been successfully used in tracking studies to quantify habitat use (e.g. Wood *et al.* 2000, BirdLife International 2004). Density distributions are represented on the maps by Utilization Distributions (UD) which provide probability contours indicating the relative time the Leatherback Turtles spent in a particular area. For example, they spent half of their time within a 50% utilization distribution contour and thus represent 'hot spots' where turtles congregate or spend more time.

RESULTS

The South African pelagic longline fishery comprises of two sectors, namely a swordfish and a tuna directed sector. These two sectors use different gear configurations and practises (refer to General Introduction, Chapter 1). The swordfish sector set a total of 10 million hooks from 1998 to 2005, 10% of which were observed. Tuna directed Asian vessels set a total of 41.5 million hooks from 1998 to 2005, 8% of which were observed.

Turtle bycatch

Bycatch data and nine explanatory variables (vessel name, target species, month, year, season, area, bathymetry, bait type and branch line length) were available for 2 256 sets. A total of 181 turtles were caught on observed sets, 1998–2005 at a rate of 0.04/1000 hooks (0–15.5/1000 hooks, SD 1.3). There was no annual trend over the time period, but catch rates peaked in 2002 (0.33/1000 hooks), when swordfish directed fishing effort was also at

its highest (2.5 million hooks). Four species of turtle were caught (Table 1). Most turtles caught (95%) were released or discarded; only 5% were retained. Most (84%) were reported to be released alive, but post-release survival is unknown. In 46% of cases, turtles were released with the hook still in the animal. Most swallowed the hook, but half the Leatherbacks were foul hooked on the body.

Table 1: Species composition and catch rates of turtles caught on South African pelagic longliners, 1998–2005.

Species	No. caught	%	Catch rate	Maximum	SD
Loggerhead	78	43.1%	0.018	15.50	1.30
Leatherback	44	24.3%	0.010	0.90	0.10
Hawksbill	5	2.8%	0.001	0.21	0.02
Green	3	1.7%	0.001	0.11	0.01
Unidentified	51	28.2%	0.011	1.20	0.10
Total	181	100.0%	0.040	3.58	0.31

Total, Loggerhead and Leatherback turtle bycatch were each modelled separately using generalised linear models (GLM), with models accounting for 55.0%, 21.8%, 11.0% of variance, respectively (Table 2). There were insufficient data to model Hawksbill and Green Turtle bycatch separately. The variables year, bait type, branch line length, bioregion, ocean and bathymetry increased the AIC and were not significant in all models and were excluded.

Table 2: Summary of the percentage variance explained from GLM analysis of total turtles, Loggerhead and Leatherback Turtles.

Parameter	Total	Loggerhead	Leatherback
Vessel	34.0	} 15.4*	} 11.0*
Target species/vessel flag	16.3		
Season	4.7	6.4	
Total	55.0	21.8	11.0

* The model produced the same fit whether vessel or target species were included, but no additional deviance was explained by adding both.
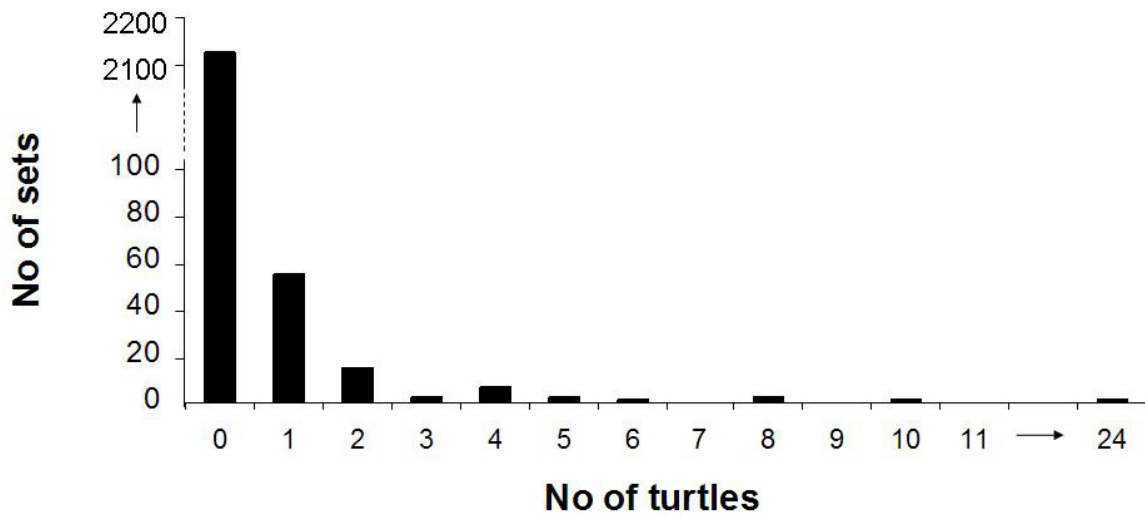
Figure 1: Frequency distribution of turtle bycatch per set in the South African pelagic longline fishery, 1998–2005.

Turtles were only caught on 3.8% of sets (n=85). Catches were clustered, with 70% of turtles caught on 1% of sets; one set caught 24 turtles, one caught 10 and two sets caught eight (Fig. 1). All sets (except one occurring on the Agulhas Bank) where three or more turtles were caught occurred on the Walvis Ridge between and on the continental shelf edge north of the Orange River (25–31 ºS and 0–15 ºE) (Fig. 2). Loggerheads were predominantly caught in the Atlantic Ocean (91.0%) especially in the above mentioned area (25–31 ºS and 0–15 ºE) at a maximum rate of 15.5/1000 hooks (Fig. 2b). Leatherback Turtles were caught in equal numbers in the Indian and the Atlantic Oceans (54% in the Atlantic, 46% in the Indian Ocean n=44), but highest catch rates were also recorded in the area between 25–31 ºS and 0–15 ºE and reached a maximum of 0.9 Leatherbacks/1000 hooks (Fig. 2c). Of the five Hawksbill Turtles caught, four were caught in the Atlantic (two on the Walvis Ridge) and one in the Indian Ocean (Fig. 2d). Two of the three Green Turtles were caught on the Walvis Ridge and one in oceanic waters off KwaZulu-Natal (Fig. 2e). Loggerheads were caught in water with an average depth of 3 300 m (500–4 670 m) whereas Leatherbacks were caught in shallower water with an average water depth of 2 300 m (300–5 000 m). Hawksbill and Green Turtles were caught in oceanic waters of > 1500 m deep (Fig. 2b-e).

Most of the variance in turtle bycatch (34.0%, $p<0.001$) was accounted for by individual vessel (Table 2). Five of 50 vessels were responsible for catching 65% of turtles at a rate of 0.4/1000 hooks (four for 61% or 0.43/1000 hooks, three for 54% or 0.5/1000 hooks, two for 46% or 0.63/1000 hooks and one for 37% of turtles or 0.9/1000 hooks). These five vessels all targeted Swordfish and predominantly fished in the Atlantic Ocean during the austral Summer. The target species was the second most important explanatory variable, accounting for 16.3%, 15.4% and 11% variance ($p<0.001$). Most turtles (89.5%) were caught

by swordfish directed vessels at a rate of 0.15/1000 hooks (n=162) (Table 3). Tuna directed vessels caught 10.5% of turtles, at a rate of 0.006/1000 hooks (n=19). This finding was consistent for both Loggerhead and Leatherback Turtles, where 97% and 82% were caught by swordfish directed vessels (Table 3). Four of five Hawksbill and all Green Turtles were caught by swordfish directed vessels.

Season was the third most important explanatory variable for Loggerhead Turtles (p<0.001) and total turtles (p<0.001), but was not significant for Leatherback Turtles (p=0.99) (Table 2). Loggerhead Turtle catch rates peaked in February (64% of mortality, 1/1000 hooks). Despite not being statistically significant, catch rates for Leatherback Turtles were higher January to June, when they were caught at a rate of 0.04/1000 hooks, compared to 0.01/1000 hooks in the remainder of the year. In general significantly more turtles were caught between January and June (0.13/1000 hooks) than in the remainder of the year (0.03/1000 hooks).

Simply extrapolating the average catch rate (0.04/1000 hooks) to the total fishing effort predicted that approximately 570 turtles were caught per year (range 20–1250) (Table 4). When the extrapolation took vessel flag into account the annual average estimate decreased to 260 turtles per year (range 20–1030) (Table 4). Based on stratification by five degree grid cell, by season and by target species we extrapolated that a total of 190 per year (100 Loggerhead and 50 Leatherback) were caught by the pelagic longline fishery (Table 4).

Figure 2/...
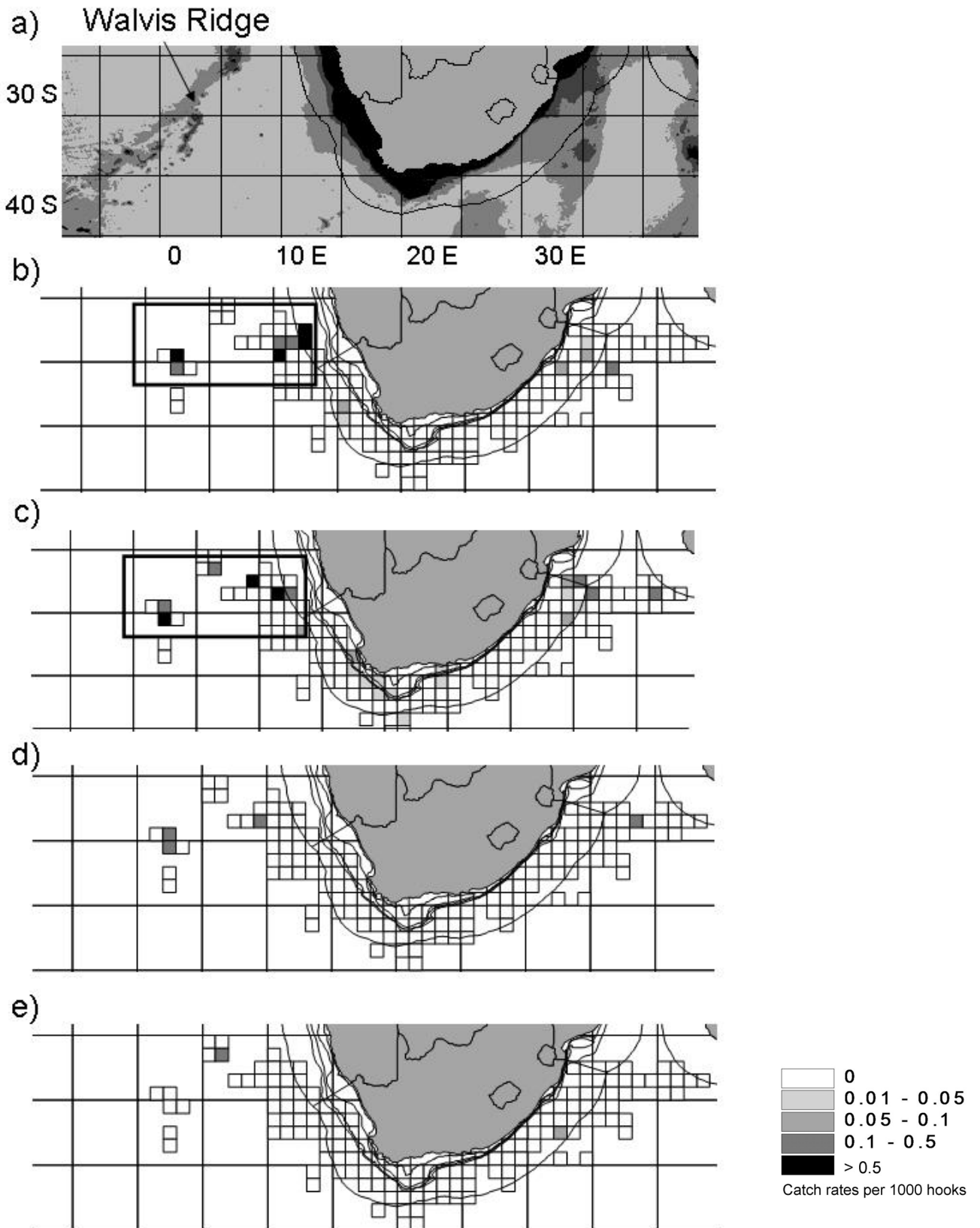
Figure 2: Map of the south east Atlantic and south west Indian Oceans depicting 1 000 m, 2 000 m and 4 000 m isobaths and the location of the Walvis Ridge (a) and the distribution by 1° grid cell of Loggerhead b), Leatherback (c), Hawksbill (d) and Green e) Turtle catches on pelagic longlines observed by fishery observers off South Africa, 1998–2005. Box shows area of high catch rates along the Walvis Ridge and off the continental shelf edge north of the Orange River.

Table 3: Summary of Loggerhead and Leatherback turtle catch rates stratified by 5° grid cell, season and target species and total extrapolated capture off southern Africa.

Species	Loggerhead Turtles	Leatherback Turtles
Tuna Jan – June	0.006	0.003
Tuna July – Dec	0.003	0.002
Average Tuna	0.004	0.02
Swordfish Jan – June	0.15	0.08
Swordfish July – Dec	0.03	0.02
Average Swordfish	0.05	0.08
Total catch rate	0.02	0.01
Total capture (1998-2005)	780	430
Average capture per year (1998 - 2005)	100	50
Estimated capture of 30 tuna vessels*	85	50
Estimated capture of 20 swordfish vessels*	375	200
Total (20+30 vessels)	460	250

*average of 200 sets per year

Table 4: Summary of total turtle capture in the South African large pelagic longline fishery using a basic extrapolation of a single catch rate applied to the total effort, an extrapolation by flag and an extrapolation stratified by 5° grid cell, season and flag, 1998–2005.

	Basic	Extrapolation by flag			Extraplation by flag, area and season		
	Total	Asian	SA	Total	Asian	SA	Total
1998	1146	19	0	19	74	19	93
1999	39	0	52	52	42	26	68
2000	858	109	0	109	70	142	213
2001	926	222	308	530	39	204	242
2002	1246	899	131	1030	9	335	344
2003	246	246	0	246	0	284	284
2004	46	52	0	52	11	162	173
2005	23	30	14	44	18	75	94
min	23	0	0	19	0	19	68
max	1246	899	308	1030	74	335	344
Average	**566**	197	63	**260**	33	156	**189**

Overlap of Leatherback tracks and fishing effort

Overlaying Leatherback turtle tracks with fishing effort revealed that there is substantial overlap, particularly on the east and south coast (Fig. 3). This may be an artefact of sampling because all devices were deployed at nesting beaches on the east coast. PTTs transmitted data for a total of 1 406 days and an average of 108 days per turtle (range 16–241, SD 64) and it may be that devices failed prior to Leatherbacks reaching the west coast. Despite this, at least one of the 10 tracked Leatherbacks moved northward along the west coast into Namibian waters indicating overlap of fishing effort and Leatherback Turtle movements on the west coast. Fishing effort also took place in a similar depth stratum to that of Leatherback Turtles movements (Fig. 4). There was no significant difference between the mean bathymetric depth of longline fishing effort and that of the Leatherback Turtle distribution (t=-0.59 p=0.563). Fishing effort peaked between 800–1 000 m. Leatherbacks spent most of their time in two depth strata. The first matches with that of the highest fishing effort, 800–1 000 m, which coincides with high Leatherback mortality. The second is in the depth strata 3 000–4 000 m, but is not an area of high longline fishing effort.
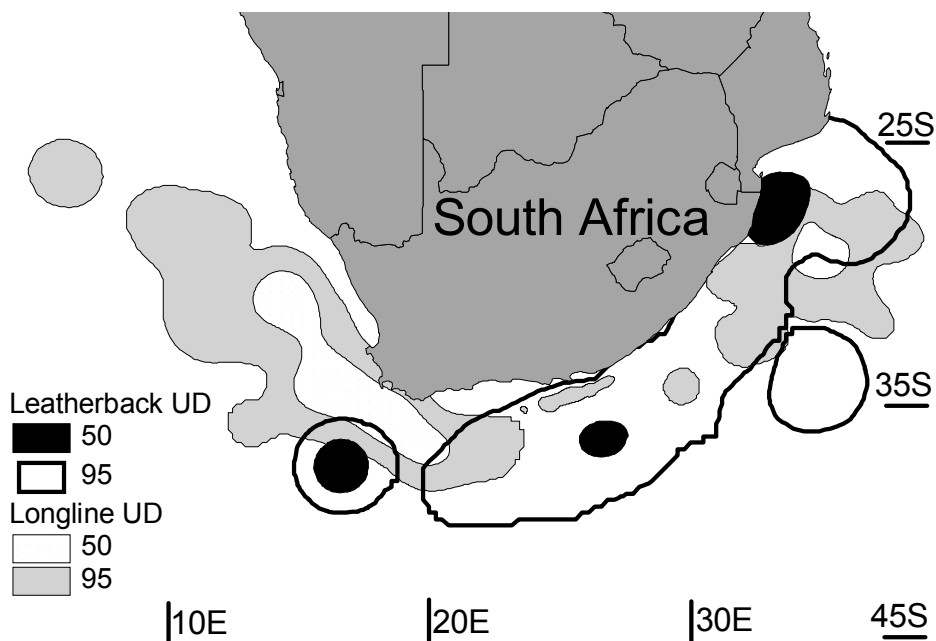

Figure 3: Map depicting 50% and 95% utilization distribution of Leatherback turtle tracks and South African pelagic longline sets, 1998–2005.
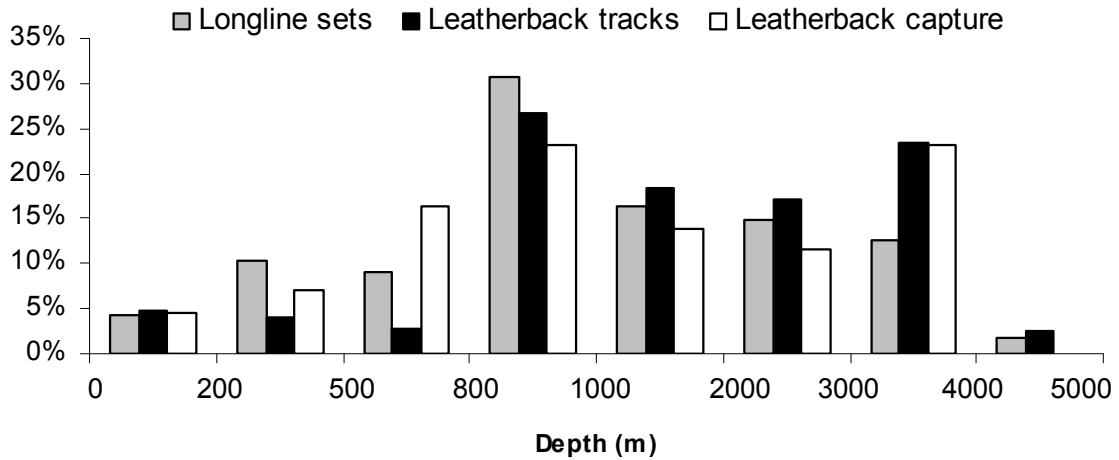
Figure 4: A bar graph of the proportional distribution of longline sets, time spent by Leatherbacks (from satellite tracking data) and location of Leatherback catches (from observer data) off southern Africa, across the bathymetric depth range, 0–5 000 m.

DISCUSSION

Magnitude of bycatch

At least four of the five species of turtles occurring in South African waters were caught by the South African pelagic longline fishery. Turtle catch rates (0.04/1000 hooks) were considerably lower than catch rates in pelagic longline fisheries reported elsewhere (0.2–6.5/1000 hooks) (Table 5). This may simply be a consequence of lower turtle abundances. It could also be that catches were underestimated. Fisheries observers were not trained to collect turtle bycatch data, nor were their data sheets designed for this purpose until 2005. This is considered to have had a relatively small effect because most of the data (73%) are from this year (2005) and catch rates did not increase post training. Also, turtle bycatch is a rare, non-random event and sampling may have been inadequate to describe turtle bycatch accurately. This finding does however suggest that turtles did not occur in high abundance throughout South African waters and were more likely to be concentrated in areas of high productivity, such as the Walvis Ridge and Agulhas Bank (Nelson & Hutchings 1983).

Table 5: Summary published studies of global turtle catch rates.

Catch rate per 1000 hooks	Date	Region	Turtle species	Sample size	Reference
0.04	1995-2005	Southern Africa	Mainly Loggerhead and Leatherback	2256 sets or 4.4 million hooks	This study
0.20	1992-1995	North Atlantic	All species	18 million hooks	Witzell 1999
0.39	2003	The Gulf of Guinea and St Helena	Leatherback	79 sets	Carranza *et al.* 2006
0.91	1994-2004	Spanish Mediterranean	Loggerhead	1546 sets	Camiñas *et al.* 2006
1.50	1996-1999	Brazil	Loggerhead, Leatherback and Olive Ridley	41 sets	Pinedo & Polacheck 2004
2.40	2000	Global	Loggerhead and Leatherback	unclear	Lewison *et al.* 2004
3.50	2000	Atlantic	Loggerhead and Leatherback	unclear	Lewison *et al.* 2004
4.60	2002	Balearic Islands	Loggerhead	162 interviews	Carreras *et al.* 2004
6.50	2000	Indian	Loggerhead and Leatherback	none	Lewison *et al.* 2004

There has been global recognition that the threat facing turtles has shifted from land to the sea (Spotila *et al.* 2000, Lewison *et al.* 2004). The bycatch of Leatherback and Loggerhead Turtles, the two species most commonly caught off southern Africa, have been recorded elsewhere in the world (Chan *et al.* 1988, Chaloupka & Limpus 2001, Pinedo & Polacheck 2003, Carreras *et al.* 2004, Lewison *et al.* 2004). Globally an estimated 200 000 Loggerheads and 50 000 Leatherbacks are killed in pelagic longline fishing gear each year (Lewison *et al.* 2004). The estimated 560, 260 or 190 turtles caught per year (depending on extrapolation technique) accounts for approximately 0.08%, 0.1% and 0.2% of the global bycatch estimate for Loggerhead and Leatherback Turtles, respectively (Lewison *et al.* 2004). This global estimate lacked data from the south west Atlantic and the Indian Ocean (Lewison & Crowder 2003, Lewison *et al.* 2004). Thus data from the present study could be used to refine this estimate and is likely to reduce the global estimated mortality by Lewison *et al.* 2004, substantially.

Factors affecting bycatch

Turtles were caught primarily by the swordfish directed fishery. Similar findings were reported by Crowder and Myers (2001) and Gilman *et al.* (2007). This is partly the result of the swordfish fishery setting relatively shallow gear. Turtles spend most of their time within 100 m of the surface (Polvina *et al.* 2002). Leatherback Turtles dive to an average depth of 200 m and exhibited diel variations in their diving activity, with longer dives at night (Sale *et al.*

2006). Loggerhead Turtles have been reported to spend 40% of their time within 1 m of the surface (Polvina *et al.* 2002). Virtually all dives of two Loggerhead Turtles monitored off Japan were shallower than 30 m (Sakomoto *et al.* 1993). Therefore turtle bycatch could be reduced by setting gear deeper (Crowder & Myers 2001, Polivina *et al.* 2002, Shiode *et al.* 2005, Gilman *et al.* 2007).

The higher capture rates in the first part of the year overlaps with Loggerhead and Leatherback Turtle nesting season in South Africa (October–February) (Wright 2004). The fact that Leatherback Turtles are caught throughout the year supports the limited available evidence that they remain within South African waters year round (Luschi *et al.* 2003, 2006). Catches were not evenly distributed amongst vessels, with a handful being responsible for the majority of recorded mortality. Similar findings have been reported for seabirds (Klaer & Polacheck 1998, Chapter 1). By working closely with the skippers of these vessels it may be possible to understand and mitigate the specific cause.

Given the importance of target species/vessel flag, season and area, the third extrapolation technique which takes these into consideration is likely to be the most accurate and therefore the most reliable estimate of total mortality is 190 turtles per year. Given the proposed increase in fishing effort to 50 rights holders, fishing effort is estimated to potentially increase threefold from an average of 6.4 million hooks per year (1998–2005) to 20 million hooks per year (average of 200 sea days and 2000 hooks per set). If we assume the distribution of fishing effort remains constant and we increase fishing effort threefold per 5 degree grid square we estimate that sea turtle bycatch could increase to approximately 770 sea turtles per year (460 Loggerhead and 250 Leatherback Turtles).

Impact on key species

Loggerhead Turtles, the most commonly caught species is Endangered and the Leatherback Turtle Critically Endangered. Population declines have been observed for both species. For example, Pacific Ocean populations of both species have declined by up to 95% in the past 20 years (Spotila *et al.* 2000, Lewison *et al.* 2004). The global nesting population estimate for Leatherback Turtles is between 26 000–43 000 down from 115 000 estimated in 1980 (Spotila *et al.* 1996, 2000). Similar population trends have been reported for Loggerhead Turtles (Ilgaz *et al.* 2007). Adult mortality from fishing activity is thought to be driving these global declines (Spotila *et al.* 1996, 2000).

Locally, Loggerhead and Leatherback nesting populations increased at a rate of 7% and 8% per year 1965 to 1975 after nesting sites were protected (Hughes 1974, Wright 2004). There was some fluctuation in nesting population numbers in the following three decades (1975–2005), but in general, numbers remained relatively stable (Wright 2004). The initial recovery rate of South African turtle populations was well within that recorded by other recovering turtle populations, however the recovery rate over the full time period (1.7% and 1.4% respectively, 1965–2005) is lower than expected. Protected nesting populations of Green

Turtles increased by 4–14% per year in six major colonies located through the Pacific and Atlantic Oceans, after conservation measures were introduced and enforced (Chaloupka & Limpus 2001, Balazs & Chaloupka 2004, Troëng & Rankin 2005). The Green Turtle population in Hawaii recovered at a rate of 5.4% per year (1973–2004) and those at Europa and Grande Glorieuse Islands (south western Indian Ocean), at 3% and 6% per year (1986–2006) respectively, after conservation measures were put in place (Chaloupka & Balazs 2007, Lauret-Stepler *et al.* 2007). Variability in population growth rates have been attributed to a number of factors, including the extent to which the nesting population was exploited, the distance between feeding and nesting grounds and adult mortality at sea (Broderick *et al.* 2006). The slow recovery of Leatherback Turtles nesting in South Africa post 1975 may be the result of at-sea mortality (Wright 2004).

Is the South African pelagic longline fishery responsible for slow recovery rates?

Loggerhead Turtles are known to travel great distances (Polivina *et al*. 2002, Hawkes *et al.* 2006) and are frequently caught in longline fisheries globally (Spotila 2004, Carreras *et al.* 2004, Lewison *et al.* 2004). Loggerhead Turtles from the South African nesting population remained in shallow shelf waters and moved northward into the western Indian Ocean post-breeding (Luschi *et al.* 2006). They are therefore only at risk of capture from the South African fishery during breeding. It is more likely that Loggerhead Turtles caught on the Walvis Ridge come from populations breeding on the west coast of Africa (Brongersma 1961, 1982, Carr & Carr 1991).

Leatherback Turtles caught by the South African pelagic longline fishery on the other hand are likely to be from the South African population because it is the only known nesting population in the western Indian Ocean, although limited evidence suggests that some may move south from breeding localities in Angola or Gabon (Carr & Carr 1991, Fretey 2007). It is unlikely that Leatherback Turtles are attracted to pelagic longline vessels, because they do not typically feed on squid and fish, preferring a diet of gelatinous planktonic prey (Den Hartog 1980, Bjorndal 1997). This is consistent with Leatherbacks being entangled in longline gear rather than hooked in the mouth (Bolten & Bjorndal 2005, Gilman *et al.* 2007, this study). Interaction with pelagic longline gear is likely to be the result of overlap in distribution. The distribution of Leatherback Turtles in South African waters is determined by a combination of suitable feeding areas and major surface currents (Luschi *et al*. 2003, Sale *et al.* 2006). Most tracked Leatherbacks moved south along the continental shelf break to the Agulhas Bank, one continued up the west coast to Namibia, one travelled north to Walter's Shoal, a pinnacle south of Madagascar and one moved as far as the southern Atlantic Ocean along the sub-tropical convergence (n=10) (Luschi *et al.* 2003, 2006, MCM unpublished data). Pelagic longline fishing effort also concentrates on the Agulhas Bank and the continental shelf break, although more so on the south and west coast than the east coast, where it moves further offshore.

There is also likely to be vertical overlap between fishing effort and Leatherback turtle behaviour. Leatherback Turtles tend to dive deeper at night. This pattern is thought to derive from turtles foraging within the deep scattering layer (Eckert *et al.* 1989, Hays *et al.* 2004, Sale *et al.* 2006), which consists of zooplankton that makes diurnal vertical movements in response to light levels, moving closer to the surface at night but descending below 600 m during the day (Eckert *et al.* 1989). Swordfish also forage at the deep scattering layer (Brill & Lutcavage 2001). South African pelagic longline fishery typically soaks (term for the time fishing gear is in the water) their gear during the day. Therefore it may be possible to reduce Leatherback catches by reducing daylight soak time.

Substantial overlap therefore exists between Leatherback movements and South African pelagic longline fishing effort. The majority (84%) of the estimated 190 turtles (50 Leatherbacks) caught each year were alive, and although post release mortality of turtles may occur, it is considered to be low and long-term survival estimated to be 50% (Aguilar *et al.* 1995). The resulting estimate is that approximately 25 Leatherback Turtles are killed each year by the South African pelagic longline fishery. The current annual nesting population in KwaZulu Natal is 200 (EKZN unpublished data). This includes nesting females only with a mean inter-nesting interval of three years, thus roughly a total population of 1 200 individuals (if we assume a 50:50 sex ratio) (EKZN unpublished data). Therefore 25 individuals per year accounts for 2% of this population and could be hampering their recover in South Africa.

Cumulative effect of multiple fisheries

Turtles are also captured by other fisheries in the region such as the purse seine, shark nets, shrimp trawl and pelagic trawl fisheries (Dudley & Cliff 1993, FAO 2007, Fennessy & Isaksen 2007). Mortality has been documented in these fisheries in other regions (Hillestad *et al*. 1982, Magnuson *et al*. 1990, Pandav *et al*. 1997, Silvani *et al*. 1999). However, the level of mortality caused by these fisheries off southern Africa is less well understood. At present, pelagic purse seine targeting sardine *Sardinella* spp and trawl fisheries for horse mackerel *Trachurus* spp operate in South African waters and could impact turtles, although preliminary investigation reveals that these fisheries are unlikely to catch large numbers (Nel *et al*. 2007, FAO 2004). Global shrimp trawl fisheries have been estimated to kill up to 55 000 turtles each year (Magnuson *et al*. 1990). Turtle bycatch has been recorded in South Africa's trawl fishery targeting shrimps (Fennessy & Isaksen 2007). Although this fishery is small (two vessels are active at present) it operates along the north coast of KwaZulu-Natal in the vicinity of breeding localities and thus it could be capturing significant numbers of turtles. At present this fishery does not employ the use of Turtle Excluder Devices (TEDs); however, research is underway to investigate implementation (FAO 2007). The extent to which these fisheries are contributing to the slow recovery rates is unknown.

Given the conservation status of Leatherback Turtles and the fact that they largely remain within South Africa's EEZ, Marine and Coastal Management, the South African agency for

managing marine resources, could consider implementing mitigation measures to reduce this interaction. Such mitigation measures will also reduce the bycatch of other endangered turtles, such as Loggerheads. Since Loggerhead Turtles caught by the South African pelagic longline fishery are unlikely to solely come from the South African breeding population, the success of these measures cannot be measured by their status alone. South Africa nevertheless has a responsibility to curb this mortality. For South African Loggerhead Turtles, which are wider ranging in their non-breeding season, the responsibility largely lies with regional fisheries management organizations such as the Indian Ocean Tuna Commission to implement management measures.

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Chapter 3

*PELAGIC SHARK BYCATCH IN THE
PELAGIC LONGLINE FISHERY
OFF SOUTHERN AFRICA*

PELAGIC SHARK BYCATCH IN THE PELAGIC LONGLINE FISHERY OFF SOUTHERN AFRICA

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ABSTRACT

Southern African waters are an interesting region to study the viability of shark capture because of the diversity of vulnerable species and the existence of three pelagic longline fleets which catch sharks either as a directed or non directed catch. A total of 20 species was caught as bycatch by the South African pelagic longline fleets targeting tuna *Thunnus* spp. and Swordfish *Xiphias gladius*, 1998-2005, six of which are considered Vulnerable and one (Scalloped Hammerhead *Sphyrna lewini*) Endangered. Blue *Prionace glauca* and Short-finned Mako *Isurus oxyrinchus* Sharks were the most commonly caught species (69.2% and 17.2% respectively of sharks caught). Swordfish directed vessels caught more sharks (11.7 Blue Sharks/1000 hooks and 1.4 Short-finned Mako Sharks/1000 hooks) compared with tuna directed vessels (1.8 Blue Sharks/1000 hooks and 0.9 Short-finned Mako Sharks/1000 hooks). The catch per unit effort (CPUE) of Blue Sharks and Short-finned Mako Shark decreased from 2001 and 2000 respectively. This was accompanied by a decrease in length frequency for both species, 2002–2007. The proposed increase in fishing effort to 50 rights holders (targeting tunas and Swordfish), could increase the fishing effort threefold from 6.4 to 20 million hooks per year. If the distribution of fishing effort remains constant shark bycatch could increase to approximately 134 000 or 3 000 t of sharks per year.

INTRODUCTION

The world's oceans may have lost more than 90% of their large predatory fish as a result of both directed catch and bycatch (Myers & Worm 2003). Reviewing the history and current status of shark fisheries worldwide, Barker and Schluessel (2005) report that shark bycatch in high-seas fisheries amounted to 50% of the total chondrichthyan catch worldwide. Numerous studies have shown the inability of sharks to sustain high levels of harvesting (Baum *et al.* 2003, Baum & Myers 2004, Beerkircher *et al.* 2002). Nearly all of the 26 shark species for which at least 10 years of catch or landing data have been submitted to the FAO, showed large population declines (Castro *et al.* 1999). Sharks are generally long-lived, slow growing, mature late and display low fecundity (Hoenig & Gruber 1990). These life history traits result in a low resilience to fishing mortality (Musick *et al.* 2000, Kitchell *et al.* 2002, Schindler *et al.* 2002, Barker & Schluessel 2005). As top predators with no natural enemies, sharks are keystone species in many marine ecosystems (Beddington 1984, Baum & Myers 2004). Changes in shark populations are therefore likely to have profound ramifications throughout the food web (Stevens *et al.* 2000, Megalofonou *et al.* 2005). Sharks may help to stabilize ecosystems by reducing variability of other stocks (Stevens *et al.* 2000).

Historically, sharks had little economic value compared to teleost fisheries and were largely ignored by fishery management agencies (Rose 1998, Barker & Schluessel 2005). As a result, few accurate data exist on the catches and landings of sharks, and species level information is almost completely lacking (Castro *et al*. 1999, Barker & Schluessel 2005). Prior to the 1980s, sharks were usually discarded to save freezer space for more valuable target species (Weber & Fordham 1997). However, the demand for shark fins, which do not require refrigeration space and take up minimal storage, has increased in Asia (Prestowitz 1996, Shivji *et al.* 2002). This had lead to sharks being targeted for their fins (Bonfil 1994). Shark finning, the removal and retention of shark fins and discarding the remainder of the carcasses at sea, is considered a wasteful practice that contravenes the responsible fishing ethos embraced by the Food and Agricultural Organisation (FAO) Code of Conduct for Responsible Fisheries (FAO 1995) and resolutions from a variety of international fisheries bodies (e.g. ICCAT 2004a, IATTC 2005, IOTC 2005, SEAFO 2006). As target stocks collapse or are seasonally unavailable, many fisheries retain more of their shark bycatch (Rose 1998). This increase in shark exploitation and their known vulnerability have not, in most cases, been translated into fisheries' managers placing priority on managing these stocks effectively (Barker & Schluessel 2005).

The FAO has called for International and National Plans of Action (IPOA/NPOAs) to be developed and adopted to specifically address the conservation and management of sharks (FAO 1999). However, despite this initiative, problems remain. Comprehensive shark management plans are largely confined to developed countries, but more than two thirds of reported shark landings are from developing countries (Barker & Schluessel 2005). Of these, less than 5% are identified to species level. In many cases, developing countries lack the

resources and capacity to effectively and appropriately manage their fisheries. In Africa especially, the impact of fisheries on shark populations is not well documented. Kroese and Sauer (1998) estimated that at least 95 000 t of sharks were landed per year in Africa, but this is likely to be a gross underestimate due to limited data sources and a lack of trained onboard observers. In addition, many high seas fisheries operate largely unrestricted. The lack of effective international mechanisms and regulations worldwide to effectively manage shark resources requires addressing (Rose 1998).

South Africa provides a unique opportunity to study the sustainability of shark fisheries for a number of reasons. Sharks have been exploited commercially since the beginning of the 19th century (Kroese & Sauer 1998, Sauer *et al.* 2002). In the 1970s, a pelagic longline fishery developed in the region targeting tunas (mainly Yellowfin *Thunnus albacares* and Bigeye *T. obesis*) and Swordfish *Xiphias gladius*, with a pelagic shark bycatch. A shark directed longline fishery commenced in 1992 (BCLME 2005). South African ports are also important for discharging catches by distant water fleets operating on the high seas in the south east Atlantic and south west Indian Ocean. South African waters also support a broad diversity of sharks including 36 species of threatened, near-threatened or data-deficient sharks, 19 of which are threatened by either directed fishing operations or bycatch (IUCN 2007). It is also a country with a fishery observer programme and hence a relatively large amount of data. Furthermore, it provides an interesting opportunity to explore the management of shark catches in a developing nation, because the large pelagic fishery targeting tuna and Swordfish is being expanded. As a result of this increased fishing effort, a larger number of sharks are likely to be caught. Given the vulnerability of shark populations to increased mortality, the managing authority is considering various options including the closure of the directed shark fishery. This chapter provides the first thorough assessment of pelagic shark catches in South African waters.

METHODS

Shark catches were recorded by independent fishery observers on board pelagic longline vessels operating in the South African fishery from 1998 to 2005. In addition to shark bycatch data (species, number, status and size, no weights are obtained at sea), as well as gear type (e.g. number of hooks, length of mainline, etc.) and operational information (time of set, position, etc.) was recorded. Vessels were licensed to fish within South Africa's EEZ as well as on the high seas. Sharks were measured at sea by recording the precaudal length (PCL) (tip of the snout to the precaudal pit) to the nearest centimetre. Total length data were collected from 2002 to 2006 and precaudal length from 2005 till June 2007.

Identification of species was made by observers at-sea. Based on the evidence of poor species identification of seabirds (Chapter 1) similar mis-identifications are likely to occur especially of rarer species. There is some confidence in the identification of at least the two most commonly caught species, Blue *Prionace glauca* and Short-finned Mako *Isurus*

oxyrinchus Sharks. Levels of bycatch were reported as catch rates (numbers of sharks or kg caught per 1000 hooks). Arcview GIS 3.2 was used to perform spatial analysis. To explain the observed variance in relation to explanatory variables generalised linear models were fitted to the data using Genstat 9 (McCullagh & Nelder 1989, Genstat Committee 2007). The Akaike's Information Criterion (AIC) was used for model selection (Quinn & Keough 2002). The logarithm of the parameter λ of a Poisson distribution was modelled as a linear combination of explanatory variables: e.g. for three explanatory variables, $\log \lambda = a + b_1x_1 + b_2x_2 + b_3x_3$. Explanatory variables investigated included calendar year, season, month, area, vessel name, moon phase (8 phases), bathymetry, time of set and bait type (fish or squid bait). Both four seasons i.e. summer=December–February, autumn=March–May, winter=June–August, spring=September–November, and two seasons i.e. summer=November–April and winter=May–October, were investigated. The following areas were recognised: Atlantic Ocean (west of 20 °E) versus Indian Ocean (east of 20 °E); and west coast (north of Cape Columbine), southern-west coast (Cape Columbine to Cape Point), Agulhas Bank, south coast and east Coast. Bathymetry was investigated as a continuous variable (to the nearest metre) and in categories i.e. 0–500 m, 500–1 000 m, 1 000–1 500 m, >1 500 m. Time of set was related to the time of nautical dawn and dusk. Sets commencing and finishing during daylight were classified as 'light'; commencing and finishing during the night were classified as 'dark' and sets either straddling nautical dawn or dusk were classified as twilight.

The three techniques used to calculate total catch were 1) basic extrapolation of total fishing effort multiplied by an average annual catch rate, 2) building on the basic extrapolation to incorporate the effect of vessel flag and 3) adding the effect of season and area. This was done based on the method used by Lewison and Crowder (2003):

$$\hat{C}b = \sum rsf\,(Cbrsf / Eorsf) * Edrsf$$

where $\hat{C}b$ = *Estimated total bycatch of a species, b*
       Cbrsf = *Observed bycatch of a species, b within region, r, season, s and flag, f*
       Edrsf = *Number of hooks deployed, d, within region, r, season, s and flag, f*
       Eorsf = *Number of hooks observed, o, within region, r, season, s and flag, f*
       b = *Bycatch species or group of species*
       r = *Region (1° grid cell)*
       s = *Season i.e. summer = November–April, winter = May–October*
       f = *Flag i.e. Asian or South African.*

RESULTS

The South African pelagic longline fishery comprises of two sectors, namely a swordfish, South African flagged sector and a tuna, Asian flagged sector. These two sectors use different gear configurations and practises (refer to General Introduction, Chapter 1 and 9). The swordfish sector set a total of 10 million hooks from 1998 to 2005, 10% of which were observed. Tuna directed Asian vessels set a total of 41.5 million hooks from 1998 to 2005, 8% of which were observed.

Species composition and bycatch rates

Twenty species were caught by the tuna and Swordfish fishery, although two species dominated catches: Blue (69.2%) and Short-finned Mako (17.2%) Sharks (Table 1). Although Blue Sharks were the most frequently caught (Table 1), they were frequently discarded (22.9%), often after being finned. Observers reported 30% and 25% of Blue Shark catches were finned in 2000 and 2001, respectively. 65.7% of Blue Sharks caught were alive on hauling. Short-finned Mako Sharks on the other hand were most commonly retained (86% retained, 10% released alive and 4% discarded dead). 46.9% of Short-finned Mako Sharks were alive on hauling. Thresher Sharks *Alopias vulpinus* were infrequently caught and equally discarded/released and retained (Table 1). Bronze Whalers *Carcharhinus brachyurus* and three species of hammerhead sharks, Scalloped, Smooth *Sphyrna zygaena* and Great *S. mokarran* Hammerhead were infrequently caught, but when they were caught they were usually retained (Table 1). Crocodile *Pseudocarcharias kamohari*, Oceanic Whitetip, Dusky (likely to include mis-identified Silky Sharks *C. falciformis*), Bigeye Thresher *Alopias superciliosus*, Porbeagle *Lamna nasus,* Zambezi shark *C. leucas* and Cookie Cutter *Isistius brasiliensis* Sharks were infrequently caught, but almost always discarded/released (Table 1). Although Crocodile Sharks were caught infrequently, they were some times caught in large numbers (e.g. 81 on one set off the east coast in February).

GLM analysis revealed that vessel name accounted for the largest proportion of the explained variance (Table 2). The term vessel name includes properties of skipper behaviour (such as preference for bait type, area, time of settings etc) and fishing operations (gear configuration, deck lighting, line setter, bait caster etc). It is likely that a number of factors affecting mortality are confounded in this term. Shark catch rates averaged 6.01/1000 hooks (range 1.07 and 69.94/1000 hooks) per vessel. Fifteen (30%) vessels caught sharks above the average catch rate, catching 46% of sharks from 10% of hooks. One vessel caught 18% of sharks from 2% of hooks. Thirteen of the 15 vessels were South African flagged, targeting Swordfish.

Vessel flag was a significant predictor of shark bycatch in all models, but a high degree of co-linearity occurred because vessels were either flagged to South Africa or Asia for the duration of the study (Table 2). Because vessel name explained more variance, vessel flag

was dropped from the final model. South African flagged vessels caught 61% of the sharks from only 25% of the effort; significantly more than Asian flagged vessels (t=103.84, p<0.001). Although South African vessels target Swordfish, the majority of the catch is made up of other species, whereas Asian flagged vessels mainly catch their target species (Fig. 1). Regulations for the swordfish directed fishery stipulate that retained shark catches may not exceed 10% of the Swordfish and tuna catches combined. For this fishery however, the reported shark catches varied between 4–162% (average 80%) of the combined directed catch. If the vessel with the highest shark catches is removed, the average decreases to 30%. If the three vessels which consistently caught substantially over the average are excluded the remaining vessels' shark catches comprised 4–35% (average 21.5%) of their catch and therefore can be considered the average shark bycatch expected when targeting Swordfish. For the tuna directed fishery the regulation stipulates that sharks may only comprise 10% of tuna catches. The tuna directed vessels caught fewer sharks (ranging 9–16%, average 15%) (Fig. 1).

Table 1: Shark catch composition and catch rates per fleet in the South African tuna and swordfish directed pelagic longline fishery as reported by fisheries observers.

| Species	IUCN category	Composition	Catch rate Asian vessels	Catch rate SA vessels
Blue Shark *Prionace glauca*	Near Threatened	69.2%	1.78	11.66
Short-finned Mako Shark *Isurus oxyrinchus*	Vulnerable	17.2%	0.87	1.41
Crocodile Shark *Pseudocarcharias kamohari*	Near Threatened	4.2%	0.12	0.64
Bronze Whaler Shark *Carcharhinus brachyurus*	Near Threatened	2.6%	0.03	0.95
Thresher Shark *Alopias vulpinus*	Vulnerable	2.2%	0.12	0.18
Thresher Bigeye Shark *Alopias superciliosus*	Vulnerable	0.3%	0.01	0.03
Oceanic Whitetip Shark *Carcharhinus longimanus*	Vulnerable	1.2%	0.00	0.18
Dusky Shark *Carcharhinus obscurus*	Near Threatened	0.9%	0.01	0.10
Porbeagle Shark *Lamna nasus*	Vulnerable	0.3%	0.00	0.06
Smooth Hammerhead *Sphyrna zygaena*	Near Threatened			
Scalloped Hammerhead *Sphyrna lewini*	Endangered	0.6%	0.00	0.04
Great Hammerhead *Sphyrna mokarran*	Data deficient			
Cookie Cutter Shark *Isistius brasiliensis*	Least Concern	0.1%	0.000	0.020
Zambezi Shark *Carcharhinus leucas*	Near Threatened	0.03%	0.000	0.006
Bigeye Sixgill Shark *Hexanchus nakamurai*	Not listed	0.01%	0.001	0.000
Soupfin Shark *Galeorhinus galeus*	Vulnerable	0.01%	0.000	0.003
Lanternshark *Etmopterus spp*	Least Concern	0.01%	0.000	0.001
Hardnose Houndshark *Mustelus mosis*	Not listed	0.00%	0.000	0.000
Tiger Shark *Galeocerdo cuvier*	Near Threatened	0.00%	0.000	0.001
Unidentified sharks		1.1%		
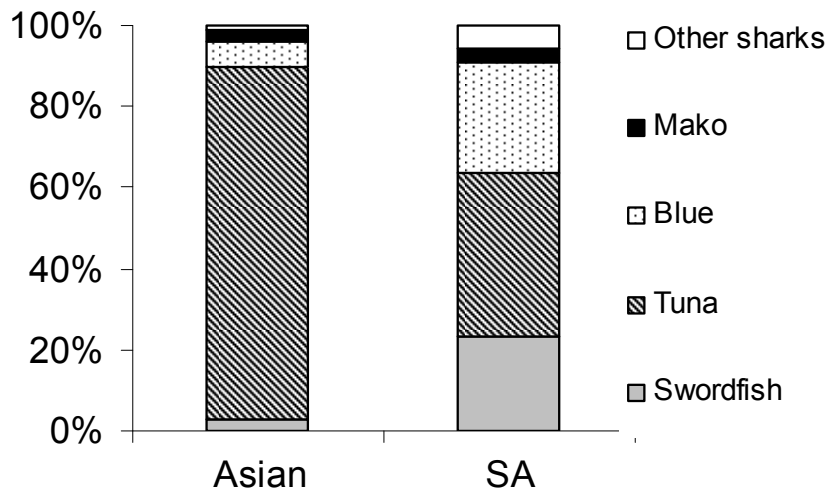
Figure 1: Catch composition of South African and Asian flagged pelagic longline vessels operating off southern Africa.

Table 2: Summary of the percentage variance explained from GLM analysis of Blue and Short-finned Mako Sharks.

Parameter	Blue Shark	Short-finned Mako Shark
Vessel Effect	56.4	9.5
Month	6.3	3.8
Year	3.6	1.9
Area	3.2	3.3
Bathymetry	0.6	2.2
Bait type	0.2	1.2
Moon phase	0.1	0
Time of set	0	0.3
Total	70.4	22.2

Seasonality

Generalised linear models showed the best fit when 'month' was included in the model in preference to season despite various monthly groupings investigated (Table 2). Significantly ($p < 0.001$) more Blue and Short-finned Mako Sharks were caught in the latter half of the year, June–December (Fig. 2).
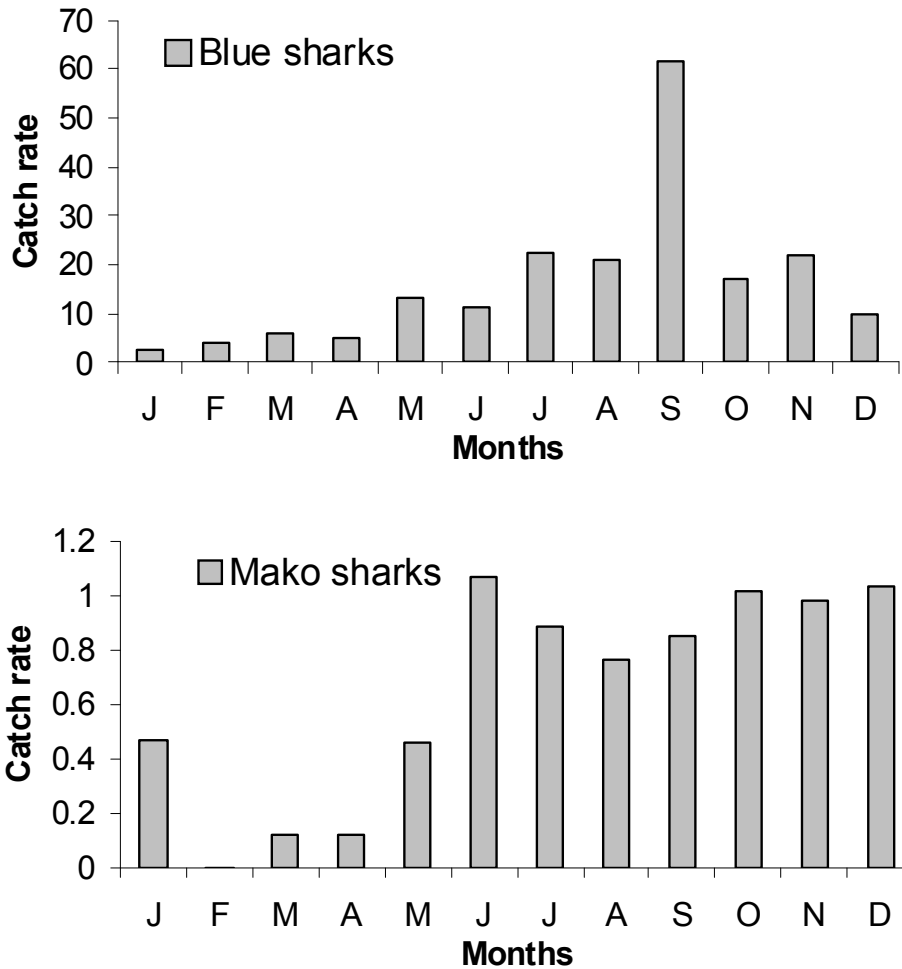
Figure 2: Standardised monthly catch rates (per 1000 hooks) for Blue and Short-finned Mako Shark bycatch in the tuna and swordfish directed fishery, 1998–2005.

Annual trend

There was a decrease in catch per unit effort (CPUE) for Blue Sharks over five years, from 21.6/1000 hooks in 2001 to 6.9/1000 hooks in 2005 (Fig.3). This explained 3.6% of the variance in catches ($p<0.001$) (Table 2). A similar trend CPUE was observed for Short-finned Mako Sharks, decreasing from 7.7/1000 hooks (2000) to 0.7/1000 hooks (2005) (Fig.3), accounting for 1.9% of model variance ($p=<0.001$) (Table 2).
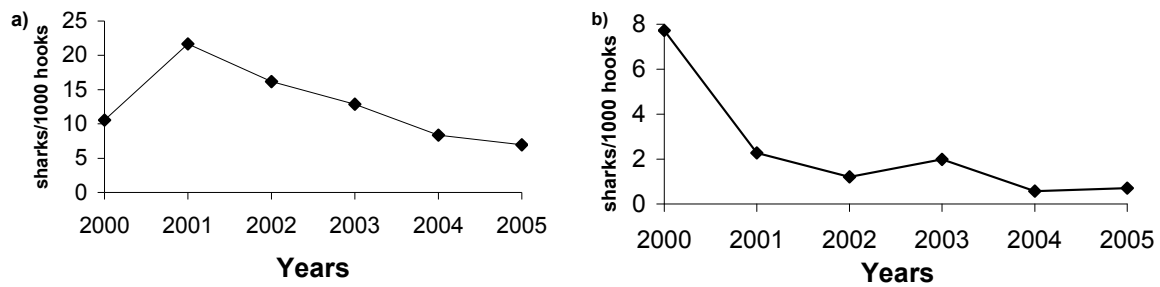
Figure 3: Standardised catch rates (sharks/1000 hooks) for a) Blue and b) Short-finned Mako Sharks, 2000–2005 as bycatch in the tuna and swordfish directed fishery (note different scales).

Length frequency

Precaudal length frequency measurements were recorded from 6 871 Blue Sharks (29.2% female, 45.7% male and 25% sex unknown) (Table 3). Males were significantly larger than females (t=1.96, p<0.001) (Table 3). Precaudal length frequency measurements were recorded from 2 722 Short-finned Mako Sharks (40.2% female, 34.8% male and 25.0% sex unknown). There was no difference in the average length of male and female Short-finned Mako Sharks (t=-1.7, p=0.1) (Table 3). There was a consistent decreasing trend in length frequency for both species over the entire period (2002–2007) (Fig. 4). Both Blue (Fig. 5) and Short-finned Mako (Fig. 6) Sharks increased in size from west to east.

Table 3: Summary of precaudal length measurements (cm) for Blue and Short-finned Mako Sharks (sexes averaged due to no significant difference observed), 2005–2007.

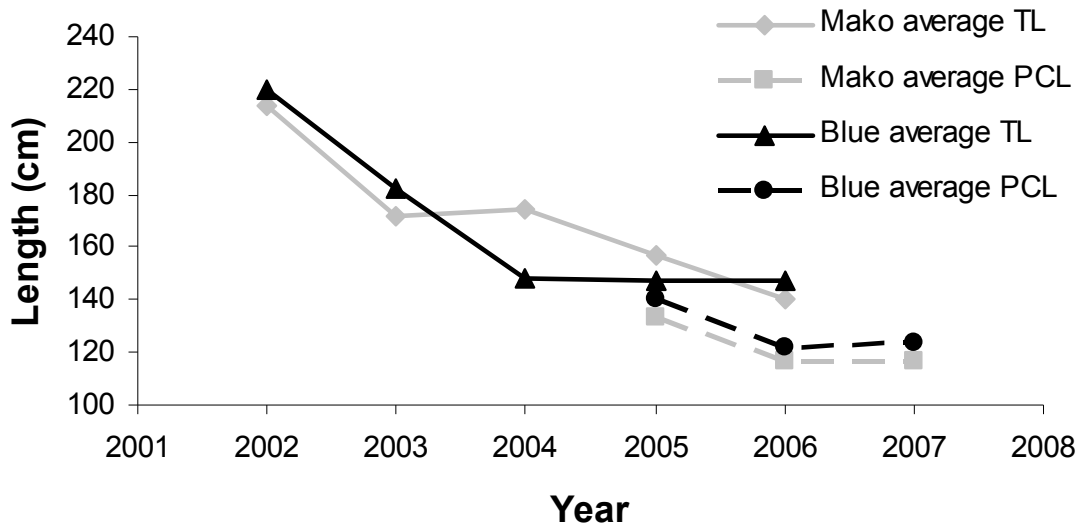| Blue Shark	n	Average	Range	SD
Male	3139	140	64-321	39
Female	2009	132	51-263	35
Short-finned Mako Shark				
Male	947	128	33-270	30
Female	1094			

Figure 4: Decreasing trend in precaudal length and total length for Blue and Short-finned Mako Sharks, 2002-2007 (observer data from tuna and swordfish directed fishery).
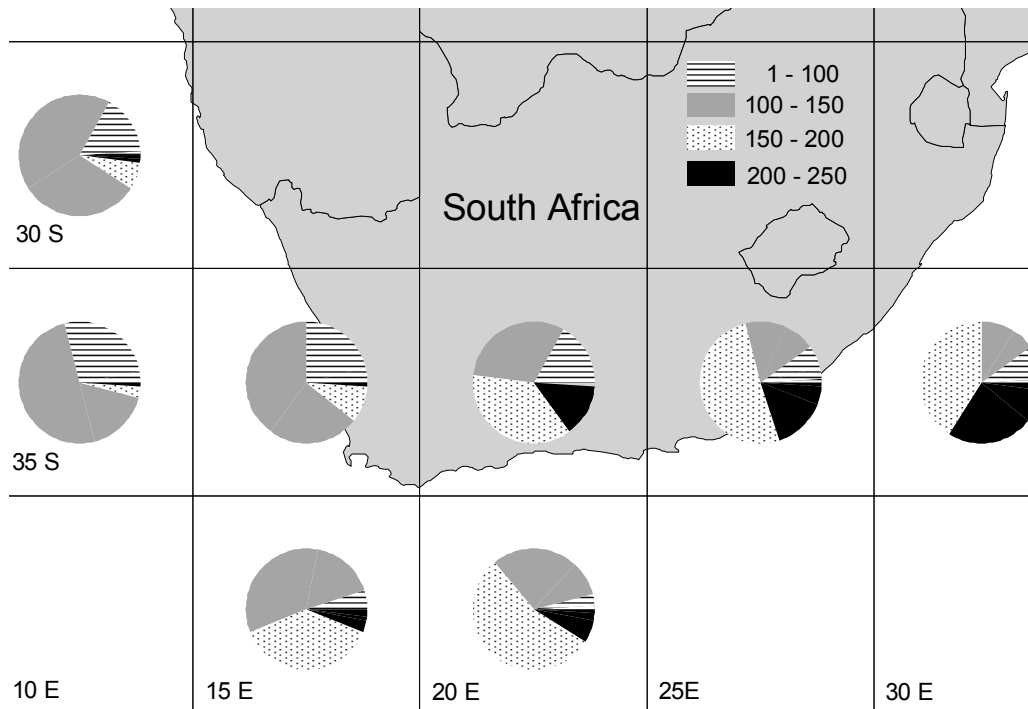

Figure 5: Proportions of Blue Sharks in four size classes (cm) summarized by 5° grid cell.
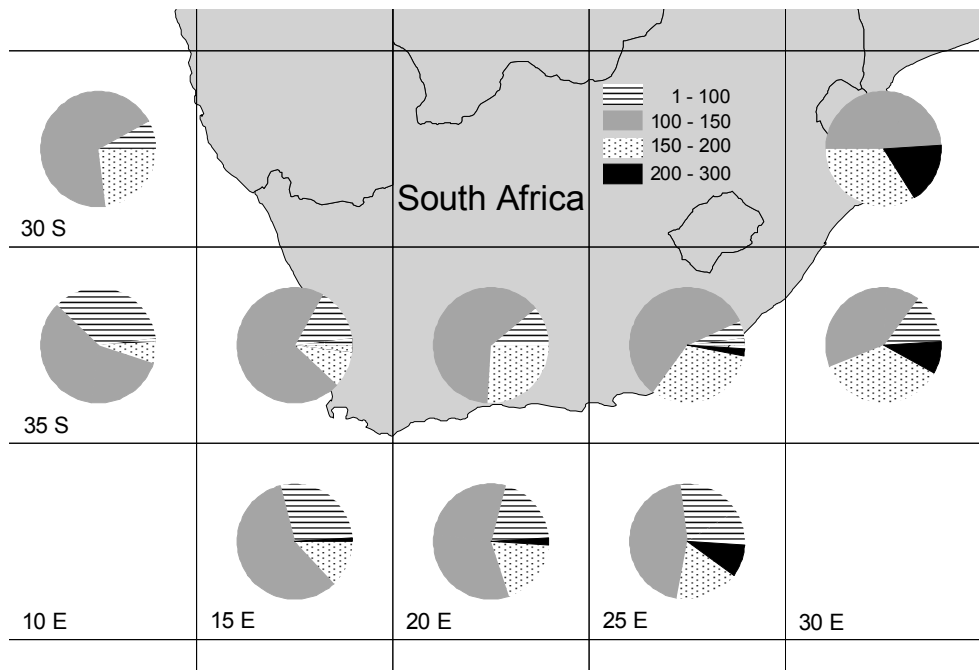
Figure 6: Proportions of Short-finned Mako Sharks in four size classes (cm) length summarized by 5° grid cell.

Distribution

Short-finned Mako Sharks (Fig. 8) were caught more widely than Blue Sharks (Fig. 8 and 9). Significantly more Blue Sharks were caught in the Atlantic Ocean (t=-24.6, p<0.001) whereas more Short-finned Mako Sharks were caught in the Indian Ocean (t=5.47, p<0.001). However, the model produced the best fit when five areas were specified (Table 2). The highest catch rate for Blue Sharks was in the southern west coast region, followed by the west coast, Agulhas Bank and south and east coasts (Fig. 8). Catches of Short-finned Mako Sharks were greatest on the west coast, followed by on the Agulhas Bank, southern west coast and south and east coasts (Fig. 9). Blue Sharks tended to be caught in water >1500 m, whereas Short-finned Mako Sharks were caught further inshore in water on average 200 m deep (Table 2, Fig. 10).

Figure 8: Distribution of Blue Shark catch rates by 1° grid cell, 1998–2005.

Figure 9: Distribution of Short-finned Mako Shark catch rates by 1° grid cell, 1998–2005.

Figure 10: Blue and Short-finned Mako Shark catch rates by bathymetric depth categories.
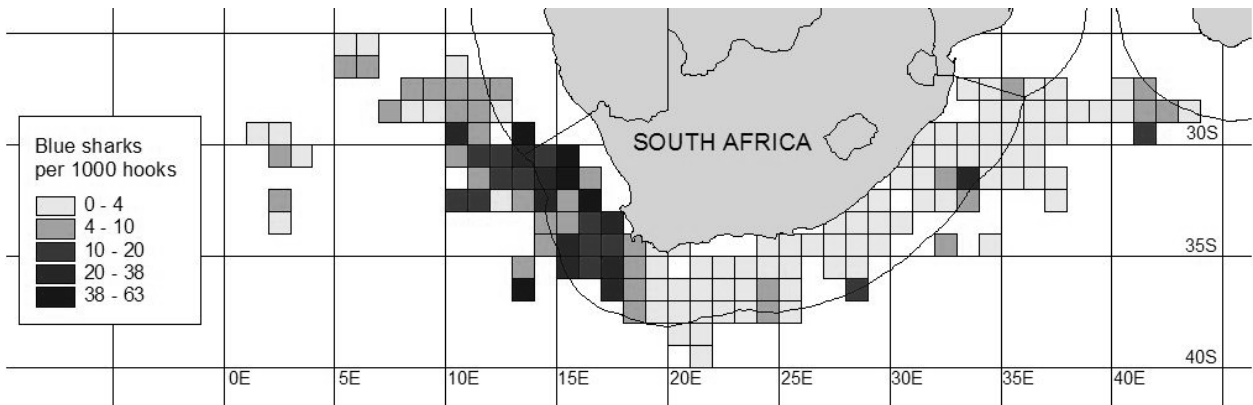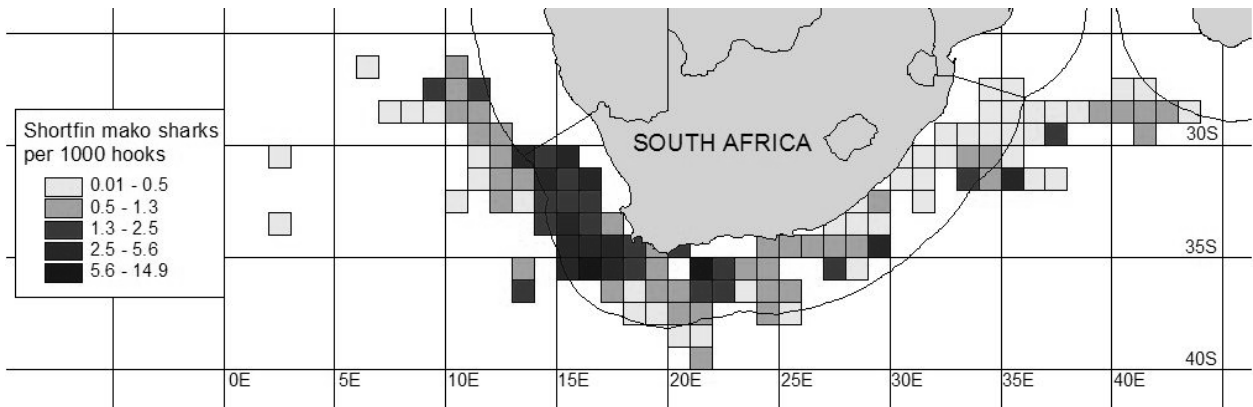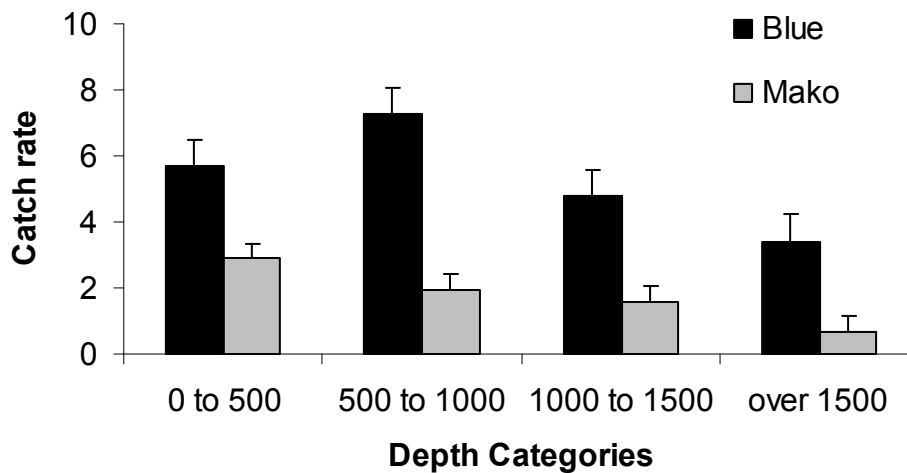
Operational differences

Blue and Short-finned Mako Sharks were more likely to be caught when using squid bait as apposed to fish bait, although high catch rates of Blue sharks also took place on a combination of squid and fish bait (Table 2). Blue Sharks were more likely to be caught during the day sets and outside of full moon periods, whereas Short-finned Mako Sharks were more likely to be caught on night sets and moon phase had no effect on catch rates.

Estimating total mortality

Simply extrapolating the average catch rate (5.9/1000 hooks) to the total fishing effort predicted that approximately 73 500 sharks were killed per year (range 15 000–165 000) (Table 4). When the extrapolation took vessel flag into account the annual average estimate decreased to sharks 39 000 per year (range 2 500–70 000) (Table 4). Extrapolation based on stratification by 5° grid cell and by flag suggested that 43 000 sharks were caught per year (range 20 000–77 000) (Table 4). It also revealed a decreasing trend from approximately 73 000 sharks in 1998 to 20 000 in 2005. Catch rates varied between 1.6 and 30.9/1000 hooks per 5° grid cell (Fig. 11).

Table 4: Summary of total shark mortality in the South African large pelagic longline fishery using a basic extrapolation of a single catch rate applied to the total effort, an extrapolation by flag and an extrapolation stratified by 5° grid cell, season and flag, 1998–2005.

|	Basic	Extrapolation by flag			Extraplation by flag and area		
	Total	Asian	SA	Total	Asian	SA	Total
1998	152936	0	2551	2551	69978	2773	72751
1999	55579	18311	4828	23139	39030	3078	42108
2000	164500	24557	20565	45122	59361	17964	77325
2001	75603	53125	16896	70021	27355	22807	50163
2002	43511	26321	28292	54613	7122	29676	36797
2003	27385	0	27385	27385	0	21144	21144
2004	52904	7521	50680	58201	6549	17674	24223
2005	15246	8716	23871	32586	9605	10713	20318
min	15246	0	2551	2551	0	2773	20318
max	164500	53125	50680	70021	69978	29676	77325
Average	**73458**	17319	21883	**39202**	27375	15729	**43104**
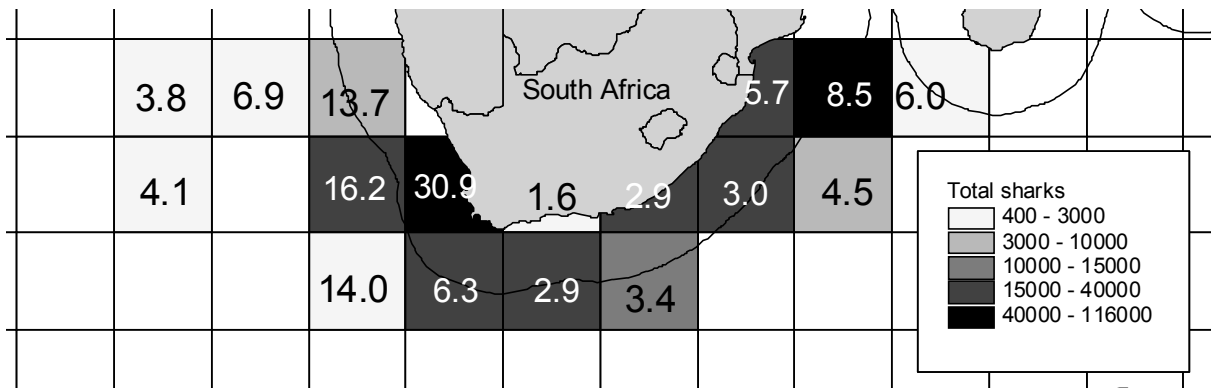
Figure 11: Average catch rates (values in grid cells) and total shark catch (shading) per 5° grid cell for the tuna and swordfish directed fishery, 1998–2005.

DISCUSSION

Magnitude of bycatch

This study estimates some 43 000 pelagic sharks were caught each year during, 1998–2005 by the South African pelagic longline fishery for tunas and Swordfish. Of the 20 species caught, six are considered Vulnerable and one Endangered (Table 1, IUCN 2007). The estimated global annual shark bycatch at the end of the 1980s was approximately 260 000 to 300 000 t, or 11.6–12.7 million sharks, predominantly comprising Blue Sharks (Bonfil 1994). Clarke *et al.* (2006) estimated that 10.7 million Blue Sharks are traded each year, with a total for all species of 26–73 million sharks are traded annually worldwide. South African shark catches comprise less than 1% of this total. The proposed increase in fishing effort to 50 rights holders in the South African large pelagic fishery for tunas and Swordfish, could increase the fishing effort threefold from 6.4 to 20 million hooks per year. If the distribution of fishing effort remains constant shark bycatch in this fishery could increase to approximately 134 000 or 3 000 t of sharks per year.

Factors affecting bycatch

The greater shark catch rates recorded by the swordfish directed fishery compared to the tuna directed fishery are likely to be due to the degree of niche overlap between Blue and Short-finned Mako Sharks, and Swordfish which tend to make vertical excursions between the surface and depths of several hundred metres during the day and are confined to depths near the thermocline at night (Bigelow *et al.* 1999). Gamblin *et al.* (2007) found lower catches with deeper set gear, consistent with the tuna directed fishery (Chapter 9) catching fewer sharks. The higher catch rate of Blue Sharks during day sets is consistent with Gamblin *et al.* (2007), who found the same pattern in the tropical Indian Ocean. The distribution of Short-finned Mako and Blue Shark catches were consistent with their preferred habitat of

epipelagic and littoral waters from the surface to depths of 500 m in the case of Short-finned Mako Sharks, and oceanic and epipelagic in the case of Blue Sharks (Holts & Bedford 1993, Sepulveda *et al.* 2004).

Sustainability of reported catches

The decreases in CPUE and shark size for Blue and Short-finned Mako Sharks is cause for concern. A factor which may contribute to the observed decline is the introduction of a 10% bycatch restriction in 2004 (MCM 2007), although the extent to which this was adhered to was low as witnessed by the average proportion of shark catches in the swordfish fishery (4–162%, average 80%). Similar decreasing CPUE and length frequencies of Blue Sharks have been reported in the north Atlantic (decreases of 5–6% per year from 1995 to 2003), the north-western Atlantic (decreases of 80% from 1985 to 1994) and the south-eastern Atlantic Oceans (decreases from 8–2/1000 hooks from 1970 to 1990), and in the vicinity of the Reunion Islands, Indian Ocean (Nakana 2000, Simpfendorfer *et al.* 2002, Campana *et al.* 2005, Poisson 2007).

Despite the potential overexploitation of sharks (Barker & Schluessel 2005), only a few studies have produced estimates of Maximum Sustainable Yield (MSY) or other reference points for sharks by species. Stock Assessments have been conducted for Blue Sharks in the North Pacific (Kleiber *et al.* 2001) and Blue and Mako sharks in the North and South Atlantic (ICCAT 2004b). Based on the work of Kleiber *et al.* (2001), Clarke *et al.* (2006) extrapolated a global MSY of 7.26–12.6 million Blue Sharks per year and an estimated 10.7 million Blue Sharks are traded annually worldwide. Thus Blue Sharks globally are being harvested at levels close to or possibly exceeding MSY. Using the Atlantic estimated MSY (ICCAT 2004b) and extrapolating globally, Clarke *et al.* (2006) estimated a global MSY of 0.73-1.09 million tonnes per year which exceeds the trade based figure suggesting a less problematic scenario. The International Convention for the Conservation of Atlantic Tunas (ICCAT) assessment of Blue and Short-finned Mako Sharks, which included a review of their biology, found no evidence of stock decline, but the assessment was conducted with severe limitations on the quantity and quality of the information and therefore the assessment was considered very preliminary in nature (ICCAT 2004b).

Although sharks in general display conservative life histories, some species of shark are more or less vulnerable than others. Of the two most commonly caught species off southern Africa, Blue Sharks are considered less vulnerable to fishing mortality because they are one of the most abundant, widespread, fecund, and fast growing shark species worldwide (Smith *et al.* 1998, Campana *et al.* 2004, Aires-da-silva & Gallucci 2007). Blue Sharks nevertheless showed evidence of fishing pressure in southern Africa. The concept of high productivity could be misleading because an analysis conducted by Aires-da-silva and Gallucci (2007) shows a strong dependence of Blue Shark population growth rates on the survival of immature sharks (0–4 years). Length frequency measurements indicate that most animals caught in the South African pelagic longline fishery are immature (Mollet *et al.* 2000, Skomal

& Natanson 2003). Thus the high numbers of immature Blue Sharks caught in the South African pelagic longline fishery could thus be having a disproportionally large effect on the population as reflected by the decreasing CPUE. The decreasing CPUE observed for Mako Sharks is less surprising because they have a more conservative life-history strategy (Campana *et al.* 2004). They have a longer gestation period (15–18 months compared to 9–12 months for Blue Sharks), smaller litter size (4–25 pups per litter compared to 4–135 Blue Shark pups per litter) and mature at a later age (7–8 years compared to five years for Blue Sharks) compared to Blue Sharks (Smith *et al.* 1998, Mollet *et al.* 2000, Skomal & Natanson 2003).

The decreasing length frequency is further evidence of an over-fished population. The lack of large sharks landed in this fishery could partly be the result of larger sharks not being retained on nylon branchlines (the use of wire traces is prohibited in this fishery) (MCM 2007, Ward *et al*. 2007), but it does not account for the decreasing trend observed in length frequency, because nylon branch lines were used throughout the study period. The release of larger sharks is important because larger (and hence older) breeders produce more young (Skomal & Natasan 2003). This concept is well known for many teleost species where older, larger fish, produce more, healthier and larger eggs (Chambers & Legget 1996, Trippel 1998, Berckley *et al*. 2004a,b). In conclusion, this paper provides the first evidence that questions the sustainability of pelagic shark catches in South African waters.

Broader consequences of removing sharks

Evidence from elsewhere suggests that decreases in pelagic shark populations can have ramifications for the functioning of pelagic ecosystems (Beddington 1984, Baum & Myers 2004). Large top predators structure aquatic ecosystems and may be essential for the maintenance and stability of food webs (Stevens *et al.* 2000). However, because most marine ecosystems contain complex food webs, predator-prey interactions maybe less tightly coupled compared to terrestrial ecosystems (Jennings & Kaiser 1998). This is important because it has been suggested that the removal of predators would allow fishers to catch more of their prey (Stevens *et al*. 2000). Simulated investigations of shark depletion on the Venezuelan shelf and in the Alaskan Gyre show that marked and unforeseen changes in the abundance of many species are likely to occur. Many species which increased in the model were not the direct prey of the sharks, suggesting that shark depletion propagates through the food web in a complex manner (Mendoza 1993, Pauly *et al.* 1996). The Hawaiian reef ecosystem and the Central North Pacific on the other hand produced a relatively small effect on the majority of the food webs in response to the removal of sharks (Polovina 1984).

Implications for management

At present no mitigation measure aimed at reducing shark catches has been widely implemented. Management has rather focused on limiting the retention of sharks through the implementation of fin to trunk ratios or catch limitations. Enforcing trunks to be landed

with fins acts as a disincentive to retain sharks because it fills freezer space which could rather be filled with more valuable species such as tunas and Swordfish (Weber & Fordham 1997). Bycatch limitations, such as the 10% shark bycatch limitation place by South African authorities on the tuna and swordfish fishery in 2004 (MCM 2007) may not be appropriate for swordfish fisheries because of the niche overlap between Swordfish *Xiphias gladius* and pelagic sharks. This study showed poor compliance to this regulation and shark catches typically comprised a large proportion of the catch. Fortunately most Blue (66%) and many Short-finned Mako (47%) are alive when caught and can be released by cutting the line as close as possible to the animal, obviating the need to discard dead sharks. The regulation is more appropriate for the tuna fisheries, which in southern Africa caught fewer sharks.

Reducing shark capture through the use of time and area closures is another option available to fisheries managers. Locally, the existence of smaller Blue and Short-finned Mako Sharks on the west coast of South Africa may indicate the presence of nursery grounds and warrants further investigation. In conclusion, the preliminary evidence of over-exploitation presented in this study and the known vulnerability of shark species, the adoption of further management measures to limit shark catches in southern Africa is encouraged.

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Chapter 4

SEABIRD BYCATCH IN THE DEMERSAL
LONGLINE FISHERY OFF
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SEABIRD BYCATCH IN THE DEMERSAL LONGLINE FISHERY OFF SOUTHERN AFRICA

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ABSTRACT

This study evaluates seabird bycatch in the demersal longline fishery for Cape hakes (*Merluccius capensis* and *M. paradoxus*) throughout the Benguela Upwelling system. A total of 14 million hooks or 2 412 sets were observed in the South African fishery accounting for 6.8% of total effort from 2000 to 2006. Only 41 seabirds were killed (0.003/1000 hooks) of 107 seabirds caught (0.008/1000 hooks). Generalised linear modelling revealed a significant decrease in catch rate from 0.033/1000 hooks in 2000 to 0.001/1000 hooks in 2006. The White-chinned Petrel *Procellaria aequinoctialis* was the most commonly caught species (36%) at a rate of 0.003/1000 hooks. Albatrosses comprised 5% of the total catch and were caught at a rate of 0.0004/1000 hooks. Only yellow-nosed albatrosses *Thalassarche chlororynchos/carteri* were identified. Shearwaters were caught at a rate of 0.001/1000 hooks and comprised 17% of the catch. Cape Gannets *Morus capensis* were caught at a rate of 0.001/1000 hooks and comprised 17% of the catch. An estimated total of 225 (range 220–245) birds are killed per year by this fishery. 'Vessel', area and light conditions were all significant predictors of seabird bycatch. The effect of tori lines and discarding practises was also investigated. Counts of seabirds associated with fishing vessels revealed White-chinned Petrels to be the most common species, followed by Great Shearwaters *Puffinus gravis* and *Pintado* Petrels *Daption capense*. Limited seabird bycatch data exist for Namibian waters. Interviews were conducted with observers (n=13) and the fishing industry (n=6) who estimated seabird bycatch to be 0.05/1000 hooks and 0.13/1000 hooks respectively, close to the observed rate of 0.14 White-chinned Petrels killed (456 000 observed hooks). One Cape Gannet (0.002/1000 hooks) and two Atlantic Yellow-nosed Albatrosses *Thalassarche chlororynchos* (0.004/1000 hooks) were also caught, but were released alive. Differences in catch rates between trips were investigated and moon phase, area and gear type were all significant. All birds were caught when a lighter gear type was used which sank significantly slower (confirmed by data collected using time depth recorders). The study concluded that the South African component of this fishery has a relatively small impact on pelagic seabirds, whereas seabird bycatch off Namibia may have a greater affect.

INTRODUCTION

The Benguela Upwelling System is one of the most productive and dynamic ecosystems in the world (Shannon & Field 1985). Its rich resources are exploited by a range of pelagic seabirds in their years prior to sexual maturity and during their non-breeding periods (Abrams 1983, 1985, Ryan & Moloney 1988, Nel & Taylor 2002, BirdLife International 2007). The Benguela Upwelling System also supports many commercial fisheries including longline and trawl fisheries which are known to impact seabirds and have been identified as their largest threat globally (Brothers 1991, Bergin 1997, Croxall & Gales 1998, Nel *et al.* 2002, BirdLife International 2007).

The five migrant pelagic seabird species occurring in the Benguela Upwelling System that are the most susceptible to the impacts of fishing operations are the Black-browed Albatross *Thallasarche melanophrys*, shy-type albatrosses *T. cauta/steadi*, Atlantic *T. chlororhynchus* and Indian *T. carteri* yellow-nosed albatrosses and the White-chinned Petrel *Procellaria aequinoctialis* (Ryan & Rose 1995, Barnes *et al.* 1997, Ryan & Boix-Hinzen 1998, Ryan et al. 2002). Of the Benguela endemics, fishery discards make up an important component of the Cape Gannet's *Morus capensis* diet (especially in winter) which makes this species vulnerable to incidental mortality by fisheries (Berruti *et al.* 1993). Abundance of all these species is the highest along the continental shelf on South Africa's south and western coasts and decreases in a northerly direction (Crawford *et al.* 1991, 2007). Although in general their abundance decreases further north in Namibian waters, many Southern Ocean species (Shy, Black-browed, and Atlantic Yellow-nosed albatrosses, Pintado Petrels *Daption capense*, White-chinned Petrels, Sooty Shearwaters *Puffinus griseus,* and Manx Shearwaters *P. puffinus*) as well as endemics to the Benguela Upwelling System (Cape Gannet, Cape Cormorant *Phalacrocorax capensis*, Kelp gull *Larus dominicanus*) still occur in substantial numbers (Roux *et al.* 2005).

This study evaluated seabird bycatch in the demersal longline fishery for Cape hakes (*Merluccius capensis* and *M. paradoxus*). These species overlap in their geographical distribution and are associated with the Benguela Upwelling System off the west coast of southern Africa. *M capensis* mainly occurs to a maximum water depth of approximately 400 m and can be found between 0 and 35 °S. This species is most abundant on the coast of Namibia and the south coast of South Africa. *M. paradoxus* overlaps with *M. capensis* between 150 and 400 m (Botha 1980, 1985, Boyer & Hampton 2001) but extends its range to almost 900 m. It mainly occurs south of 22 °S making it more abundant along the west coast of South Africa (Payne & Punt 1995). There is evidence that at least the *M. capensis* stock is shared between Namibia and South Africa (von der Heyden *et al.* 2007). Furthermore, there is evidence of birds moving between South African and Namibian fishing grounds (Chapter 7). Thus it is relevant to evaluate the impact of this fishery on seabirds in the Benguela as a whole. The concept of managing fisheries across boundaries delineated according to physical and biological characteristics as opposed to national boundaries is one that is

gaining momentum globally (Duda & Sherman 2002). Regionally, the Benguela Current Commission was developed and ratified in 2006. Thus the large marine ecosystem approach of this study will assist this commission to understand the impact this fishery has on pelagic seabirds, and provide insights into mitigation and management practices that sustain ecosystem function and health. Also the Agreement for the Conservation of Albatrosses and Petrels (ACAP) is based on the premise of trans-boundary movements of pelagic seabirds and the need for collaborative and continuous management actions throughout their range (ACAP 2006).

METHODS

At-sea data collection

Data were collected by fisheries' observers in South Africa from 2000 to 2006. This included seabird bycatch information (species, number and status), as well as gear (e.g. number of hooks, length of mainline etc.) and operational (time of set, position etc.) information. Most birds were not returned to port and thus identification could not be confirmed in most instances, although four Great Shearwaters and one White-chinned Petrel were returned to port since 2000. Observers received seabird identification training in 2004-2006 and were equipped with a reference and identification manual at sea in since 2005 (Petersen & Honig 2005). Experience from the large pelagic longline fishery where 1 191 birds were returned to port confirms that observers frequently misidentified birds (Chapter 1). No routine observer data are available from Namibia. Seabird bycatch data were therefore collected from four vessels operating out of Lüderitz, Namibia in November 2006.

Levels of bycatch (or catch rates) were summarised as numbers of birds caught per 1000 hooks summarised by vessel, area, year, season, moon phase and with/without tori lines. Observers did not record the specifications of the tori line, although in general it followed the regulation which requires fishers to fly a tori line of 150 m in length, 28 paired streamers which reach the water and have a device to increase the drag that the end. Total catch for each species was estimated by summing the extrapolated catch by area and season, as done by Lewison and Crowder (2003):

$$\hat{C}b = \sum rs \; (Cbrs \, / \, Eors) * Edrs$$

where $\hat{C}b$ = *Estimated total bycatch of a species, b*
        Cbrs   = *Observed bycatch of a species, b within region, r and season, s*
        Edrs   = *Number of hooks deployed, d, within region, r and season, s*
        Eors   = *Number of hooks observed, o, within region, r and season, s*
        b   = *Bycatch species or group of species*
        r   = *Region (one degree grid cell)*
        s   = *Season i.e. summer = November–April, winter = May–October*

Arcview GIS 3.3 was used to provide spatial representations of information. Counts of the number of seabirds killed by each longline set were modelled using a generalised linear model with a Poisson distribution and logarithmic link function (McCullagh & Nelder 1989). Genstat 9 (Genstat Committee 2007) was used for model fitting and Akaike's Information Criterion (AIC) was used to guide model selection (Quinn & Keough 2002). The logarithm of the parameter λ of a Poisson distribution was modelled as a linear combination of explanatory variables: e.g. for three explanatory variables, $\log \lambda = a + b_1 x_1 + b_2 x_2 + b_3 x_3$. Explanatory variables investigated included year (January–December), month, season, area, vessel name, moon phase (eight phases), bathymetry, use of a tori line (true or false), offal discarding practices (whether offal was discarded on the opposite side to hauling) and time of set. In the case of season, four seasons i.e. summer (December–February), autumn (March–May), winter (June–August), spring (September–November) and two season i.e. summer (November–April) and winter (May–October) were investigated. Furthermore, the trend observed when modelling each month separately was used to investigate further combinations (see results). The spatial distribution of seabird mortality was investigated by comparing the fit of models using various areas. Two areas i.e. Atlantic Ocean (west of 20 °E) and Indian Ocean (east of 20 °E), four areas i.e. West coast (North of Cape Columbine), Cape Point (Cape Columbine to Cape Point), Agulhas Bank and South coast (21–25 °E). Bathymetry was investigated as a continuous variable (to the nearest metre) and in categories i.e. 0–500 m, 500–1 000 m, 1 000–1 500 m. Time of set was investigated by calculating nautical dawn and dusk from almanac tables. Sets commencing and finishing during daylight (i.e. after nautical dawn) were classified as 'light', commencing and finishing during the night (i.e. after nautical dusk) were classified as 'dark' and sets either commencing during daylight and finishing during the night or vice versa were classified as twilight. The number of hooks per set was used as an offset variable.

Interviews

A questionnaire was developed to supplement data in Namibia. Observers and fishers were asked whether they had witnessed seabirds being caught on longlines. If so, they were required to estimate how many and which species were most frequently caught. They were also asked whether in their opinion the reported level of mortality was affecting seabird populations. This perception may indicate willingness to implement mitigation.

TDR deployment and calculation of line sink rates

Time-depth recorders (TDRs) manufactured by Wildlife Computers (Mk9) were deployed to collect hook depth over time or line sink rates on Namibian demersal longliners to understand the relationship between seabird mortality and gear configuration. Two gear types were investigated: the first gear type consisted of a monofilament line main line with 3.5 kg weights spaced approximately 88 m apart and set their gear at a speed of 6 knots (11.1 km.hr^{-1}). The second system consisted of polysteel main line with weights of 4.2 kg

mass, spaced approximately 110 m apart and set their gear at a speed of 8 knots (14.8 km.hr^{-1}). The Mk9 TDR measures depth, temperature, light intensity and wet/dry conditions. TDRs were attached at the point the weight or the dropper. No floats were used. South African vessels use a weighting regime of 6 kg every 100 m. Full details of deployment and description can be found in Chapters 9 and 12. In brief, line sink rates (m.s^{-1}) were calculated as depth of hook over time. Line sink rates were calculated as the time taken from immersion until the recorder reached depths of 2 m, 5 m, 10 m and 15 m. Depth (m) attained after 26 s was also calculated because this is the average time the line may be protected by a well-constructed tori line (aerial extent of approximately 100 m) (Melvin *et al.* 2004).

Seabird attendance patterns

Instantaneous counts of seabirds associated with fishing vessels were recorded hourly from sunrise to sunset. Time, position, fishing activity (no fishing, start of haul, hauling, end of hauling) and level of discards during processing (none, little, medium, heavy) were recorded at each count. Generalised linear modelling was used to investigate whether seabirds accumulated over the course of a trip or day, and the effect of fishing activity on the level of processing.

RESULTS

South Africa

Approximately 210 million hooks were set from 2000 to 2006, with annual effort ranging between 15.2 and 43.6 million hooks (average 30 million per year, SD 11.7 million hooks per year) on 22 100 sets (3 150, SD 967 per year) (Fig. 1). This fishery set an average of 9 300 (range 1 000–29 000, SD 3 660) hooks per set (Fig. 1). A total of 14 million hooks or 2 412 sets were observed, representing 6.8% of total effort. Observed effort varied between 74 and 657 sets or 450 000 to 4 million hooks per year.

Figure 1/...
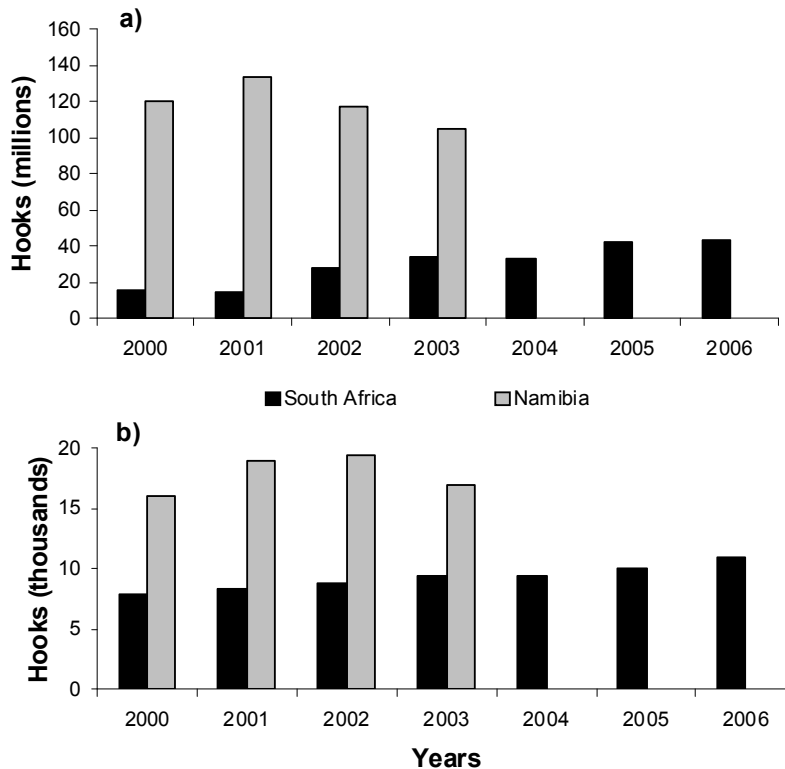
Figure 1: The total number of hooks (a) and average number of hooks per set (b) for the South African (2000–2006) and Namibian (2000–2003) hake longline fishery.

A total of 107 seabirds was caught (0.0075/1000 hooks), 38% of which were dead (0.0029/1000 hooks) (Table 1). The remaining birds were alive on capture and released, but the condition of the released birds and their subsequent survival is unknown. Generalised linear modelling revealed a significant decreasing trend in catch rate from 0.0329/1000 hooks in 2000 to 0.0012/1000 hooks in 2006 (Fig. 2), accounting for 3.8% of the variance ($p<0.001$). There was no significant seasonal trend. The White-chinned Petrel was the most commonly caught (36%) species at a rate of 0.0027/1000 hooks. A further 17% of the catch was reported as 'unidentified petrels' which are most likely to also be White-chinned Petrels, in which case their catch rate is likely to be 0.0039/1000 hooks (Table 1). Albatrosses comprised 5% of the total catch and were caught at a rate of 0.0004/1000 hooks. Only yellow-nosed albatrosses were identified. Shearwaters were caught at a rate of 0.0005/1000 hooks and comprised 17% of the catch. Most were identified as Great Shearwaters *Puffinus gravis.* The Cape Gannet was caught at a rate of 0.0013/1000 hooks and comprised 17% of the catch (Table 1).

Table 1: Seabird bycatch species composition, number caught, percentage killed and catch rates for the South African demersal longline fishery, 2000–2006.

| Species	% composition	Total	% dead	Catch rate
White-chinned Petrel	36%	38	58%	0.0027
unidentified petrels	17%	18	6%	0.0013
Total petrels	**52%**	**56**	**41%**	**0.0039**
yellow-nosed albatrosses	3%	3	33%	0.0002
unidentified Albatrosses	2%	2	50%	0.0001
Total albatrosses	**5%**	**5**	**40%**	**0.0004**
Great Shearwater	10%	11	27%	0.0008
unidentified shearwaters	7%	7	14%	0.0005
Total shearwaters	**17%**	**18**	**21%**	**0.0013**
Cape Gannet	17%	18	22%	0.0013
unidentified birds	9%	10	80%	0.0007
Total other	**26%**	**28**	**51%**	**0.0020**
Grand Total	**100%**	**107**	**38%**	**0.0075**
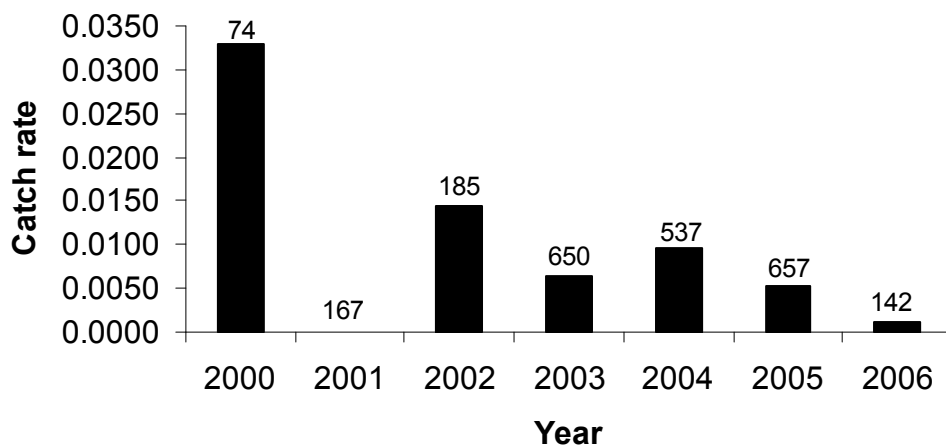
Figure 2: The annual trend in seabird catch rates in the South African hake longline fishery, 2000 to 2006.
The number of observed sets is indicated for each year.
No seabird mortality was reported in 2001 despite 167 sets observed.
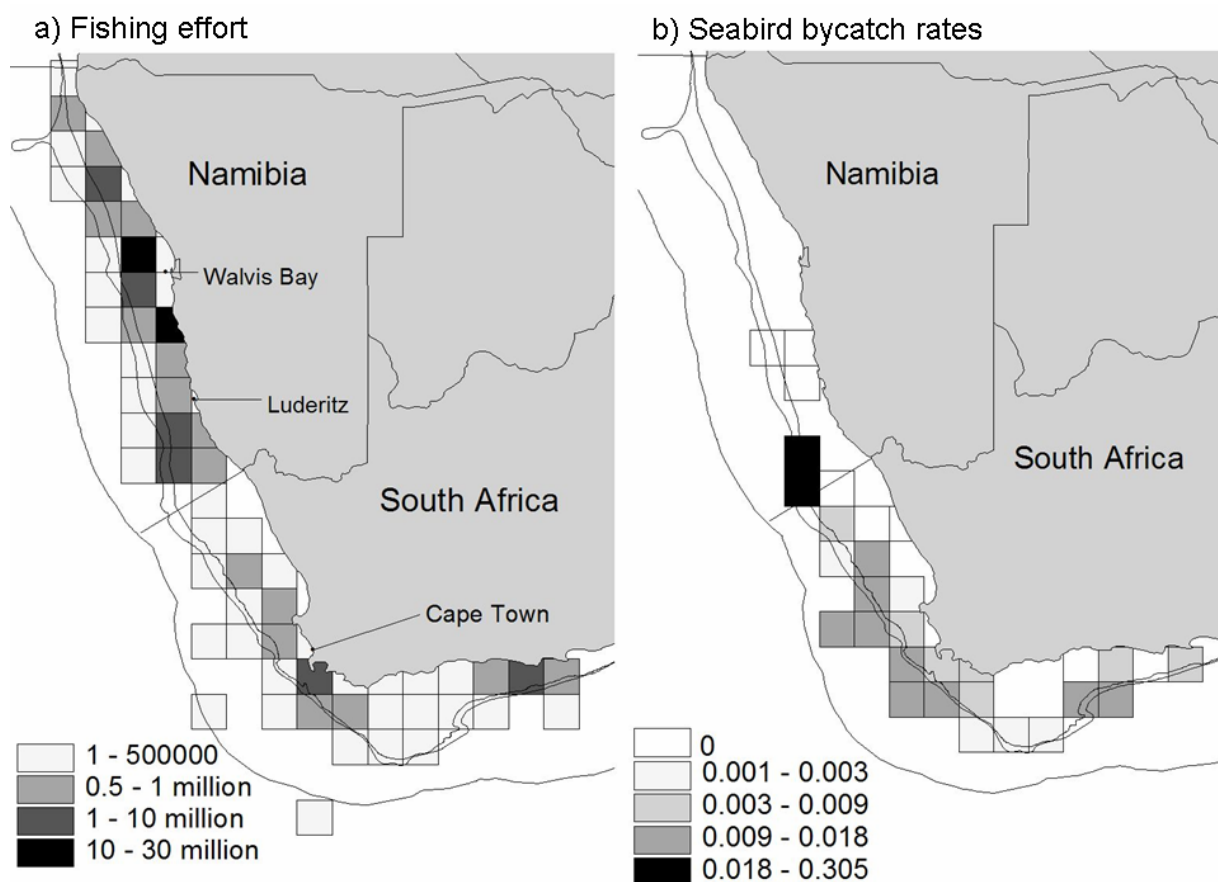
Figure 3: Average annual fishing effort (a) and average seabird bycatch catch rates (b) summarized by one degree grid cell for the hake longline fishery in the Benguela Upwelling System.

'Vessel' was the single most important variable, accounting for 24.9% of the variance in catch rate (p<0.001). 'Area' accounted for 1.8% of the variance (p<0.001). The largest catch rates were recorded on the continental shelf (500–1 000 m), particularly off the west coast of South Africa (Fig. 3). Day versus night set and moon phase accounted for 0.7%, although neither was significant (p=0.7 and 0.2). Setting mostly (94.5%) occurred at night (2.0% during the day and 3.5% at twilight). Birds were caught at a rate of 0.006/1000 hooks during the dark phases of the moon (waxing crescent to waning crescent) compared to 0.009/1000 hooks during the light phases of the moon (waxing gibbous to waning gibbous). A tori line was used on 217 of 2 198 (9%) sets. Seven birds were caught when a tori line was flown at a rate of 0.006/1000 hooks compared to 100 birds caught at a rate of 0.008/1000 hooks when a tori line was not used. The use of a tori line accounted for 0.6% of the variance (p=0.05). Offal was discarded on the same side as hauling on 90% of sets and on no occasion during setting.

Seabird attendance patterns at South African hake longliners

A total of 220 counts were recorded from 26 sets on six trips undertaken on the Agulhas Bank, from April to November 2005. Eight species were recorded foraging within 100 m of the fishing vessel (Table 2). The most common species was the White-chinned Petrel (average 28.3, SD 15.1, range 0–80) (Table 2). The Great Shearwater was the second most common species (average 12.2, SD 25.7, range 0–200), followed by the Pintado Petrel (average 9.8, SD 26.7, range 0–200). The most common species of albatross was shy-type albatrosses (average 6.0, SD 6.2, range 0–50), most of which were juveniles or immatures (Table 2). There was a significant difference in the numbers of White-chinned Petrel, juvenile Shy albatrosses and juvenile and adult Black-browed and yellow-nosed albatrosses between fishing days (Table 3). However, there was no apparent trend or accumulation in seabird numbers over any of the six trips for any species (Fig. 4 and Table 3). Fishing activity and level of processing were the best predictors of seabird abundance independent of the time of day (Fig. 4 and Table 3).

Table 2: Seabird attendance at South African commercial hake longliners (n=220 hourly counts on 26 days, April–November 2006).

| Species	Species name	Frequency of occurance	Average	Std dev	Range	% Juvenile
White-chinned Petrel	*Procellaria aequinoctialis*	1.00	28.30	15.06	0-80	-
Great Shearwater	*Puffinus gravis*	0.78	12.24	25.65	0-200	-
Cape Petrel	*Daption capense*	0.65	9.81	26.72	0-200	-
giant petrel	*Macronectes halli/ giganteus*	0.52	2.80	5.90	0-45	0.5
shy-type albatrosses	*Thalassarche cauta/ steadi*	0.95	7.90	6.90	0-53	0.76
Black-browed Albatross	*Thalassarche melanophrys*	0.81	3.70	5.00	0-30	0.48
yellow-nosed albatrosses	*Thalassarche carteri/ chlororhynchos*	0.73	1.90	2.10	0-15	0.13
Sub-Antarctic Skua	*Catharacta antarctica*	0.43	1.12	2.02	0-15	-
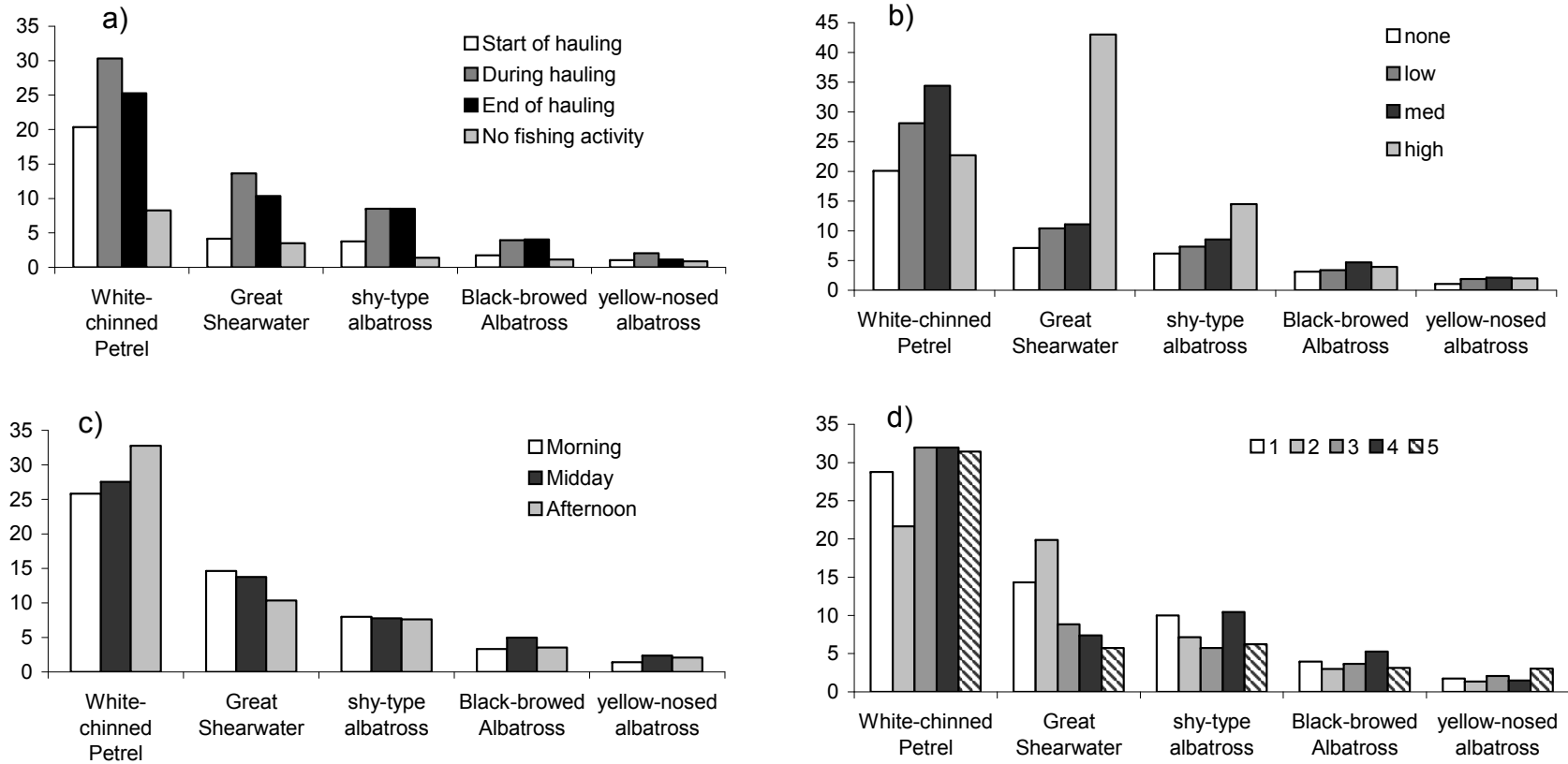
Figure 4: The effects of a) fishing activity, b) level of processing, c) time of day and d) day of trip on seabird attendance patterns at South African hake longliners (May–November). Note: seabird attendance patterns could not be recorded during setting because setting took place after dark.

Table 3: Summary of generalised linear modelling results of seabirds foraging at South African commercial hake longliners, April–November 2005. R^2 is expressed as a percentage, NSE = no significant effects, factor levels as in Fig. 4.

| Parameter | White-chinned Petrel | | Great Shearwater | | shy-type albatross | | | | Black-browed Albatross | | | | yellow-nosed albatross | | | |
| | | | | | Adult | | Juvenile | | Adult | | Juvenile | | Adult | | Juvenile | |
	NSE	R^2	NSE	R^2	NSE	R^2	NSE	R^2	NSE	R^2	NSE	R^2	NSE	R^2	NSE	R^2
Set number	1	6.7	0	3.0	0	1.1	3	5.8	1	0.2	2	2.7	1	6.1	1	0.8
Time of day	0	2.2	0	0.0	0	0.1	0	0.0	0	0.0	0	1.8	1.0	3.3	0	0.2
Fishing activity	2	9.3	0	0.0	2	2.5	2	4.0	0	0.2	0	0.1	0	2.4	0	0.0
Level of processing	1	8.9	3	8.4	0	1.2	3	7.3	0	0.0	0	0.9	0	1.0	0	0.8
% variance explained	27.1		11.0		4.9		17.0		0.4		5.5		12.8		1.8	

Namibia

This fishery set, on average, approximately 120 million hooks or 6 700 sets per year (Fig. 1). Effort has remained fairly constant over the time period (Fig. 1) and is concentrated along the continental shelf with hotspots south of Lüderitz and around Walvis Bay (Fig. 2), The average number of hooks per set was 17 800 (range 9000–37 000, SD 5 600) which increased from an average of 16 000 hooks per set in 2000 to 19 500 hooks per set in 2002 (Fig. 1). Most (80%) sets occurred around 04h00 in the morning with hauling activity peaking around midday.

Twelve of the 13 observers interviewed estimated that 0–10 albatrosses and 0–20 Cape Gannets were caught per day, with an average of 7 (range 0–120, SD 25) seabirds per trip of eight sets (144 000 hooks per trip). This suggests an approximate catch rate of 0.05/1000 hooks. Interviews with fisheries observers suggested that seabird bycatch was likely to be unacceptably high and resulting in population decreases. Six interviews with the fishing industry (skippers, shore skippers and managers) were conducted in Walvis Bay in 2004 and 2006. They estimated a bycatch of 19 (range 0–100, SD 20) birds (mainly petrels) caught each trip, representing an average catch rate of 0.13/1000 hooks. In general, skippers were not aware of the issue and therefore did not report seabird mortality in their logbooks.

Seabird bycatch data were collected from four trips from 8 to 27 November 2006. Twenty-one sets (456 000 hooks) were observed and 63 White-chinned Petrels killed (0.14 birds per 1000 hooks; Table 4, Fig. 2). One Cape Gannet (0.002/1000 hooks) and two Atlantic Yellow-nosed Albatrosses (0.004/1000 hooks) were also caught, but were released alive (Table 4).

Table 4: Seabird bycatch species composition, number caught, percentage killed and catch rates for the Namibian demersal longline fishery, November 2006.

Species	% composition	Total	% dead	Catch rate
White-chinned Petrel	95%	63	100%	0.138
Atlantic Yellow-nosed Albatross	3%	2	0%	0.004
Cape Gannet	2%	1	0%	0.002

Table 5: Summary of Namibian seabird bycatch data per trip, November 2006.

	Trip 1 and 2	Trip 3 and 4
Gear	Light	Heavy
Moonphase	Last quarter to Full	New to first quarter
Fishing grounds	28-29 °S	25-29 °S
Number of hooks observed	191 000	265 500
White-chinned Petrel mortality	63	0
Catch rate	0.33	0.00

All seabird mortalities occurred on trips with light gear undertaken when the moon was approaching full (Fig. 5 and Table 5). 'Trip' explained 79.9% of the variance (p<0.001) in bycatch rate. Including moon phase, gear type and area did not improve the fit of the model, but when modelled separately each accounted for 33.2% (p=0.01), 18.9% (p=0.03) and 8.3% (p=0.1), respectively. This is most likely due to factors which influence seabird bycatch converging. Conditions either favoured low seabird bycatch rates i.e. heavy gear, dark moon and area where few birds were found or conditions favoured high seabirds bycatch rates i.e. light gear which sinks slowly, light moon where birds can see the bait and fishing took place in an area with a high abundance of birds. The first two trips were south of Lüderitz (28 °S to 29 °S) whereas the subsequent two trips were farther north mainly between 26 °S and 27 °S (Fig. 2). All fishing took place on the continental shelf in water 200–440 m deep. Daily counts of seabirds in the vicinity of the vessel revealed larger numbers of White-chinned Petrels in the area between 28 °S and 29 °S (42, range 6–250, SD 39) compared to the more northerly region (15, range 2–64, SD 11).
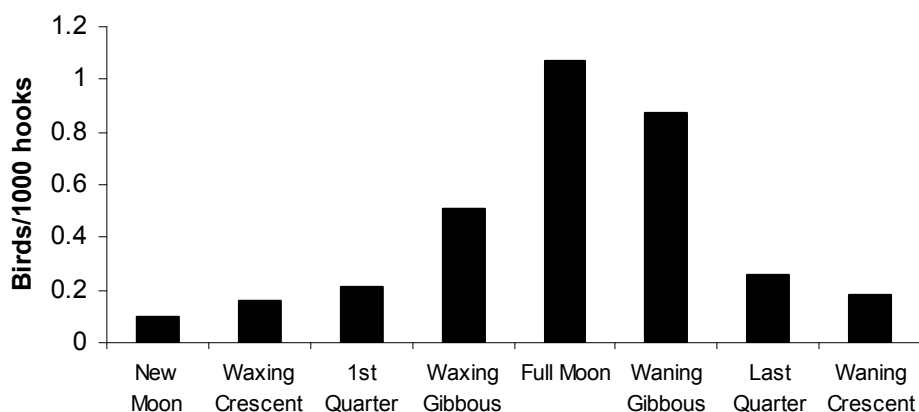
Figure 5: The influence of moon phase on seabird catch rates in the demersal longline fishery, 2000–2006.

The weighted portion of the light gear type (3.5 kg weights) sank significantly slower than the heavier (4.2 kg weights) gear type (0.08 m.s^{-1}, 0.13 m.s^{-1}, 0.16 m.s^{-1} and 0.16 m.s^{-1} compared to 0.48 m.s^{-1}, 0.51 m.s^{-1}, 0.35 m.s^{-1} and 0.38 m.s^{-1} to 2 m, 5 m, 10 m and 15 m) (p<0.001) (Fig. 6), but there was no significant difference in the sink rates of droppers (0.10 m.s^{-1}, 0.11 m.s^{-1}, 0.14 m.s^{-1} and 0.13 m.s^{-1} to 2 m, 5 m, 10 m and 15 m). Similarly there was a significant difference between the depth after 26 s for the weighted (8.0±4.0 m versus 2.3±0.3 m for light and heavy gear, respectively; p=0.003) but not for the dropper portion (3.0±3.0 m and 2.8±0.6 m respectively, p=0.08).

Figure 6: Graphical representation of line sink rates (m.s-1) to 2 m, 5 m 10 m and 15 m for light (dropper and weight) and heavy (dropper and weight) gear.

Estimated overall impacts

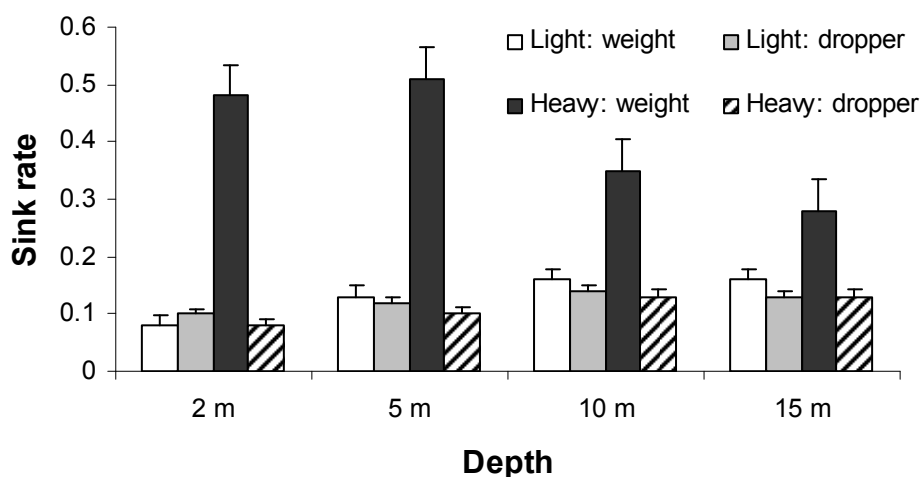In South African total bird mortality was calculated by simply extrapolating the average catch rate (0.0075/1000 hooks) to the total average annual fishing effort, and it was estimated that approximately 225 birds were killed per year (range 107–327, SD 88) (2000–2006). By extrapolating annual average catch rate by the corresponding fishing effort in that year (2000–2006), it was estimated that 245 birds were killed per year (range 0–502, SD 181, 2001 no birds were caught on observed sets). In Namibia an estimated 16 700 birds (range 14 600–18 800, SD 1 700) were killed each year, based on an average catch rate of 0.14/1000 hooks. If area is taken into consideration in extrapolation by summarising annual fishing effort (2000–2006) and catch rates by one degree grid cell (Fig. 2), a minimum of 60 and 1 650 birds were killed per year by the South African and Namibian demersal longline fleets, respectively. In the case of the South African fishery, grid cells sampled for seabird bycatch accounted for 27.4% of the distribution of South African fishing effort, whereas grid squares sampled for seabird bycatch accounted for 8.2% of the distribution of Namibian fishing effort. Therefore if these estimates were extrapolated to total fishing effort in 2006 approximately 220 and 20 200 birds were killed by South African and Namibian demersal longline fleets.

DISCUSSION

Seabird catch rates in the South African fishery were lower than those reported in demersal longline fisheries elsewhere (Ashford *et al.* 1995, Cherel *et al.* 1996, Weimerskirch *et al.* 1999, Favero *et al.* 2003, Moreno *et al.* 2006), and also lower than previous estimates for this fishery. Barnes *et al.* (1997) reported that the South African demersal longline fishery killed approximately 8 000 White-chinned Petrels (0.4/1000 hooks) per year in 1995. However, that study was based on a limited sample of 12 sets. The present study was based on 2 412 observed sets, of which 49 sets were personally observed in winter and spring of 2004–2006, and only four (two White-chinned Petrels and two Great Shearwaters) bird capture events were witnessed (0.01/1000 hooks). The reasons for the apparent reduction in numbers of seabirds killed since the Barnes *et al.* (1997) study and the decreasing trend in catch rate in this study are unclear. It is possible that some seabird mortality went unrecorded by fishery observers, but this in itself cannot explain the dramatic decrease. Training of observers to collect and identify seabirds commenced in 2004, thus a subsequent increase in reported bycatch might have been expected.

The introduction of mitigation measures post the 1995 study could at least partially explain the observed decrease. The conditions of hake longline fishing permits in South Africa include the following mitigation measures: lines should be set in the dark, a tori line should be flown during setting, and offal discarding should take place on the opposite side of the deck to hauling (Cooper & Ryan 2003). Compliance with the first of these measures is

reasonably good (94.5% of line setting during 2000–2006 occurred in the dark), but this is partly coincidence. Hake display a diurnal migratory pattern, occurring on or just above the seabed during the day and moving up through the water column to middle depths at night (Botha 1980, 1985). Demersal longliners targeting hake therefore tended to use relatively buoyant gear which they set after midnight to catch fish returning to the seabed in the early morning (Japp 1993). This explains why lines were set mainly at night during the Barnes *et al.* (1997) study despite the fact that this is not a condition of fishery permits. Night setting may explain the small numbers of albatrosses, which generally feed by day (Harrison *et al.* 1991, Hedd *et al.* 2001), caught by the fishery, it does not explain the relatively small numbers of White-chinned Petrels which do feed nocturnally (Cherel *et al.* 1996, Weimerskirch *et al.* 2000).

Despite being a condition of fishery permits, tori lines were seldom used (only 9% of sets). Many fishers are averse to using tori lines, citing that they result in gear entanglements and consequently in gear loss. Moreover, the fact that approximately 62% of seabirds caught were released alive implies that most were hooked during hauling, not setting. Fisheries provide a substantial food source for seabirds that feed opportunistically through the discarding of fisheries waste (Abrams 1983, 1985, Furness *et al.* 1988, Ryan & Moloney 1988, Garthe & Hüppop 1994, Garthe *et al.* 1996, González-Zevallos & Yorio 2006). Therefore by discarding offal on the same side of the vessel to where hauling is taking place, seabirds are attracted to the area where they are likely to become hooked (Brothers *et al.* 1999). Vessels in this fishery typically had a shelter deck making discarding on the opposite side to hauling difficult. Vessel configuration should be amended to accommodate discarding on the opposite side to hauling through chutes or other relevant technology. Furthermore, the large proportion of live birds caught highlights the need for an education programme that provides training on appropriate handling and release techniques. The poor implementation of tori lines and offal discarding regulations, suggests the discrepancy between the seabird catch rate in the Barnes *et al.* (1997) study and the present study cannot be ascribed to the implementation of mitigation measures alone and is more likely to be the result of an increased sample size.

At the time of the Barnes *et al.* (1997) study, the demersal longline fishery was at an experimental stage. The two vessels that caught most birds were large stern trawlers (30–38 m) converted for longline operations. Smaller vessels, typical of the boats operating in the fleet from 2000–2006, caught few birds. The stern trawlers were rumored to have a lot of deck lighting. It is possible that a combination of inexperience and setting from the stern trawlers caused the hooks to be exposed or more visible to foraging seabirds for longer periods of time and resulted in inflated mortality levels. Birds are also more accustomed to foraging from trawlers than from longliners (Ryan & Moloney 1988, Chapter 7).

A decreasing catch per unit effort (CPUE) is generally indicative of a decreasing population (Seber 1982), and numbers of many species of pelagic seabird affected by longlining are decreasing (Croxall & Gales 1998, Nel *et al.* 2002, BirdLife International 2007). Although

there are few reliable population estimates for White-chinned Petrels, the species most commonly caught by this fishery, there is evidence of decreases from South Georgia (Berrow *et al.* 2000a) and it is classified as Vulnerable (Barnes 2000, BirdLife International 2007, Simmons & Brown 2008). White-chinned Petrels are the most common pelagic seabird in southern African waters (Summerhayes *et al.* 1974, Ryan & Rose 1995). Off central Namibia, their relative abundance has declined by 56% over 14 years (1997–2002) (Boyer & Boyer 2005). White-chinned Petrels are also killed by other fisheries throughout their range (Cherel *et al.* 1996, CCAMLR 1997, Olmos 1997, Weimerskirch *et al.* 1999, Nel *et al.* 2002). Therefore the decreasing seabird bycatch rate may in part result from a decreasing White-chinned Petrel population.

Fisheries information on non-target species is frequently poorly collected, recorded and maintained. This is especially true in developing countries (Barker & Schluessel 2005). The state countries bordering the Benguela Upwelling System (South Africa, Namibia and Angola) are no exception. South Africa has an observer programme which has been collecting data on fishing operations, including bycatch, since 2000. Consequently the most reliable and comprehensive data for the region comes from this programme. However, even in South Africa estimates are based on relatively small sample sizes which will result in poor confidence in estimates. This study represents a marked increase in sample size from previous studies (Barnes *et al.* 1997), but it still only represents 6.8% of the fishing effort off South Africa. The situation off Namibia is much worse. There is an observer programme for this fishery, but bird bycatch is not reported. This is the first assessment of seabird bycatch for the Namibian hake longline fishery, but was based on 21 sets or 0.31% of the average annual fishing effort. It is likely that as more information is collected, the total estimated annual catch of 20 200 will be refined. Interestingly, the higher than expected catch rate observed in Namibia was supported by interviews conducted with the fishing industry.

Because seabird abundance and species composition are not uniform throughout the region (Crawford *et al*. 1991), catch rates from South Africa cannot simply be used to extrapolate for the entire region. There is a decrease in the abundance of albatrosses in a northerly direction and species composition changes from mostly shy-type and Black-browed Albatrosses dominating in the south, to an increase in the relative proportion of Atlantic Yellow-nosed Albatrosses in the northern Benguela (Crawford *et al*. 1991). Because of this decrease in seabird density in the North of the Benguela, one would expect a decrease in seabird catch rates in Namibian compared to South African waters. However, this is not observed. White-chinned Petrels, which are commonly sighted in Namibian waters, albeit at a lower density (Summerhayes *et al.* 1974, Crawford *et al.* 1991), apparently are caught in substantial numbers.

Despite seabirds occurring at lower densities in Namibian than in South Africa waters, the observed difference in the catch rate of seabirds may be attributed to differences in fishing gear and technique between the two fleets. The South African fishery typically set 7 500 hooks per set whereas the Namibian fishery set approximately 19 000 hooks per set,

increasing the effort substantially and therefore the likelihood of catching a bird per set. A larger number of hooks will also take longer to set which may result in setting taking place closer to dawn when White-chinned Petrels are known to be particularly prone to capture (Barnes *et al.* 1997). If the target CPUE decreases further it is possible that effort may increase and further exacerbate seabird bycatch in Namibian waters. A further consideration is the difference in the gear configuration used between the two fleets. The South African fishery used weights with an average mass of 6 kg spaced approximately 100 m apart and attached to multi-filament line whereas the Namibian vessels which used weights of an average mass of 3.5-4.5 kg weights spaced 88–100 m apart attached to monofilament or polysteel line. The weighted portion of the South African line sinks at a rate of 0.39 m.s^{-1} compared to 0.26 m.s^{-1} to 10 m (Chapter 12). As a consequence, the gear is within reach of the birds for longer in Namibian waters resulting in a greater likelihood of a bird becoming hooked.

White-chinned Petrels forage throughout the Southern Ocean (Weimerskirch *et al*. 1999, Berrow *et al*. 2000b) and are killed in large numbers throughout this range. Mortality has also been recorded in the Ling *Genypterus blacodes* fishery off Argentina (Favero *et al.* 2003) and in Patagonian Toothfish fisheries around the Prince Edward Islands, South Africa (Nel *et al*. 2002), off Argentina (Favero *et al.* 2003) and Chile (Moreno *et al*. 2006), and around South Georgia (Moreno *et al*. 1996) and Kerguelen Island (Delord *et al.* 2005). The cumulative mortality of multiple fisheries is likely to further impact this already vulnerable species.

In conclusion, the South African component of this fishery has a relatively small impact on pelagic seabirds, whereas seabird bycatch off Namibia may have a greater affect. This study has highlighted the paucity of information available for the region. Although this needs to be addressed, sufficient information to encourage the immediate implementation of mitigation measures such as the use of tori lines for the Namibian hake longline fishery is provided here. Interviews with the fishing industry highlighted that the lack of awareness, and therefore implementation of mitigation, should be accompanied with relevant education materials and activities. The general opinion of observers is that seabird bycatch is threatening seabirds. This may indicate their willingness to assist in education and implementation of mitigation measures.

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Chapter 5

*CHONDRICHTYAN BYCATCH IN DEMERSAL
LONGLINE AND TRAWL FISHERIES
OFF SOUTH AFRICA*

CHONDRICHTYAN BYCATCH IN DEMERSAL LONGLINE AND TRAWL FISHERIES OFF SOUTHERN AFRICA

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ABSTRACT

This study assesses the catches of sharks and skates in the South African demersal longline fishery targeting Cape hakes *Merluccius capensis* and *M. paradoxus* using fisheries dependent observer data. A total of 30 species of sharks and skates from 17 genera and eight families were caught as bycatch in the longline fishery. The four most commonly caught species were *Squalus mitsukurii* (12.0%), *Holohalaelurus regani* (5.9%), *Scyliorhinus capensis* (3.2%) and *Raja straeleni* (1.9%). These species were also caught in the demersal trawl fishery targeting Cape hakes and operating in a similar area. The overall catch rate was estimated to be 10.5, 2.19, 0.46 and 1.46 sharks per 1000 hooks or 31.5 kg, 3.3 kg, 0.7 kg and 4.4 kg per 1000 hooks in the longline fishery and 68.32 $kg.nm^{-2}$, 54.34 $kg.nm^{-2}$, 12.62 $kg.nm^{-2}$ and 358.11 $kg.nm^{-2}$ in the trawl fishery for *S. mitsukurii*, *H. regani*, *S. capensis* and *R. straeleni* respectively (nm=nautical mile=1852 m). There was no evidence of a decline in the biomass index, calculated from annual survey data, from 1986 to 2007 for *S. mitsukurii* and *H. regani*. However, the abundance index of *S. capensis* decreased by 44% on the west coast of South Africa and by 50% on the south coast and that of *R. straeleni* decreased by 69% on the west coast and by 65% on the south coast from 2000 to 2007. Management measures which could be considered for implementation are discussed.

INTRODUCTION

Demersal communities on the continental shelf off southern Africa have been relatively poorly studied (Smale *et al.* 1993, Roel 1987). Cartilaginous fishes including sharks and skates comprise a large component of the demersal fish community (Compagno *et al.* 1991, Compagno 2000). The role which demersal sharks play in the Benguela Upwelling System is not clearly understood. This is complicated by the lack of detailed knowledge of their reproductive biology, feeding ecology, abundance and distribution (Japp *et al.* 1994, Smale 1992). However, chondrichthyan life histories (generally long-lived, slow growing, late to mature oviparous, ovoviviparous or viviparous reproduction and displaying low fecundity) are poorly suited to fishing pressure and their abundances are likely to decrease rapidly if they are exploited (Hoenig & Gruber 1990, Japp *et al.* 1994, Musick *et al.* 2000, Barker & Schluessel 2005).

Many species occurring in the Benguela Upwelling System are endemic to this region or have limited geographical distribution which makes them vulnerable to over-exploitation (Gaston 1994). The problem of over-exploitation of demersal sharks is particularly acute in trawl and longline fisheries in which sharks and skates are only a relatively minor bycatch (Compagno 1990); such fisheries often continue long after the collapse of the cartilaginous fish stocks (Graham *et al.* 2001). The susceptibility of chondrichthyans to fishing pressure and consequently, stock collapse, has been widely reported (Anderson 1990, Compagno 1990, Hoenig & Gruber 1990, Stevens *et al.* 2000).

In contrast, there is evidence that fishery discards may benefit some shark species and the removal of certain species could lead to changes in the relative abundance of other species (Walker & Hislop 1998). For example, in the North Sea some species of skate decreased in abundance (Common Skate *Raja batis* and Thornback Skate *Raja clavata*), and others (Starry Skate *Raja radiata*) have increased (Walker & Hislop 1998). Similarly, *Holohalaelurus regani* in South African waters increased in abundance as a result of fishery discards (Richardson *et al.* 2000).

Many demersal sharks and skates have low commercial value. Locally, most species are therefore discarded and only a few species such as the Soupfin Shark *Galeorhinus galeus* have become commercially important (da Silva & Burgener 2007). Globally information on catches of discarded species is lacking in catch returns (Barker & Schluessel 2005). In addition, data on shark landings are seldom recorded at a species level due to problems associated with the identification. Therefore, on a global scale, few accurate data exist on the catches and landings. Due to these data limitations, there are few published studies of chondrichthyan bycatch in demersal fisheries operating globally or in South African waters (Coelho *et al.* 2003). Several species of chondrichthyan have been reported caught in trawl fisheries locally (Japp *et al.* 1994, Walmsley *et al.* 2007). Both Japp *et al.* (1994) and Walmsley *et al.* (2007) reported that discard rates of demersal shark, skate and rays in both

the inshore and deep sea trawl fisheries were as large as 95%; however certain high value products were retained: the wings of Biscuit Skate *Raja straeleni* Twineyed Skate *Raja miraletus* and the trunks of hound sharks *Mustelus* spp. and *Callorhinus capensis*.

This study assesses the catches of sharks and skates in the demersal longline fishery targeting Cape hakes *Merluccius* spp. Because a similar suite of species is likely to be caught in the demersal trawl fishery targeting the same species and operating in a similar area and depth strata, the impact of the trawl fishery is also evaluated for the four species most commonly caught by the longline fishery. Between 2000 and 2006, the total allowable catch (TAC) for hake was predominantly allocated between the trawl (90%) and the longline (6.6%) sectors (DEAT 2005). Over this period, fishing effort in these sectors averaged 60 000 trawls and approximately seven million hooks per year (Chapter 1 and 4). Other sectors that may catch a similar suite of species and are not evaluated in this study are the hake handline sector (3.4% of the TAC) and a demersal shark directed sector targeting Triakidae and Callorhinchidae (usually three active permits during this period) (DEAT 2005). The demersal shark sector may also target pelagic sharks. The combined effort of these two sectors was approximately 267 000 hooks per year (Petersen *et al.* 2007). However, no observer data for these sectors exist and the majority of species most at risk from capture by longlines are discarded at sea (da Silva & Burgener 2007). Thus the impact of these two sectors could not be evaluated; however, they are likely to be small compared to the hake longline and trawl sectors because the overwhelming majority of fishing effort is undertaken by these sectors. Fisheries independent annual survey data are used to evaluate the possible impact fishing operations may have on the population biomass index of the four most commonly caught species. Although data are limited, this study is based on the best available information and represents the first assessment of chondrichthyan bycatch in these fisheries. It also provides recommendations for their management.

MATERIALS AND METHODS

Fisheries independent sampling

Samples were collected during routine hake biomass surveys conducted by Marine and Coastal Management, Department of Environmental Affairs and Tourism (DEAT), using the research vessel FRS *Africana* off the west and south Coasts of South Africa between Port Elizabeth and the Orange River annually from 1986 to 2007. A detailed description of gear, station selection and biomass index methodology was described by Payne *et al.* (1985). The gear used over this time period varied slightly. From 1986 to 2003 and in 2006, a German 180 ft two panel trawl net with a rope wrapped chain footrope was used. The mouth opening was 2 m vertically and 26 m horizontally. It was held open by two WV doors each weighing 1500 kg which had a spread of 120 m. In 2004, 2005 and 2007 a four-panelled German 180 ft trawl net with a modified rockhopper footrope was used. The mouth of the net was 4–5 m vertically and 28 m horizontally which were held open by multipurpose trawl doors weighing

1 500 kg each and a spread of 60–70 m. Survey biomass (abundance) indices obtained from the two types of trawl gear are not directly comparable because: a) the rockhopper gear raises the belly of the net slightly above the bottom whereas a rope wrapped chain footrope does not, and; b) the door spread reduced from 120 m to 60–70 m. The latter may affect the herding behaviour of fish or sharks (R. Leslie pers. comm.). These changes in gear configuration disrupt the time series. Research trawl nets were 35 mm whereas the commercial nets are 75 mm (inshore) or 110 mm (offshore). Furthermore, all stern trawlers use rockhopper gear with at least 45 cm hoppers with 15 cm spacers. The modified rockhopper footrope used in research trawls has 20 cm hoppers and 5 cm spacers resulting in only 15 cm gap compared to 30 cm gap in commercial gear. This has implications for extrapolation from research gear to commercial fishing gear. However, it is likely to present the best available information.

Trawl stations were selected using a pseudo-random, depth-stratified sampling design. Trawls were completed during daylight hours only, and each tow was designed to last for 30 minutes; however because of irregular topography of the seabed some were of shorter duration. Both regions were surveyed annually with the exception of the south coast in 2001 and 2002. The west coast was surveyed in summer from the continental shelf and upper slope south of the international border with Namibia (*c.* 29 °S) to west of Cape Agulhas (20 °E) to the 500 m isobath. The south coast was surveyed annually in spring from the continental shelf east of Cape Agulhas (20 °E) to Port Elizabeth (26 °E) to the 500 m isobath. The entire catch of sharks and skates at each station was identified to species and weighed (to the nearest 100 g).

The trawl survey data were used to calculate catch rates ($kg.h^{-1}$) and density ($kg.nm^{-2}$) (nm=nautical mile=1852 m). The density was extrapolated for the four most commonly caught chondrichthyan species in the longline fishery to yield survey abundance indices by the swept-area method using the following equations:

$$a_{ij} = s_{ij} \times \frac{t_{ij}}{60} \times \frac{w_{ij}}{1852} \qquad (1)$$

$$d_{ij} = \frac{C_{ij}}{a_{ij}} \qquad (2)$$

$$\overline{d_{i.}} = \frac{\sum_{j=1}^{n_{ij}} d_{ij}}{n_{ij}} \qquad (3)$$

$$B = \sum_{ij} \left(\overline{d_{ij}} \times A_{ij} \right) \qquad (4)$$

Where,

i = depth stratum i.e. coast to 100 m, 100–200 m and 200–500

j = West (west of 20 ºE) or east (east of 20 ºE) coast

a = area (nm^2)

s = speed (knots = $nm.hr^{-1}$)

t = duration of trawl (min)

w = mouth width (m)

d = species density ($kg.nm^{-2}$)

C = catch (kg)

$\overline{d}$ = average density ($kg.nm^{-2}$)

n = number of samples

A = total area of strata (nm^2)

B = total biomass index (kg)

Fisheries dependent sampling

Longline data

Data were collected by fisheries observers on hake longline vessels operating in the South African fishery from 2005 to 2007. The total catch was observed continuously during the hauling process for an average of two hours or 20% per set. Observers received training on species identification prior to deployment and were equipped with reference material at sea (Petersen & Honig 2005). However, in many cases species were only identified to genus. Gear (e.g. number of hooks, length of mainline etc.) and operational (time of set and position etc.) information were recorded for each set. These data were supplemented with specialised observer data during 11 voyages (49 sets) on four commercial demersal longliners targeting hakes from March 2005 to October 2007. The total catch was recorded for 10 minutes every 20 minutes during hauling, amounting to approximately 3.5 hours or 35% per set. All specimens were identified to species. In addition random samples of shark and skate bycatch were sexed; Precaudal Length (PCL) or Disc Width (DW) were measured to the nearest mm and evaluated for its status: dead or alive on hauling.

Catch rates (number of animals per 1000 hooks) were calculated per one degree grid cell and extrapolated using the total longline effort in 2006 per one degree grid cell. The fishing effort data were obtained from vessel logbooks supplied to Marine and Coastal Management, DEAT. The total number of animals caught was converted to tonnes by averaging records for which the weights were available. The conversion factors used were 3 kg for Shortspine Spiny Dogfish *Squalus mitsukurii* and Biscuit Skate *Raja straeleni* and 1.5 kg for Izak Catshark *Holohalaelurus regani* and Yellow-spotted Catshark *Scyliorhinus capensis*.

Trawl data

Catch rates (tonnes per hour) were estimated from the survey data per one degree grid cell and extrapolated using the total commercial fishing effort (hours trawled) in 2006 per one degree grid cell. Fishing effort data were obtained from vessel logbooks supplied to Marine and Coastal Management, DEAT. There are, however differences in the gear configuration between the commercial fleet and the research trawls as detailed above. Also note that trawl catches were only calculated for those species most commonly caught in the longline fishery in order to understand the cumulative affect. For this reason, *Squalus megalops*, the most commonly caught demersal shark species in the trawl fishery is not evaluated here (Walmsley *et al.* 2007).

Statistical analysis

Counts of sharks killed by each commercial longline and survey trawl set were modelled using a generalised linear model with a Poisson distribution and logarithmic link function (McCullagh & Nelder 1989). Genstat 9 (Genstat Committee 2007) was used for model fitting and Akaike Information Criterion (AIC) was used to guide model selection (Quinn & Keough 2002). The logarithm of the parameter λ of a Poisson distribution was modelled as a linear combination of explanatory variables: e.g. for three explanatory variables, $\log \lambda = a + b_1 x_1 + b_2 x_2 + b_3 x_3$. Explanatory variables investigated included year, month, area (i.e. west of 20 ºE and east of 20 ºE) and bathymetry.

Differences in length frequency between sexes were investigated using a two sample *t*–test. Arcview GIS 3.3 was used to provide spatial representations of information.

RESULTS

A total of 30 species from 17 genera and eight families was caught as bycatch in the hake longline fishery (Table 1). By numbers, Squalidae comprised the majority (61%) of the chondrichthyan bycatch, followed by Scyliorhinidae (15.6%), Rajidae (6.6%), Carcharinidae (2.5%), Triakidae (0.6%), Lamindae (0.3%), Callorhinchidae (0.02%) and Cetorhinidae (0.003%).   The four most commonly caught species were *Squalus mitsukurii* (12%), *Holohalaelurus regani* (5.9%), *Scyliorhinus capensis* (3.2%) and *Raja straeleni* (1.9%). 60.9% of Squalidae, 18.6% of Scyliorhinidae, 16.7% of Triakidae and 31.8% of Rajidae caught were identified to genus (Table 1). As a result all catch rates reported for individual species represent minimum catch rates by the commercial longline fleet.

Squalidae, Scyliorhinidae, Cetorhinidae and Rajidae were discarded. Carcharinidae and Lamindae were occasionally retained.   Triakidae and Callorhinchidae were frequently retained because of their commercial value. Most (87%, n=138) Squalidae, Scyliorhinidae, Cetorhinidae and Rajidae were alive when hauled on board. However they were frequently dead by the time they were discarded: firstly because they suffered fatal injuries while being hauled through the line rollers and secondly because fishing operations focused on the target catch so that bycatch species were typically neglected on deck or in the hauling basin.

Bycatch data and five explanatory variables (vessel affect, year, month, area i.e. west of 20 ºE and east of 20 ºE, and depth strata i.e. coast to 100 m, 100–200 m and 200–500 m) were available for 847 commercial longline sets and four explanatory variables (year, month, area i.e. west of 20 ºE and east of 20 ºE, and depth strata i.e. coast to 100 m, 100–200 m and 200–500 m) for 6 551 research trawls. The four species most commonly caught in the South African demersal longline fishery were each modelled separately.

Table 1/…

Table 1: Species composition of chondrichthyan bycatch from South African demersal longline activities, 2005–2007.

Family	Genus	Species	Common name	%
Squalidae	*Squalus*	*mitsukurii*	Shortspine Spiny Dogshark	12.0%
	Squalus	*megalops*	Shortnose Spiny Dogshark	1.2%
	Squalus	*acanthias*	Spotted Dogshark	1.1%
	Squalus		unidentified spiny dogsharks	38.0%
	Centroscyllium	*fabricii*	Black Dogshark	1.1%
	Centroscymnus	*crepidater*	Longnose Velvet Dogshark	0.2%
	Etmopterus		Unidentified lanternshark	0.1%
	Etmopterus	*lucifer*	Black Lucifer	0.02%
	Etmopterus	*brachyurus*	Shorttail Lanternshark	0.04%
			unidentified dogsharks	7.3%
Sub-total				**61.0%**
Scyliorhinidae	*Holohalaelurus*	*regini*	Catshark Izak	5.9%
	Scyliorhinus	*capensis*	Yellow-spotted Catshark	3.2%
	Haploblapharus	*edwardsii*	Puffadder Shy Shark	2.2%
	Haploblapharus		Unidentified shy sharks	0.6%
	Halaelurus	*natalensis*	Tiger Catshark	0.0%
	Poraderma	*pantherium*	Leopard catshark	0.7%
	Poraderma	*africanum*	Pajama (striped) Catshark	0.3%
			unidentified catsharks	2.9%
Sub-total				**15.6%**
Triakidae	*Galeorhinus*	*galeus*	Soupfin Shark	0.5%
	Mustelus	*mustelus*	Houndshark	0.03%
	Mustelus	*palumbes*	White Spotted Houndshark	0.003%
	Mustelus		unidentified houndshark	0.1%
Sub-total				**0.6%**
Carcharhinidae	*Prionace*	*glauca*	Blue Shark	2.4%
	Carcharhinus		Unidentified reef shark	0.1%
	Carcharhinus	*brachyurus*	Bronze Whaler	0.02%
Sub-total				**2.5%**
Lamnidae	*Isurus*	*oxyrinchus*	Mako Shark	0.3%
Callorhinchidae	*Callorhincus*	*capensis*	St Joseph's Shark	0.02%
Cetorhinidae	*Cetorhinus*	*maximus*	Basking Shark	0.003%
Unidentified sharks			unidentified sharks	13.4%
Sub-total				**13.7%**
Rajidae	*Raja*	*straeleni*	Biscuit Skate	1.9%
	Leucoraja	*wallacai*	Yellowspot Skate	1.3%
	Dipturus	*pullopunctata*	Slime Skate	0.7%
	Dipturus	*springeri*	Roughbelly Skate	0.3%
	Rostroraja	*alba*	Spearnose Skate	0.1%
	Rajella	*caudaspinosa*	Munchkin Skate	0.01%
	Rajella	*ravidula*	Smoothback Skate	0.01%
	Cruriraja	*parcomaculata*	Roughnose Legskate	0.03%
			unidentified rays	2.2%
Sub-total				**6.6%**
Total				**100.0%**

Table 2: Summary of average catch rates for South African demersal trawl (kg.nm^{-2}) and longline (no. per 1000 hooks) operations, 2005–2007.

Longline	*Squalus mitsukurii*			*Holohalaelurus regani*			*Schyliorhinus capensis*			*Raja straeleni*		
Depth	West	South	Total	West	South	Total	West	South	Total	West	South	Total
100 m to 200 m	53.57	0.04	2.50	1.50	0.12	0.18	26.79	0.09	1.32	250.37	0.44	11.93
200 m to 500 m	1.70	0.82	1.66	3.94	2.57	3.88	0.57	0.81	0.58	0.41	0.43	0.41
500 m to 1000 m	0.13	-	0.13	0.12	-	0.12	0.01	-	0.01	0.03	-	0.03
Total	10.82	8.22	10.50	2.41	0.62	2.19	0.49	0.24	0.46	1.61	0.44	1.46

Trawl	*Squalus mitsukurii*			*Holohalaelurus regani*			*Schyliorhinus capensis*			*Raja straeleni*		
Depth	West	South	Total	West	South	Total	West	South	Total	West	South	Total
coast-100m	5.71	1.42	2.16	0.80	1.18	1.11	0.01	2.40	1.99	66.68	529.44	449.75
100 m to 200 m	56.94	12.38	32.79	49.42	32.11	40.04	1.42	24.85	14.12	191.88	647.04	438.57
200 m to 500 m	134.53	378.80	158.96	123.67	81.87	119.49	13.86	60.05	18.47	219.86	96.11	207.48
500 m to 1000 m	23.90	941.68	166.23	4.97	2.10	4.53	0.63	60.52	9.91	16.51	34.14	19.24
Total	90.77	41.91	68.32	81.19	22.75	54.34	7.54	18.60	12.62	188.58	557.51	358.11

Squalus mitsukurii

Generalised linear modelling of *S. mitsukurii* catches accounted for 88.4% of the variance in the case of the longline data and 22.9% of the variance for the trawl data. The overall rate was estimated to be 10.5 or 31.50 kg per 1000 hooks (commercial longline data) or 3.96 kg.hr^{-1} or 68.32 kg.nm^{-2} (research trawl data) (Table 2). In the case of the commercial longline data, there was a significant vessel effect (p=0.05 or p<0.005, significance level referred to throughout) which accounted for 23.4% of the variance. There were also significant differences between areas which were consistent between the two data sets and accounted for 39.9% and 3.0% of the variance for longline (p<0.001) and trawl (p<0.001) data respectively. *S. mitsukurii* were caught at higher catch rate on the west coast (10.82/1000 hooks or 90.77 kg.nm^{-2}) compared to the south coast (8.22/1000 hooks or 41.91 kg.nm^{-2}) (Table 2, Fig. 1 and 2). Depth strata accounted for 24.9% and 16.5% of the variance for longline (p<0.001) and trawl (p<0.001) data respectively. In the case of the longline data *S. mitsukurii* were caught at the highest catch rate in the 100–200 m depth strata on the west coast (53.57/1000 hooks) and in the 200–500 m depth strata on the south coast (0.82/1000 hooks) (Table 2). The trawl data revealed highest catch rates in the 200–500 m depth strata on the west coast (134.53 kg.nm^{-2}) and 500–1000 m depth strata on the south coast (941.68 kg.nm^{-2}) (Table 2).

The total catch was estimated in 2006 to be 353 t in the trawl fishery and 722 t in the longline fishery. Annual differences in catch rates accounted for 5.3% of the variance in the trawl data and although the differences between the years were significant (p<0.001) they did not reveal a trend from 1986 to 2007 (Fig. 3). The coefficient of variation of the annual biomass index averaged 28%. The variable 'month' was not significant and was therefore not included in the model.

Holohalaelurus regani

Generalised linear modelling of *H. regani* catches accounted for 61.5% of the variance in the case of the longline data and 33% of the variance in the case of the trawl data. The overall catch rate was 2.19 or 3.3 kg per 1000 hooks for the commercial longline data and 54.34 kg.nm^{-2} for the research trawl data (Table 2). There were significant differences between areas which were consistent between the two data sets and accounted for 27.9% and 12.6% of the variance respectively. *H. regani* were caught at higher catch rates on the west coast (2.14/1000 hooks and 81.19 kg.nm^{-2}) compared to the south coast (0.62/1000 hooks and 22.75 kg.nm^{-2}) (Table 2, Fig. 1 and 2). The variable 'depth strata' was also significant (p<0.001) for both data sets and accounted for 11.6% and 16.5% of the variance for longline and trawl data respectively. *H. regani* were predominantly caught between the 200 and 500 m depth contours on both the west and south coasts for both longline and trawl data (Table 2).

The total catch in 2006 was estimated to be 390 t in the trawl fishery and 38 t in the longline fishery. Although the variable 'year' accounted for 29% and 1.8% and 'month' 14.4% and 0.9% of the variance in the longline and trawl data respectively it was not significant and did not reveal a trend. There was no significant trend in biomass index estimate from 1986 to 2007 (Fig. 3). The coefficient of variation of the annual biomass index averaged 11%.

Scyliorhinus capensis

Generalised linear modelling of *S. capensis* catches accounted for 64.2% of the variance in the case of the longline data and 22.1% of the variance in the case of the trawl data. The overall catch rate was 0.46 or 0.7 kg per 1000 hooks in the commercial longline fishery and 12.62 kg.nm^{-2} in the research trawl data. There are no significant differences between catch rates by longliners on the south and west coast (t=−1.31, df=746, p=0.191) (Table 2, Fig. 1). There was however a significant difference between these two areas for the trawl data. *S. capensis* were caught at a higher rate on the south coast (18.6 kg.nm^{-2}) than on the west coast (7.54 kg.nm^{-2}) (Table 2, Fig. 2). The variable 'depth strata' accounted for 12.8% and 12.7% of the variance for longline and trawl data respectively. In the case of the longline data *S. capensis* were predominantly caught between 100 and 200 m on the west coast and between 200 and 500m on the south coast. The trawl data however revealed a higher density in trawled areas in the 200 to 500 m depth strata on the west and south coasts.

Although the variable 'month' accounted for 6.0 % and 0.8 % of the variance in the longline and trawl data respectively it was not significant and was therefore not included in the model.

The total catch was estimated in 2006 to be 92 t in the trawl fishery and 25 t in the longline fishery. There was a significant decreasing trend in biomass index estimate from 1986 to 2007 (Fig. 3) which accounted for 3.8% and 3.3% for longline and trawl data respectively. Furthermore confidence in the trend is supported by an average coefficient of variation of the annual biomass index of 19%. The biomass index increased from 1986 till 2000 when it reached a maximum of 873 t on the west coast and 2495 t on the south coast (Fig. 3). Although it fluctuated a little over the time period, there was an overall decrease of 44% to 492 t on the west coast and 50% to 1240 t on the south coast between 2000 and 2007.

Raja straeleni

Generalised linear modelling of *R. straeleni* catches accounted for 96.1% of the variance in the case of the longline data and 21.0% of the variance in the case of the trawl data. The overall catch rate was 1.46 sharks or 4.38 kg per 1000 hooks (commercial longline data) and 358.11 kg.nm^{-2} (research trawl data) (Table 2). There were no significant differences between catch rates by longliners on the south and west coasts (t=–0.87, df=750, p=0.387) (Table 2, Fig. 1). There were however significant differences between research trawl survey data catch rates on the south (557.51kg.nm^{-2}) and west (188.58 kg.nm^{-2}) coast (Table 2, Fig. 2). Although the variable 'month' accounted for 2.2% and 1.1% of the variance in the longline and trawl data respectively it was not significant and did not reveal a trend.

The total catch was estimated in 2006 to be 2 546 t in the trawl fishery and 69 t in the longline fishery. There was a significant (p<0.001) decreasing trend in biomass index estimate from 1986 to 2007 (Fig. 3) which accounted for 0.1% and 4.7% for longline and trawl data respectively. Furthermore confidence in the trend is supported by an average coefficient of variation of the annual biomass index of 19%. The biomass index was relatively stable until 1999 for both the west and south coasts, but showed a sharp decline from 2000 onwards when it decreased from 10 525 t to 3 244 t in 2007 on the west coast, an overall decrease of 69.2%, and a decrease from 9 893 t to 3 396 on the south coast, an overall decrease of 65.2% (Fig. 3).

Figure 1/...

Figure 1: Estimated total catch (numbers of individuals) of *Squalus mitsukurii*, *Holohalaelurus regani, Scyliorhinus capensis* and *Raja straeleni* by the commercial longline fleet per 1° grid square in 2006.
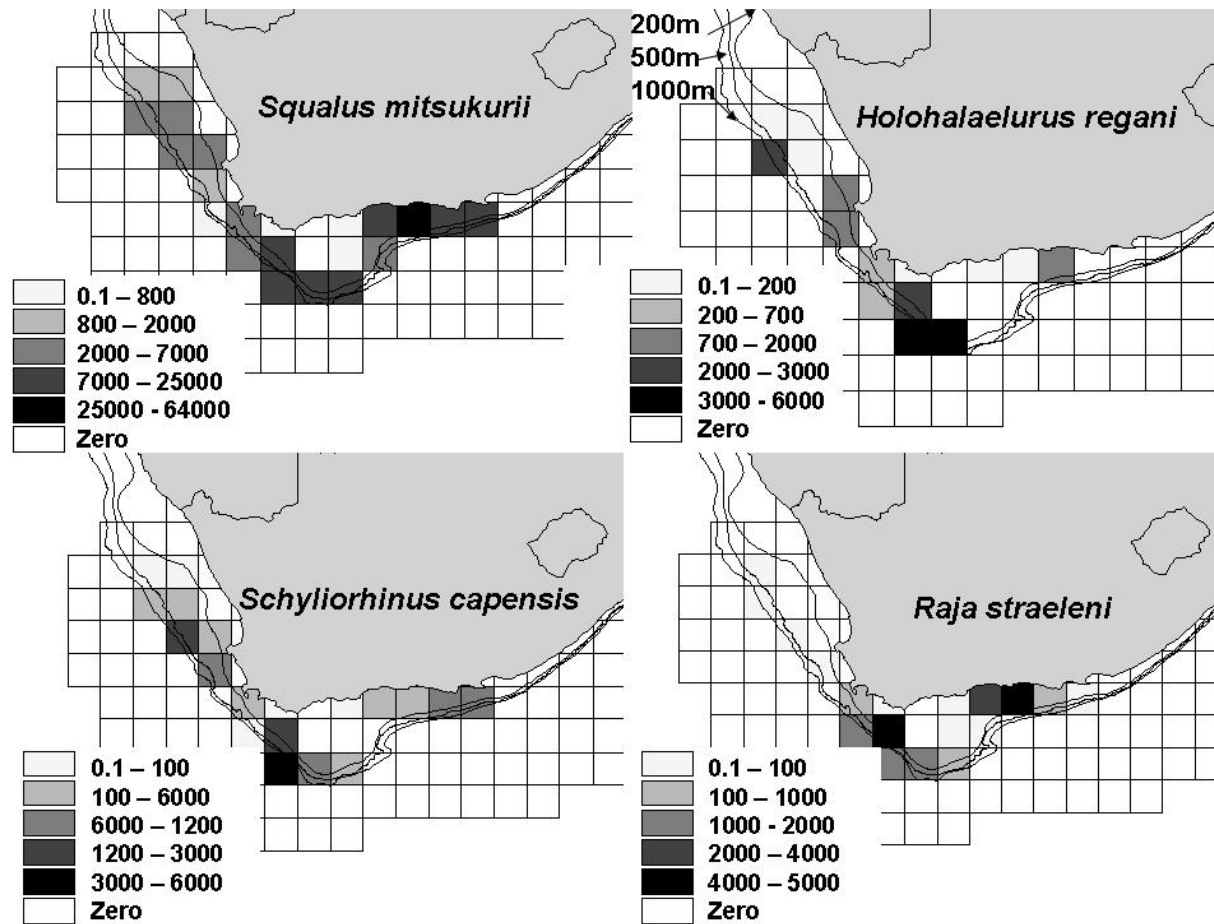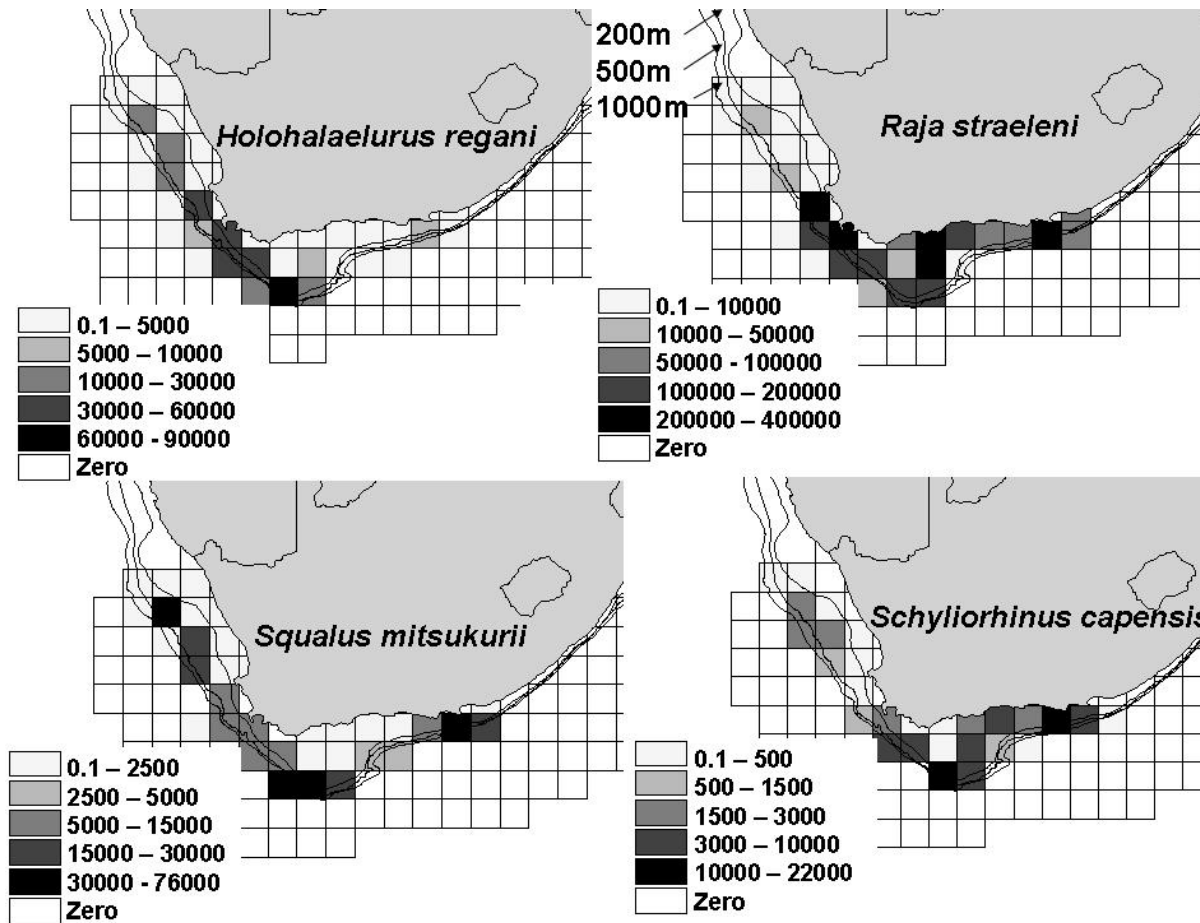
Figure 2: Estimated total catch (kg) per 1° grid square of *Squalus mitsukurii*, *Holohalaelurus regani, Scyliorhinus capensis* and *Raja straeleni* using research trawl data to calculated catch rates and raised to total commercial trawl effort in 2006.
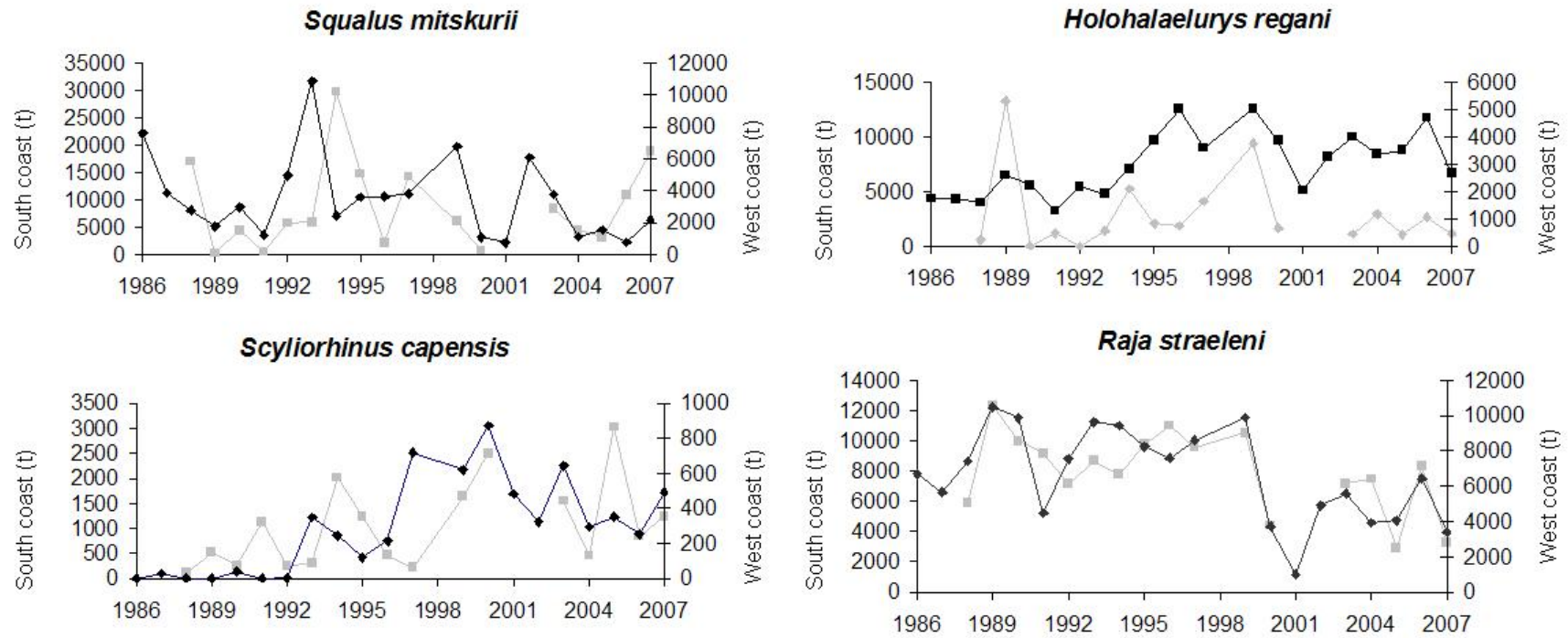
Figure 3: The trend in biomass index for *Squalus mitsukurii*, *Holohalaelurus regani, Scyliorhinus capensis* and *Raja straeleni,* calculated from annual research trawls on the south (grey) and west (black) coasts of South Africa from 1986 to 2007.

Length frequency

Two species, *Squalus mitsukurii* and *Holohalaelurus regani*, were caught in sufficient numbers to investigate difference in the length frequency between sexes. More female (80.4%, n=628) *S. mistukurii* were caught than males (19.6%, n=153). Females were significantly larger (average 46.2 cm, range 20.0–80.0, SD 7.2) than males (average 39.1 cm, range 31.0–60.0, SD 4.0) (t=16.06, p<0.001, df=399.40). A similar number of female (51.7%, n=105) and male (48.3%, n=98) *H. regani* were caught. No difference in length between the sexes was observed (t=–0.12, p=0.91, df=201). The average length of *H. regani* was 53.0 cm (range 25–80 cm, SD 10.7 cm). The length of *S. capensis* averaged 64.7 cm (range 33–82 cm, std dev 9.6 cm, n=34, 28 females and 6 males). The average disc width of *R. straeleni* was 53.2 cm (range 36–6 cm, SD 7.7 cm, n=45).
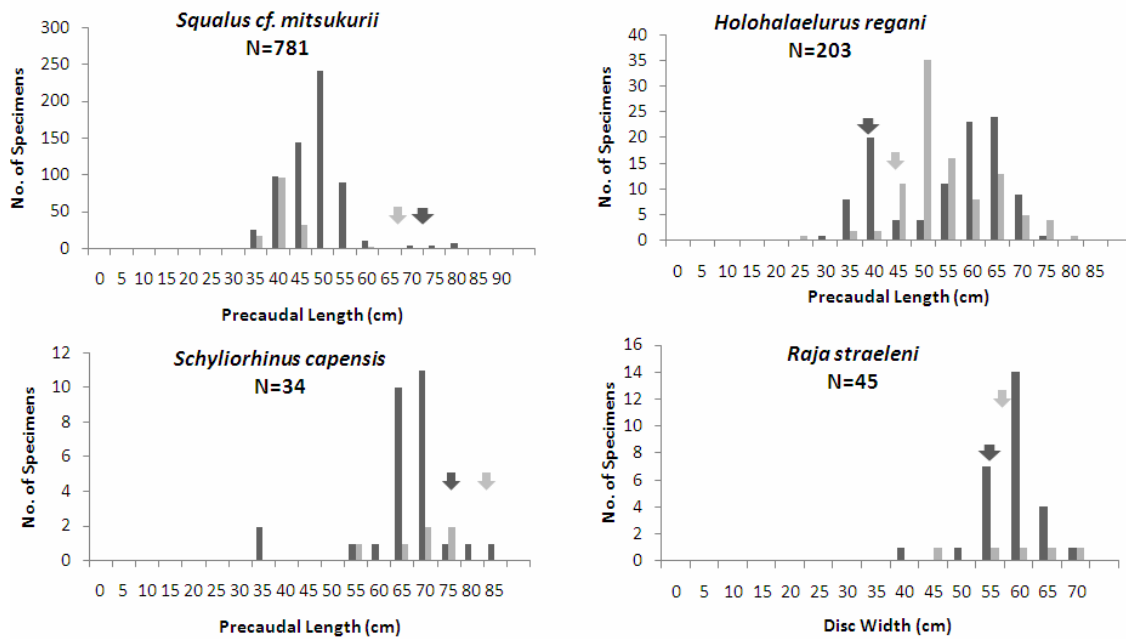
Figure 4: Length frequency distributions of *Squalus mitsukurii*, *Holohalaelurus regani, Scyliorhinus capensis* and the *Raja straeleni* caught as bycatch in the commercial longline fishery. Males are represented by pale grey bars and females by dark grey bars. Pale and dark grey arrows indicate size at maturity for males and females respectively (Richardson *et al*. 2000 (*H. regani*), Ebert *et al*. 2006 (*S. capensis*), Ebert *et al*. 2007 (*R. straeleni*), Compagno 1984 (*S. mitsukurii*)).

DISCUSSION

Squalus mitsukurii

Catch rates were the highest on the west coast which is consistent with the known distribution of this species i.e. Angola to KwaZulu-Natal, South Africa (Compagno *et al*. 1989). The catch rates were higher closer inshore (100–200 m) for the longline fishery than for the trawl fishery (200–500 m on the west coast and 500–1000 m on the south coast). The differences between the fisheries may represent the habitat preference of the fisheries. Longline fisheries tended to focus their effort on hard grounds (especially in the early years of the fishery i.e. 1998–2003) compared to the trawl fishery which is largely limited to softer grounds (Japp & Wissema 1999). *Squalus mitsukurii* has been reported throughout the depth range 4–954 m, but mostly between 100–700 m (Compagno *et al*. 1989). The catch consisted predominantly of immature males although the confidence in aging these animals is reduced because there are considerable differences in size at maturity between adult males and females of various populations reported in the literature (Compagno 1984, Wilson & Seki 1994).

No decrease in the biomass index of *S. mitsukurii* was evident over the 20 year period. Thus, despite being the most commonly caught chondrichthyan bycatch species in the hake longline fishery, it appears to be sufficiently robust to withstand this level of mortality. A survey conducted off the west coast of the South Island, New Zealand, report similar findings of little impact of fishing operations on the biomass for this species (Anderson *et al.* 1998). The species' resilience to over-exploitation by trawl and longline operations could be explained by its non-selective feeding behaviour (Kaiser *et al.* 1994). *S. mitsukurii* diet consists of hake, Lantern Sharks *Etmopterus* spp, Snoek *Thyrsites atun* and Conger Eels *Conger conger* (Ebert *et al.* 1992, Bianchi *et al.* 1999), cephalopods and crustaceans (Fischer *et al.* 1990, Ebert *et al.* 1992). These species include both the catch and discard of both fisheries (Japp *et al.* 1994, Walmsley *et al.* 2007), and thus possibly supplement the diet of *S. mitsukurii* .

This finding is nevertheless surprising given their biology which is considered not to be sufficiently fecund to withstand continued exploitation pressure (Garcia *et al.* 2008). It is ovoviviparous, its gestation period is up to two years and its litter size is 4–9 pups (Compagno 1984). It is long-lived; males live up to 18 years and females 27 years of age (Wilson & Seki 1994). Garcia *et al.* (2008) estimated that the average fishing mortality needed to drive a deep-water chondrichthyan species, including squaliformes, to extinction ($F_{extinct}$) was 38–58% of that estimated for oceanic and continental shelf species, respectively.

Moreover, *S. mitsukurii* is commonly caught in demersal fisheries elsewhere where there has been evidence of population declines. For example, catches of *S. mitsukurii* declined by 97% between 1976 to 1977 and 1996 to 1997 in a heavily trawled area off New South Wales,

Australia, where it is classified as Endangered (Graham *et al.* 2001, Daly *et al.* 2002). Furthermore, a demersal gillnet fishery for the Little Gulper Shark *Centrophorus uyato* was terminated in Western Australia in the mid 1990s due to declines in the catches of *S. mitsukurii.* The catches of *S. mitsukurii* on the Hancock Seamount in the Western North Pacific declined by 80% between 1985 and 1988, suggesting vulnerability of populations of this species at small seamounts to overfishing (Wilson & Seki 1994). A closely related species, the spiny dogfish *S. acanthias*, is vulnerable to the deleterious effects of overfishing on Beerkircher *et al.* (2002).

Holohalaelurus regani

H. regani were caught at a higher rate on the west coast in both the trawl and longline fisheries. This is consistent with the known distribution which extends from southern Namibia to southern Mozambique where they are endemic (Compagno *et al.* 1989). They were predominately caught between the 200–500 m isobaths which overlaps with their known depth range of 160–740 m. Nursery areas are found inshore and females also have a more inshore distribution than males (Richardson *et al*. 2000). This may explain the sexual and age bias reported in this study i.e. mainly males in deeper waters.

No trend in biomass index was observed from 1986 till 2007. This finding is consistent with that of Richardson *et al*. (2000), who reported an increase in the biomass index from 1986 to 1993, despite a high bycatch rate by the demersal trawl fishery. This is most likely to be due to a combination of factors that yield *H. regani* less susceptible to over-exploitation by fisheries (Richardson *et al.* 2000). Firstly the species is relatively fecund and breeds all year (Richardson *et al.* 2000). Secondly, they are opportunistic feeders and may benefit from offal discarded by the trawl and longline fishery (Richardson *et al*. 2000).

Scyliorhinus capensis

Catch rates were the highest on the south coast which is consistent with the known distribution of this species, which is endemic to the coast from Lüderitz, Namibia, to central KwaZulu–Natal, South Africa (Compagno *et al.* 1989). The highest catch rates were reported in the 200–500 m depth zone which overlaps with the known depth distribution of 26–495 m (Compagno *et al.*1989).

The hake longline fishery commenced as a commercial fishery in 1998, shortly before the observed decrease in biomass index of 44% on the west coast and 50% on the south coast (DEAT 2005). This fishery predominantly operates on hard ground (Japp 1993, Japp *et al.* 1994) which is the preferred habitat for this species (Compagno *et al*. 1989). Catches in 2005 and 2006 were predominantly immature animals, which may further indicate an over-exploited population. There are untrawlable rocky reef areas within the range of this species, which prior to the introduction of the longline fishery, may have acted as refuges for *S.*

capensis. The extent to which the gear change in 2005, 2005 and 2007 affected this trend is unknown, but since the decrease is sustained since 2000 it is considered minimal.

S. capensis has a particularly vulnerable life history. It is oviparous and only lays one egg from each of the paired oviducts at a time although the rate of deposition per year is unknown (Compagno 1984). The combination of a vulnerable life history and the introduction of a fishery predominantly operating on its known habitat are likely to be the leading causes for the observed decrease in biomass index for this species.

Skates

Three species of skate which are endemic to South Africa were identified by observers: Slime *Dipturus pullopunctatus*, Munchkin *Rajella caudaspinosa* and Yellowspotted *Leucoraja wallacei* skates. These are therefore especially vulnerable to the impact of South African trawl and longline fisheries. The most frequently caught species, *R. straeleni*, is the most common skate found in South African waters (Compagno *et al.* 1991). *R. straeleni* is found from Namibia to Algoa Bay, South Africa, and further north along the West African coast. It occurs at an average depth of 353 m in southern Africa and 690 m in West Africa. It is oviparous, and demonstrates a low population doubling time of 4.5–14 years (Dulvy & Reynolds 1997).

This study reports declines of 69% on the west coast and 65% on the south coast of South Africa. It was not possible to quantify the extent to which the gear change in 2004, 2005 and 2007 contributed to this observed decrease, but is considered not to have affected the overall trend since the decrease is sustained since 1999. This species is vulnerable to trawl fisheries because its preferred habitat is soft bottom substrates; therefore it is not surprising that a large proportion of the trawl bycatch consists of this species (Ebert & Sulikowski 2007). Ebert and Sulikowski (2007) reported that trawl fisheries in the North Atlantic have impacted the abundance, population structure, and distribution of several skate species, causing several to be given commercially prohibited status. Skates and rays are also a common bycatch in the prawn trawl fishery off Argentina which has reported landings of up to 15 000 t per year. Although rajids are released alive at sea, post-catch mortality is unknown (Cedrola *et al.* 2004). It has been suggested that the vulnerability of skates to extinction increases with an increase in body size (Dulvy & Reynolds 2002), and that the removal of larger skates may lead to an increase in the abundance of smaller skates due to greater food availability (Dulvy *et al.* 2000).

Conclusion and management recommendations

The South African demersal longline and trawl fisheries collectively catch a large diversity of cartilaginous fishes, most of which are discarded. The impact of trawling and longlining on the biomass of South African continental slope sharks and skates appears to have followed a

similar pattern to most exploited demersal sharks stocks worldwide, particularly *S. capensis* and *R. straeleni*.

No management controls have been implemented for any species of demersal shark or skate in the hake trawl and longline fisheries to date. The fishery however, is managed by a Total Allowable Catch (TAC) and Effort (TAE) which will indirectly restrict the number of sharks and skates caught as bycatch (DEAT 2005). The lack of direct management measures should be reviewed in light of the decreases in *S. capensis* (endemic to Southern Africa) and *R. straeleni* reported in this study.

Release of live animals

At least in the case of longline fisheries, animals are frequently alive on capture. Fishers should be required to release unwanted live chondrichthyan bycatch rapidly. Post release survival is unknown and delayed effects such as the greater vulnerability to predation (Davis & Olla 2001), delayed infection (Neilson *et al.* 1989) and immuno-suppression (Lupes *et al.* 2006) have been observed elsewhere and in other species (Davis 2002). Mandelman and Farrington (2007) evaluated post capture survival of Spiny Dogfish *Squalus acanthias* in the Northwest Atlantic trawl fishery for 72 hours after capture and found that this species was relatively resilient to the effect of trawling although it suffered high mortality rates when the trawl net was heavy (full). The nature of the life history of sharks makes them vulnerable to extinction; this fact, and the observed population declines, should be communicated to fishers to facilitate implementation of this measure.

Gear manipulation

Demersal longline fishing gear is held on or near the bottom by means of weights. In the case of longline fisheries, it was found that the catch rate of the four most commonly caught species of shark and skate was higher on the portion of line nearest to the weight, compared to the suspended portion of the line near the dropper (Chapter 12). A similar finding was reported in the longline fishery for hake *M. merluccius* in southern Portugal, where catch rates of sharks (e.g. Blackmouth Catshark *Galeus melastomus*) were significantly higher near the weight, whereas the catch of hake was higher on the mid portion of the line (Coelho *et al.* 2003). This study found that shark catches could be reduced by reducing the numbers of hooks close to the weight (Coelho *et al.* 2003). This measure could be considered for the South African fishery and is unlikely to affect target catches because hake was equally distributed on the weight and the dropper portion of the line (Chapter 12). Such an experiment could alternate a section of line with 10 hooks removed on either side of the weight with a section where hooks were not removed and catches compared. The loss of the total number of hooks could be compensated for by adding further sections to the line.

In the case of trawl fisheries, Bycatch Reduction Devices (BRDs) could be tested and if found effective, implemented. BRDs have been used elsewhere, particularly in prawn trawl

fisheries, for the reduction of unintended bycatch including chondrichthyans (Watson *et al.* 1986, Kaiser *et al.* 2000, Hall & Mainprise 2005, Brewer *et al.* 2006, Fennessy & Isaksen 2007). In general two types of BRD exist depending on whether they are designed to release large animals such as elasmobranchs and turtles or smaller fish; both were essentially developed to decrease bycatch in the prawn trawl fishery (Isaksen *et al.* 1992, Brewer *et al.* 1998, Broadhurst 2000). The former guides larger animals out of the net through an escape panel by employing a physical obstruction such as a solid grid into the entrance of the cod-end. The latter depends on the behavioural differences between prawns and fish, where fish continue to swim actively in the net, and eventually locate an escape opening and are thereby released. The former is more likely to be appropriate to decrease chondrichthyan bycatch in the South African hake trawl fishery.

The efficacy of BRDs to release elasmobranchs has been tested, although these studies were limited to prawn trawl fisheries. For example, a Nordmøre grid was found to successfully release large (>5kg) elasmobranchs, but had limited effect on smaller elasmobranchs which passed through the grid and into the cod-end of the net (Brewer *et al.* 1998, Fennessy & Isaksen 2007). These studies proposed that a greater size range of elasmobranchs may have been released if the spacing between the bars of the grid were smaller than the tested 10 cm (Brewer *et al.* 1998, Fennessy & Isaksen 2007). The application of BRDs for the release of demersal chondrichthyans has not as yet been tested.

Habitat protection

In the past closed areas or marine protected areas have been used conservatively as a fishery management tool mainly because fisheries managers have tended to focus their attention on abundant species with high biological productivity, whereas closures are a more essential management tool for managing less abundant species with low biological productivity such as chondrichthyans (Stevens 2002, Walker 2004). Closed areas have been used in the management of chondrichthyans catches elsewhere. For example, closed areas to shark longline fishing were instituted in 1954 and extended in the 1960s in the inshore waters of northern and south-eastern Tasmania. These closures were implemented to protect the nursery areas of school shark (*Galeorhinus galeus*) (Punt & Walker 1998). A system of 10 closed areas were established in 2002 in New South Wales, Australia to avoid unintentional capture of the grey nurse shark (*Carcharias taurus*) by longline fishing (Walker 2004).

At present no offshore marine protected area exists in South African waters and therefore no demersal shark or skate vulnerable to high levels of mortality from fishing activities has any formal habitat protection. Demarcating boundaries for closed areas requires extensive data to provide detailed information on distribution and biological condition of vulnerable species (Walker 2004). Limited biological and distribution data are available for *S. capensis* and *R. straeleni*, the two species of most concern from longline fishing for hake.

Prior to the introduction of the longline fishery in 1998, untrawlable, rocky reef grounds were essentially protected and provided a refuge for these species. Since the introduction of longline fisheries the following no fishing areas exist: no trawling or longlining for hake may take place within 5 nautical miles of the coastline or within False Bay on the west coast and no longlining may take place within water depths less than 110 m or within 20 nautical miles from the coast (whichever is the greater distance from the coast) on the south coast, although trawling does take place here (DEAT 2005). Furthermore, no trawling or longlining may take place during the period 1 September to 30 November within the quadrilateral described by lines joining the following four points: 34°48'S 24°00'E, 34°38'S 25°00'E, 34°44'S 25°00'E and 34°57'S 24°00'E (DEAT 2005). Although this measure was implemented for the protection of Kingklip, a collapsed linefish stock, they could serve as a refuge for demersal sharks, especially the over exploited *S. capensis*. It is unlikely to protect *R. straeleni* which is found in deeper waters of on average 353 m (Compagno *et al.* 1989).

This study was based on the best available information, albeit limited, and highlights the need for further investigation of the impact of longline and trawl fisheries on demersal chondrichthyans. Improved species identification is essential as well as the development of dedicated data collection protocols for chondrichthyan catch in both fisheries dependent and independent sampling.

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Chapter 6

INTERACTIONS BETWEEN SEABIRDS AND DEEP-WATER HAKE TRAWL GEAR: AN ASSESSMENT OF IMPACTS IN SOUTH AFRICAN WATERS

INTERACTIONS BETWEEN SEABIRDS AND DEEP-WATER HAKE TRAWL GEAR: AN ASSESSMENT OF IMPACTS IN SOUTH AFRICAN WATERS

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ABSTRACT

Many seabirds are killed or injured by fishing gear, contributing to the high proportion of threatened seabirds. This study estimates the impact of the South African deep-water hake *Merluccius* spp. trawl fishery on seabirds. At least 30 birds were killed in 190 hours of dedicated observations of trawl warps during 2004 and 2005. Most were killed when their wings were entangled around the trawl warp and they were dragged under by the force of the water passing over the warp. Albatrosses were killed most frequently: shy albatrosses *Thalassarche cauta* (43% of all birds killed) and black-browed albatrosses *T. melanophrys* (37%), with smaller numbers of white-chinned petrels *Procellaria aequinoctialis* (10%), Cape gannets *Morus capensis* (7%) and sooty shearwaters *Puffinus griseus* (3%). Mortalities occurred mainly during dumping of fishery wastes, and were more frequent in winter, when more birds attended fishing vessels. Average mortality rates were 0.56 (95% CI 0.32-0.82) birds killed per hour during dumping in winter, 0.21 (0.07-0.38) during dumping in summer, and 0.09 (0.02-0.19) when not dumping in winter. No birds were killed in the absence of dumping in summer. Albatrosses suffered a disproportionately high mortality rate, with 15% of birds dragged under drowning, compared to 4% of all other species. Deaths resulting from entanglement in fishing nets mainly affected Cape gannets *Morus capensis*, and occurred at an average rate of 3.0 (0.9-5.4) birds per 100 trawls (n=331 trawls). Serious warp incidents were independent of age among albatrosses, but there was a tendency for immature gannets to have a higher interaction rate than adults. Crude extrapolation suggests that total mortality is some 18,000 (8,000-31,000) birds per year, of which 85% are killed on warps and 15% entangled in nets. These estimates are of the same order of magnitude as estimates of long-line bycatch in South African waters. Mitigation measures have been implemented to reduce mortality in this fishery.

INTRODUCTION

Seabirds have a disproportionately large proportion of threatened species, in part because many species face threats both at sea and on their breeding sites (BirdLife International 2004). Over the last few decades there has been considerable concern about the impacts of long-line fishing on seabird populations (Brothers, 1991; Nel, Ryan & Watkins, 2002). Thousands of birds are caught and drowned on long-lines annually, and the magnitude of this mortality is sufficient to account for the observed decreases in several threatened seabird populations (BirdLife International 2004). Fatal seabird interactions with demersal trawl warps were first reported by Bartle (1991), but these fisheries have received less attention, mainly because the problem was perceived to be much smaller. Initial reports implicated netsonde cables in most mortalities (Bartle 1991). Netsondes, monitoring devices deployed at the entrance to nets, are rarely used in the South African deep-water hake fishery, so it was assumed that this fishery had few negative impacts on seabirds. More recently, collisions with trawl warps (the steel cables used to tow nets) have been recognised as a significant problem in trawl fisheries around major seabird breeding islands such as Kerguelen (Weimerskirch, Capdeville & Duhamel, 2000) and the Falkland Islands (Sullivan and Reid 2002, 2003; Sullivan, Reid & Bugoni, 2006), raising concerns about impacts in other areas where trawlers attract large numbers of scavenging seabirds.

The demersal trawl fishery for hakes *Merluccius* spp. off the west and south coasts of South Africa is substantial, with 79 vessels in the offshore fleet in 2005 making some 60,000 trawls per year. Discards of offal and non-target species provide food for a range of pelagic seabird species, probably resulting in fundamental changes in seabird communities off South Africa since the fishery commenced in the 1950s (Abrams, 1983, 1985; Ryan and Moloney 1988). Some birds have been reported to be killed either entangled in nets or caught on warp splices, but the numbers killed were assumed to be trivial. However, recent studies have shown that accurate estimates of mortality require dedicated observation of trawl warps, as only a small proportion of birds killed are hauled aboard (Sullivan *et al*., 2006). This study assesses the impact of the South African deep-water hake fishery on seabirds and identifies factors that exacerbate seabird mortality.

METHODS

Data collection

The South African deep-water hake fishery targets *Merluccius paradoxus* and *M. capensis* along the continental shelf off the west and south coasts of South Africa (Fig. 1). A fleet of 79 stern-trawlers operates year round to catch about 140 000 tonnes of hake annually. Three dedicated seabird observers specifically trained to detect seabird mortalities were placed on commercial trawlers operating in from mid-2004 to the end of 2005. Trawls were observed throughout the major fishing area (Fig. 1), although most trawls were observed between

Cape Town and Danger Point. During daylight hours, observers estimated the numbers of birds attending vessels in relation to fishing activity, recording the numbers of birds within 50 m of the stern of the vessel. Counts were made prior to setting the net and every 20 minutes thereafter during trawling (when the net is dragged over the seabed). Birds were identified to species and, where possible, subdivided into adults and immatures.
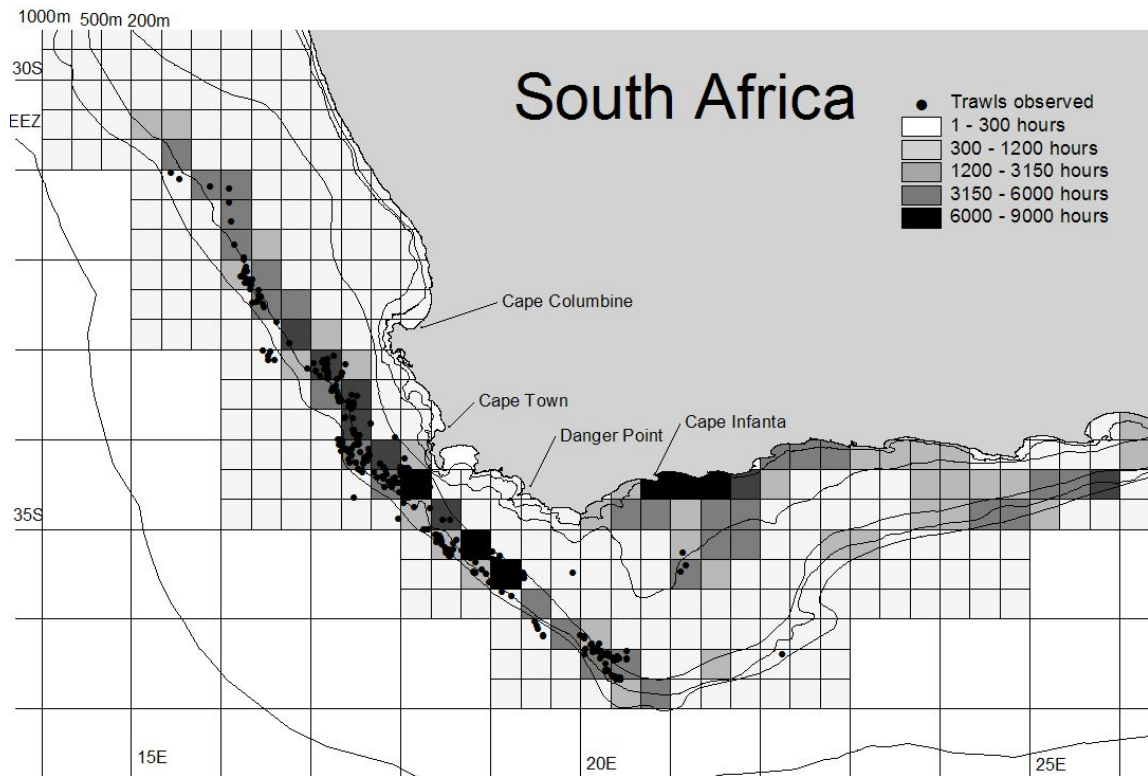

Figure 1: The distribution of commercial trawling effort in the South African hake fishery (shaded by quarter degree blocks) with study trawls indicated by points.

Dedicated warp observations were conducted in a minimum of 5-minute observation periods during trawling. These recorded all seabird interactions with the warp, including numbers of birds colliding with the warp, and being dragged underwater. Collisions were divided in to 'light' and 'heavy', following Sullivan *et al*. (2006). Heavy collisions resulted in significant deviation in a bird's flight path. Birds were assumed to have drowned if they were dragged under and did not surface within approximately 30 seconds. In some cases, there were too many birds to be sure whether a bird re-surfaced; these incidents were scored as possible mortalities. During the first half of the study, a video camera was mounted on the stern of the vessel over the warps to record interactions. Video footage was analysed after returning to port to confirm whether birds dragged under had not re-surfaced, as well as to check for interactions overlooked at sea. Use of the video camera ceased when the device malfunctioned and could not be repaired. However, there was no significant difference in the number of incidents reported with and without the aid of the video footage (controlling for

season and vessel activity). Observations took place above the warp where most seabird activity occurred, typically on the side where the majority of offal discharge was taking place.

In addition to seabird observations, the following data were collected: co-ordinates at the beginning and end of each trawl, water depth, wind strength (Beaufort scale) and relative wind direction. The extent of fishery discards was recorded from high (5) to low (1) and discharge was scored as either constant or intermittent. The relative proportion made up by whole fish (by species), heads and guts was scored. Other fishing parameters recorded included: duration of set (when the net entered the water, when the trawl doors (otter boards that ensure the net stays open) entered the water, and when the winches stopped paying out and the gear was on the sea floor); trawl duration (from end of set to start of haul); hauling duration (from start of haul to when the net is on deck) and the length of the warp pay-out. On each vessel the height of the pulley supporting the warp above the sea surface was measured when docked, because this determines the distance behind the vessel where the warps enter the water. We also measured the distance between the warps at the stern, the distance between the vessel side and the pulleys, and the presence/absence of warp splices.

Estimating seabird mortality

Bird mortality and the total numbers of warp interactions (birds touching or colliding with warps, or being dragged by the warp) were expressed as average rates per hour of observation. Interactions tended to occur sporadically, typically when large numbers of birds competed for access to discards. They were thus not normally distributed, and bootstrapping was used to assess confidence intervals around the mean interaction rates. This approach makes no assumptions about the statistical distribution of samples. For each set of data, we produced 1,000 random re-samplings (with replacement) of the original data set, calculated a mean interaction rate, and then sorted the results to obtain the 95% confidence interval of the mean.

Observations were summarised into winter (April-September) and summer (October-March), given seasonal differences in the abundance and species composition of birds attending vessels. It was evident that most interactions occurred during dumping, when numbers of birds close to the vessel peaked, and thus data also were divided into dumping and non-dumping periods. To assess the effect of other parameters such as wind speed and direction on interactions, GLMs were fitted to the data in Genstat, using Akaike's Information Criterion for model selection. The logarithm of the Poisson parameter λ was modelled as a linear combination of explanatory variables: $\log \lambda = a + b_1 x_1 + b_2 x_2 + b_3 x_3 + ...$ Explanatory variables included dumping, season, fishing activity (setting, trawling or hauling), wind strength (Beaufort scale) and relative wind direction (angle in radians from the bow).

A simple model was developed to estimate seabird mortality across the fishery. Total fishing effort was estimated to be 60 000 trawls per year, based on 79 vessels fishing for 70% of the year and making on average three trawls per day (B. Rose pers. comm.). Most trawls are

conducted during daylight due to the vertical migration of hake. The first tow of the day thus seldom results in seabird interactions with the warp because no offal is discharged; these tows were excluded from warp interactions. Also excluded were four vessels with macerators (devices to finely chop discards, making them less attractive especially to large birds) as well as vessels that retain wastes to make fish meal, either on board (four freezer vessels) or returning it to land (eight wet fish boats on the last three days of their trips). This resulted in a total of ca 35 000 trawls with dumping taking place. Due to longer days in summer, effort was weighted 55% to summer trawls. Dumping was assumed to average 1 hour per trawl, with total trawl duration averaging 3 hours. These estimates are conservative; utilization of vessels averages above 70%, some dumping takes place from vessels with fish meal plants and macerators, and dumping lasted 1.2 hours on average during our study trips. The effort data were used to calculate the total amount of time spent trawling in winter and summer both while dumping and not dumping, and this was used to extrapolate total mortality from the observed estimates of warp mortality. Mortality of birds entangled in nets can happen on any trawl, because it results from birds attempting to steal fish from the net. It is independent of dumping activity, and thus all trawl effort was included in the extrapolation of this mortality.

RESULTS

Observations were made on 331 trawls during eight summer (n=162 trawls) and 12 winter trips (n=169 trawls) on 14 different vessels. Two additional trips were made but were excluded from the analyses: only trial data were gathered during the first voyage, and one trip in winter 2005 was excluded because fishing took place inshore. Observers covered only 0.5% of annual fishing effort, but this was as much as could be achieved given the need for dedicated, well-trained seabird observers.

Seabird attendance at trawlers

Eighteen seabird species or species groups were observed attending trawlers regularly (Table 1). During fishing operations, peak numbers of birds within 50 m of the stern averaged about 1000 in winter and 500 in summer (Table 1). Among species at risk, numbers of black-browed albatrosses *Thalassarche melanphrys* and Cape gannets *Morus capensis* were much higher in winter than in summer (Table 1). There was surprisingly little difference in the total numbers of birds during dumping and non-dumping operations, but there were marked increases in some species, notably great shearwaters *Puffinus gravis* in summer, adult black-browed albatrosses, giant petrels *Macronectes* spp. and sooty shearwaters *Puffinus griseus* in winter, and Cape gannets throughout the year (Table 1).

Table 1: Average maximum numbers of seabirds attending trawling operations in the 50 m observation zone in summer and winter, during dumping and non-dumping (excluding species where <10 individuals recorded)

Species	Summer		Winter	
	Dump	**No dump**	**Dump**	**No dump**
Subantarctic skua *Catharacta antarctica*	3.3	6.5	13.0	17.6
Kelp gull *Larus dominicanus*	56.7	56.9	6.2	12.3
adults	54.5	54.6	6.1	12.2
immatures	2.2	2.3	0.1	0.1
Sabine's gull *Larus sabini*	4.3	4.6	0.0	0.02
Terns *Sterna* spp.	1.6	4.2	0.0	0.1
Cape gannet *Morus capensis*	23.7	14.3	173.2	118.7
adults	22.8	13.8	153.9	112.3
immatures	0.8	0.5	19.3	6.4
Wilson's storm-petrel *Oceanites oceanicus*	16.2	52.7	21.7	52.2
European storm petrel *Hydrobates pelagicus*	0.3	2.3	0.01	0.1
Shy Albatross *Thalassarche cauta*	57.1	78.5	58.6	77.7
adults	9.0	9.2	29.9	15.7
immatures	30.6	69.3	13.9	13.6
Black-browed albatross *T. melanophrys*	16.6	16.7	128.6	86.9
adults	5.7	7.2	83.9	41.7
immatures	4.5	7.1	20.6	10.6
Atlantic yellow-nosed albatross *T. chlororhynchos*	11.6	6.4	0.8	0.8
Indian yellow-nosed albatross *T. carteri*	11.3	10.0	10.8	6.8
Yellow-nosed albatross spp.	1.0	1.4	0.8	0.9
Unidentified mollymawks *Thalassarche* spp.	1.7	2.1	1.6	3.3
Southern giant petrel *Macronectes giganteus*	0.4	0.9	0.8	0.6
Northern giant petrel *M. halli*	8.6	2.6	1.0	0.5
Giant petrels *Macronectes* spp.	0.1	0.1	0.5	0.3
Cape petrel *Daption capense*	13.2	11.5	330.6	259.7
Antarctic prion *Pachyptila desolata*	0.0	0.0	0.3	0.6
White-chinned petrel *Procellaria aequinoctialis*	164.8	203.7	164.1	247.3
Great shearwater *Puffinus gravis*	52.9	47.1	13.6	17.8
Sooty shearwater *P. griseus*	3.5	3.2	90.5	38.8
Total	**449.1**	**526.3**	**1016.7**	**943.4**

Seabird interactions with trawl nets

Ten birds were killed after becoming entangled in trawl nets: nine Cape gannets and one subantarctic skua *Catharacta antarctica*. These birds were drowned either during setting, when they were enmeshed in the trawl wings, or during hauling, when diving or swimming into the net entrance or trapped while standing on the net as the mesh diameter closes. The problem is greater during setting if the nets are not cleaned and during hauling if the net has split during trawling operations. Net mortalities occurred at an average rate of 3.02 (95% CI

0.91-5.44) birds per 100 trawls. All birds killed in nets were hauled aboard; it is not known whether additional birds were drowned but not hauled aboard.

Seabird interactions with trawl warps

Almost 2 500 collisions with trawl warps were recorded during 189.8 hours of dedicated warp observations (Table 2). Most were light or heavy aerial collisions, that usually had little apparent impact on birds. Only one white-chinned petrel suffered a broken wing as a result of an aerial collision. More serious were the birds dragged underwater when their wings became entangled and they were pulled under by the force of the water passing over the warp. Many birds dragged under re-surfaced, but 30 drowned, mainly shy *Thalassarche cauta* (*sensu lato*) and black-browed albatrosses (Table 2). At least 26 other birds (16 albatrosses, 7 Cape gannets and 3 white-chinned petrels *Procellaria aequinoctialis*) were dragged under and not seen to re-surface, but conditions were too crowded to confirm that they drowned. Birds were dragged under during setting, trawling and hauling operations. Most birds were dragged when they were struck by the warp as they were landing or were scavenging on the water with raised wings. Loose ends of splices in trawl warps may have exacerbated the problem, and certainly assisted birds to remain on the warp throughout the trawl and thus be hauled aboard, but they were not necessary for birds to become entangled. Of the 30 birds observed killed on the warps, only 2 albatrosses were hauled aboard. The bodies of at least three other albatrosses killed outside observation periods also were hauled aboard on the warps during the 20 study trips.

Albatrosses were killed most frequently, because their long wings were most easily tangled around the warp. They suffered a disproportionately high mortality rate, with at least 15% of albatrosses that were dragged under drowning, compared to 4% of all other species (Table 2, $\chi^2=8.45$, $P<0.005$). They also were more prone to be dragged underwater, with 16% of warp interactions resulting in albatrosses being dragged under, compared to 8% of all other species (Table 3, $\chi^2=26.01$, $P<0.001$). Other species killed included white-chinned petrels (10% of birds killed), Cape gannets (7%) and sooty shearwaters (3%). Small petrels frequently collided with the warp, but were seldom dragged under or drowned (Table 2). Proportions of serious warp interactions did not differ between adult and immature albatrosses, but there was a tendency for immature gannets to be dragged under more often (20% of all interactions) than adults (12%, $\chi^2=2.80$, $P<0.1$).

Virtually all warp mortality occurred during dumping operations (Table 3), and the rate of mortality was greater in winter (Table 3), when more birds attended fishing vessels. The average mortality rates were 0.56 (95% confidence interval 0.32-0.82) birds killed per hour during dumping in winter, 0.21 (0.07-0.38) during dumping in summer, 0.09 (0.02-0.19) when not dumping in winter, and 0.00 (–) when not dumping in summer (Fig. 2). These are minimum mortality estimates, because only one warp was observed (the one closest to the discard stream and thus with the largest number of birds) and they exclude incidents of possible mortality (birds dragged under, outcome unknown). Rates of all warp interactions

were 51.6 (40.2-64.8) and 18.6 (14.1-23.8) birds per hour during dumping in winter and summer, respectively, and 3.9 (2.5-5.5) and 0.7 (0.4-1.1) birds per hour when not dumping in winter and summer (Fig. 2).
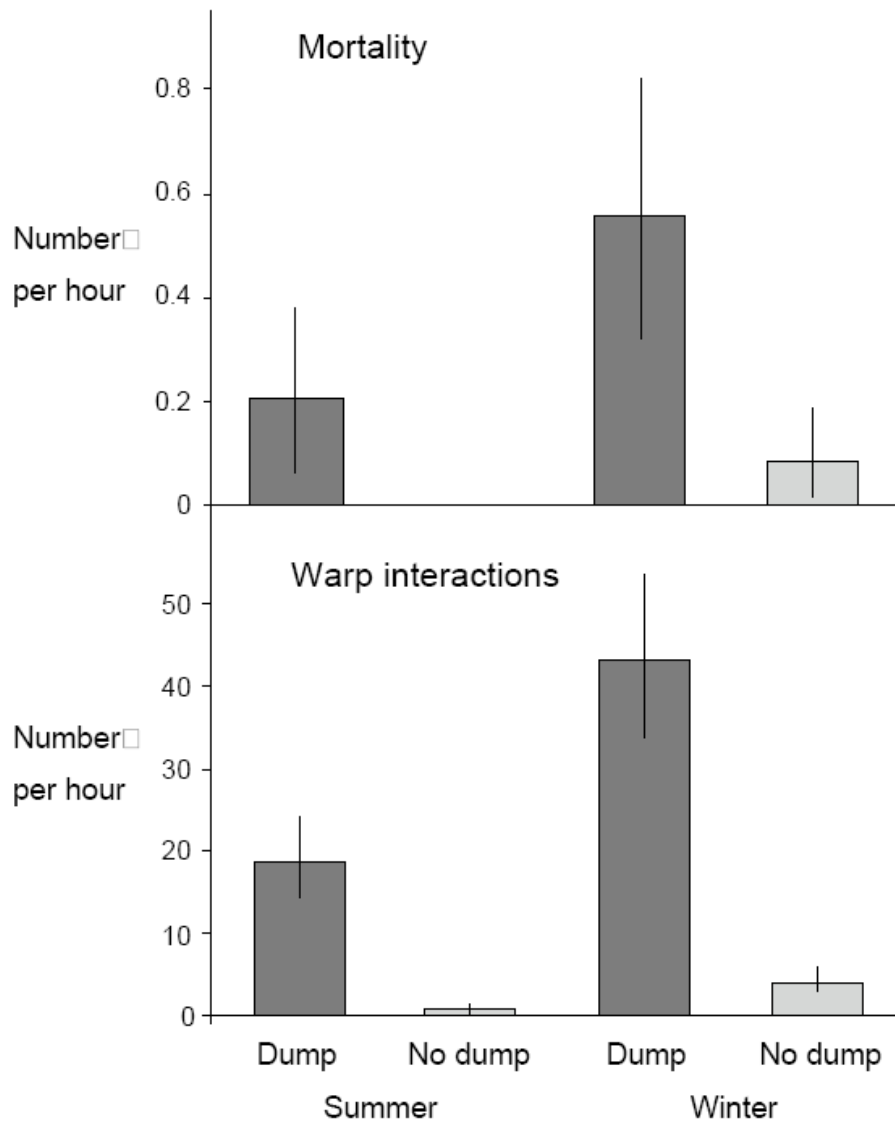

Figure 2: Average rates of warp mortalities and all warp interactions by season in relation to dumping and non-dumping activity. Vertical bars = 95% confidence intervals.

Table 2: Total numbers of interactions between seabirds and trawl warps during 190 hours of dedicated observations

Species	Killed *	Dragged	Collisions		Total
			Heavy	Light	
Subantarctic skua		2	16	40	58
Kelp gull		0	3	10	13
Cape gannet	2	50	111	177	340
Wilson's storm petrel		0	0	1	1
Shy albatross	13	59	82	205	359
Black-browed albatross	11	63	88	269	431
Yellow-nosed albatrosses		10	15	36	61
Giant petrels		1	8	15	24
Cape petrel		24	147	299	470
White-chinned petrel	3	45	220	307	575
Great shearwater		4	28	63	95
Sooty shearwater	1	5	10	11	27
Total	**30**	**263**	**728**	**1433**	**2454**

includes broken wings from collisions, but excludes birds entangled in nets.

Table 3: Mortalities of seabirds during dedicated warp observations in relation to season and dumping. A further 26 birds may have drowned, but could not be confirmed

Species*	Summer		Winter		Total
	Dump	No dump	Dump	No dump	
	29.1 h	69.3 h	34.0 h	57.4 h	189.8 h
Cape gannet*	0	0	2	0	2
Shy albatross	5	0	6	2	13
Black-browed albatross	0	0	9	2	11
White-chinned petrel	1	0	2	0	3
Sooty shearwater	0	0	0	1	1
Total	**6**	**0**	**19**	**5**	**30**

excludes birds killed after being entangled in the net (9 Cape gannets, 1 Subantarctic skua).

Including a suite of potential explanatory variables, the best predictor of seabird-warp interactions was a model including season, dumping, relative wind direction and wind strength (Table 4). Interactions were more frequent when dumping was taking place in winter, and when there was a strong wind from the stern. Most bird interactions occurred at wind strengths greater than 5 on the Beaufort scale. Strong winds usually are associated with larger swell conditions, pitching the vessel and causing the warps to rise and fall erratically. More birds were drowned during winter and when dumping was taking place (Table 4).

Table 4: Results of the GLM model showing that warp interactions were more likely to occur during dumping, in winter, when the wind is from the stern, and the wind is strong. The model explains 15.6% of total variance

Parameter *	Estimate	s.e.	t	p
Constant	-1.52	0.05	-28.1	<0.001
Wind direction (cosine in radians)	-0.33	0.03	-9.62	<0.001
Dump (yes)	1.11	0.06	19.04	<0.001
Season (winter)	0.29	0.06	4.73	<0.001
Wind strength (Beaufort >5)	0.19	0.08	2.50	0.012

Reference value or factor level in parenthesis

Estimated impact of the fishery

Extrapolating the observed mortality rates across the deep-water trawl fleet suggests a total of some 18000 birds killed each year, with a 95% confidence interval of 8 000 to 31 000 (Table 5). This estimate is most sensitive to changes in bycatch rates; changes in fishery effort parameters had little impact on the projected total mortality. Overall, 85% of projected mortalities result from warp collisions, with 15% entangled in nets. Of the birds killed, 41% are shy albatrosses, 39% black-browed albatrosses, 14% Cape gannets and 9% White-chinned petrels. These are minimum estimates, because conservative effort parameters were used and only confirmed kills were included in the estimation of mortality rates.

Table 5: Estimated annual impacts of trawl warp interactions and net entanglements across the South African deep-water hake trawling fleet. Numbers are rounded averages of bird mortality with bootstrapped 95% confidence intervals

Species	Warps	Nets	Total
Shy albatross	7 000 (3 000-12 500)	0	7 000 (3 000-12 500)
Black-browed albatross	5 000 (2 500-8 500)	0	5 000 (2 500-8 500)
White-chinned petrel	1 500 (800-2 500)	0	1 500 (800-2 500)
Sooty shearwater	500 (100-1 200)	0	500 (100-1 200)
Cape gannet	900 (500-1 400)	1 600 (500-3 000)	2 500 (1000-4 400)
Subantarctic skua	0	180 (0-500)	180 (0-500)
Total	**15 500 (7 000-26 000)**	**1 800 (500-3 200)**	**18 000 (8 000- 31 000)**

DISCUSSION

The productive shelf waters off southern Africa are globally important foraging areas for a wide diversity of seabirds, including several threatened species (Sinclair, 1978; Ryan and Moloney, 1988). Discards from the deep-water hake fishery make up more than half the diet of white-chinned petrels (Jackson, 1988; Crawford, Ryan & Williams, 1991), and are important for black-browed and shy albatrosses (Crawford *et al.*, 1991) as well as Cape gannets (Berruti et al., 1993). As a result, large numbers of birds gather at hake trawlers. Warp interactions appear to have been largely overlooked in the past because few dead birds are hauled, usually only when they are caught on a cable splice. It requires dedicated observers to assess the frequency of serious interactions.

Our study indicates that there is a significant conservation issue that requires urgent attention. Although confidence intervals are broad due to the low observer coverage, the extrapolated estimate of bird mortality is quite alarming. Even if mortality lies at the lower end of the confidence interval, it suggests that at least 5 000 albatrosses are killed each year, which is unsustainably high given their conservative life histories that cannot support even moderate increases in adult mortality. The best estimate is more than 12 000 albatrosses killed each year, which is similar to the estimated impact of the pelagic long-line fishery off South Africa (Ryan, Keith & Kroese, 2002). The four species killed most frequently are all listed as globally threatened or near-threatened (BirdLife International 2004). The shy albatross is currently listed as near-threatened, but this may be upgraded to Vulnerable given recent estimates of mortality in pelagic long-line fisheries off South Africa, as well as the data from this study. The black-browed albatross is listed as Endangered due to recent rapid population decreases at its main breeding sites. Cape gannets and white-chinned petrels are both listed as Vulnerable, also as a result of population decreases.

The mortality rates in the South African trawl fishery for hake are higher than those reported in other Southern Ocean shelf fisheries. Off New Zealand, mortality rates in the hoki *Macrourus novaezelandiae* and squid trawl fisheries vary regionally from 0.007 (SE=0.008) birds per tow in Cook Strait to 0.198 (SE=0.032) on the Stewart Island-Snares Island Shelf (Baird 2004). Comparable figures for the South African fishery are *ca* 0.30 birds per tow, averaged across the fleet. The average mortality recorded off the Falkland Islands is 0.47 birds/vessel/day (Sullivan and Reid 2003), less than the *ca* 0.90 birds per day for the South African fishery. However, the Falklands data are based only on recovered carcasses, and are acknowledged to underestimate actual mortality (Munro, 2005). The really worrying aspect is that the large size of the South African fishery results in much larger extrapolated mortality than the other fisheries. The estimated 18 000 birds killed in the South African hake fishery is an order of magnitude greater than the estimated 1 400 black-browed albatrosses killed in the Falklands fishery (Sullivan and Reid, 2003) and 1 100 seabirds in the hoki and squid fisheries off New Zealand (Baird, 2004).

Black-browed albatrosses were the predominant species killed during the 2002-2003 season in the Falklands (Sullivan *et al.*, 2006) and this species had significantly more contacts when discharge was taking place. Mortality rates differed by area within Falkland waters with the greatest number taking place in the south-west during August-September. Greater rates of contacts occurred with increasing sea or swell heights and more contacts took place during tail winds than head winds. Gonzalez-Zevallos, Yorio & Caille (in press) reported that in a study off Argentina an estimated 2703 birds, predominantly kelp gulls and black-browed albatrosses, were killed in the hake trawl fishery (cable mortality rate 0.14 birds/haul). A plastic traffic cone attached to the warps reduced contacts by 89% and no seabirds were killed.

Of the abundant birds attending deep-water hake trawlers off South Africa, only Cape gannets and kelp gulls breed in southern Africa. Most species have traveled from distant breeding areas in the sub-Antarctic, Australasia or the North Atlantic (Hockey, Dean & Ryan, 2005), with some, such as the white-chinned petrel, even commuting to forage while breeding on sub-Antarctic islands more than 2,000 km away (Weimerskirch *et al.*, 1999). South Africa has an international obligation to conserve these species in terms of the Agreement on the Conservation of Albatrosses and Petrels (ACAP), an agreement under the Bonn Convention on Migratory Species, which requires *inter alia* that steps be taken to reduce unnecessary mortality of seabirds due to fishing activities. South African national legislation also requires that fishing activities do not cause significant impacts on other components of marine ecosystems (Marine Living Resources Act of 1998, objective 2e).

There is an urgent need for mitigation to reduce seabird mortality in the South African trawl fishery for hake. Managing fishery discards is one way to reduce seabird mortalities (Munro, 2005; this study). Equipping vessels with macerators or offal holding tanks to avoid dumping while trawl gear is in the water should greatly reduce warp interactions. Another option is to use discards to make fish meal aboard vessels. The other way to reduce mortality is to keep birds away from the warps during fishing operations. Bird-scaring lines have proved effective in keeping birds away from baited hooks during setting of long-lines, and their use has been a permit condition for some time in long-line fisheries operating in South African waters. Trials at the Falkland Islands suggest they are equally effective at reducing warp interactions behind trawlers (Munro 2005). Initial trials off South Africa suggest that a pair of short bird-scaring lines set over the warps greatly reduce the numbers of birds entering the danger zone where the warps enter the water. Given these promising results, use of bird-scaring lines was required as a permit condition in the South African deep-water trawl fishery from August 1 2006. However, further studies are needed to assess the most efficient design for bird-scaring lines or other mitigation devices, in terms of both efficacy at reducing seabird mortality and ease and safety of use by the industry. There is also a pressing need to conduct mitigation trials in winter, when bird numbers and warp interactions peak.

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MITIGATING BYCATCH

Chapter 7

ALBATROSS OVERLAP WITH FISHERIES IN THE BENGUELA UPWELLING SYSTEM: IMPLICATIONS FOR CONSERVATION AND MANAGEMENT

ALBATROSS OVERLAP WITH FISHERIES IN THE BENGUELA UPWELLING SYSTEM: IMPLICATIONS FOR CONSERVATION AND MANAGEMENT

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ABSTRACT

Comparatively few studies have examined the relative importance of fisheries waste compared with natural prey in the diet of pelagic seabirds. Trawl activity on the continental shelf break off South Africa provides large quantities of high quality and predictable food in the form of offal and discards for a range of species, including non-breeding Black-browed Albatrosses *Thalassarche melanophrys* and White-capped Albatrosses *T. cauta.* As large numbers of both species are killed in collisions with trawl warp cables, mitigation measures have been introduced that include limitation of discard levels, yet little is known about the consequences of reduced food supply for scavenging birds. In this study, adult and immature albatrosses were tracked in the southern Benguela in the austral winter of 2005 and 2006, and their distribution examined in relation to fisheries, bathymetry and remotely-sensed oceanography. Kernel analysis revealed striking differences in distribution. Whilst in the south east Atlantic, White-capped Albatrosses spent most (85.0%) of their time on the Southern African trawl grounds, whereas Black-browed Albatrosses spent only 39.2% of their time in these areas, and the remainder either in deeper waters or on return oceanic foraging trips typically of 8.4 days or 2 540 km (max. 5 320 km) in duration. While foraging in South African waters, the presence of trawlers, but not longliners, was a strong predictor of both White-capped and Black-browed Albatross distribution. The limited proportion of time spent overall on the trawl grounds by Black-browed Albatrosses suggests that they are either out-competed by the larger White-capped Albatross behind vessels, or are better adapted for oceanic foraging. While on oceanic foraging trips, Black-browed Albatrosses moved predictably along the margins of warm and cold eddies, typically areas of enhanced productivity and thus high prey concentrations. This study presents evidence that Black-browed Albatrosses, in particular, forage to a large extent on natural prey, despite the high availability of discards from fishing vessels in the Benguela. Therefore, given the high incidence of albatross collisions with trawl cables, the benefit of a management decision to limit discarding as a mitigation measure is likely to far outweigh the disadvantage of reduced food in the form of fisheries waste.

INTRODUCTION

Albatrosses are K-selected, displaying high adult survival, delayed onset of first breeding and low reproductive rates (Warham 1996). Life history characteristics reflect evolutionary responses to environmental conditions (Lack 1968) and have, in the case of albatrosses, been attributed to the sparse, patchy and unpredictable nature of the marine resources upon which they depend. In the non-breeding season, albatrosses are not restricted to waters within commuting distance of colonies, and often migrate long distances across ocean basins to feed in areas where food is presumably more predictable (Croxall *et al.* 2005, Phillips et al. 2005a). Bathymetric features such as continental shelves or sea-mounts, and oceanic frontal systems, are areas of enhanced productivity that are often targeted by foraging seabirds (Hunt & Schneider 1987, Hunt 1991, Weimerskirch *et al*. 1993, Pakhomov & McQuaid 1996, Brothers *et al*. 1998, Prince *et al*. 1998, Waugh *et al*. 1999, Nel *et al.* 2001, Waugh & Weimerskirch 2003, Phillips *et al*. 2005a).

Areas of high productivity are also commonly areas where commercial fisheries have developed. It is well documented that fisheries, particularly trawl, represent a substantial food source for opportunistic seabirds through the discarding of waste offal and non-target catch (Abrams 1983, 1985, Furness *et al.* 1988, Ryan & Moloney 1988, Garthe & Hüppop 1994, Garthe *et al.* 1996, Wanless 1998, Phillips *et al.* 1999, González-Zevallos & Yorio 2006, Sullivan 2006). Indeed, the presence of fisheries has been identified as an important determinant of seabird distribution at sea and changes in fisheries practices can have major implications for seabird community structure (Wahl & Heinemann 1979, Ryan & Moloney 1988, Furness *et al.* 1992, Garthe 1997, Votier *et al*. 2004). Moreover, as seabirds have limited diving ability (Prince *et al.* 1994), demersal trawl fisheries clearly permit access to deep water resources, such as hake *Merluccius* spp, that would otherwise be unavailable. Nevertheless, several studies of seabirds generally considered to rely heavily on discarding have found that natural prey are much more important across part or all of the range, or during certain times of year (Phillips *et al*. 1999, Cherel *et al*. 2002). In addition, feeding on fisheries discards from trawlers or on baited longline hooks is associated with risk of injury or mortality and is the leading cause of observed population declines in many albatrosses and petrels (Croxall & Gales 1998, Sullivan 2006, BirdLife International 2007).

The upwelling system off the west coast of South Africa, known as the southern Benguela ecosystem, is one of the world's most productive marine ecosystems (Shannon & Field 1985). Large-scale commercial longline and trawl fisheries for hake, and longline fisheries for tunas *Thunnus* spp provide large amounts of fisheries discards, and as a consequence have led to long-term changes in distribution of seabirds in the region (Abrams 1983, Duffy *et al.* 1987, Ryan & Moloney 1988, Crawford *et al.* 1991). The South African Exclusive Economic Zone (EEZ) is a key foraging area for over-wintering and non-breeding birds of both these species (Brothers *et al.* 1998, Prince *et al.* 1998, Phillips *et al.* 2005a), which are of global conservation concern (Croxall & Gales 1998, BirdLife International 2007). The Black-browed

Albatrosses that over-winter in the Benguela region breed at South Georgia in the south-west Atlantic Ocean, and at Crozet and Kerguelen in the Southern Indian Ocean, where populations have decreased dramatically since the mid 1970s (Croxall *et al.* 1998, Weimerskirch & Jouventin 1998, Poncet *et al*. 2006). Although there are currently few reliable trend data for White-capped Albatrosses at breeding colonies, both longline and trawl fisheries operating in the southern Benguela are clearly having severe impacts on both species (Barnes *et al*. 1997, Ryan *et al.* 2002, Petersen *et al.* 2007, Chapter 1, 4 and 6).

Given the increasing concern world-wide about impacts on non-target species, many fisheries, including those operating in the Benguela ecosystem, have introduced mitigation protocols that aim to reduce incidental mortality of seabirds, turtles and sharks (Brothers *et al*. 1999, CCAMLR 2002, MCM 2007, Read 2007). These include improved discard management to reduce the attraction of seabirds to vessels, often in tandem with other measures. This, however, has potentially important resource implications, depending on the extent to which the affected seabirds rely, or can revert to feeding on natural prey. This study investigated the foraging strategies of non-breeding White-capped and Black-browed Albatrosses in relation to fisheries, bathymetry and oceanographic features. Results are discussed in the context of likely responses of these species to a reduction in discard and offal availability in the region.

METHODS

Birds

Satellite transmitters or Platform Transmitter Terminals (PTTs) were deployed on eight Black-browed (three adults and five immatures) and five White-capped Albatrosses (all immatures) in South African waters. In 2005, eight PTTs were deployed between 13 July and 20 August on four immature Black-browed Albatrosses and four immature White-capped Albatrosses (Table 1). In 2006, PTTs were deployed between 24 June and 16 September on one immature and three adult Black-browed Albatrosses, and one immature White-capped Albatross (Table 1). Based on molecular separation, shy-type albatrosses foraging in southern African waters are most likely to be White-capped Albatrosses, *Thalassarche steadi* that breed in New Zealand, with Shy Albatrosses *T. cauta* from Australia representing only *c.* 5% of birds returned to port (Abbott *et al*. 2006, Baker *et al*. 2007). Therefore for simplicity, we refer to all shy-type albatrosses tracked in this study as White-capped Albatrosses.

Deployed devices were Microwave telemetry PTTs (30–50 g) or Kiwisat 202 PTTs, Sirtrack (32 g). The former were set to transmit every 90 s throughout the day, whereas the latter were duty cycled 12 hours on and 12 hours off. On average, 14 positions per day (12–21) were obtained using Microwave PTTs, and 7 positions per day (5–11) using Kiwisat PTTs. Birds were caught at sea approximately 40 km southwest of Cape Point, South Africa, using a barbless hook, and lifted into the vessel by means of a net. PTTs were attached to the feathers on the bird's mid-back (mantle) using waterproof Tesa tape. Data collection took

place from July to December in 2005 and June to November in 2006. Locations were obtained using the ARGOS (Advanced Research Global Observation Satellite) system. Data received from ARGOS were filtered according to flight speed (maximum velocity was set at 100 km.h^{-1}) and ARGOS location quality (all positions with a quality code of A, B and Z were excluded). Spatial analysis (including production of kernels and mapping of density distributions) was performed using Arcview 3.2. This involved the calculation of fixed kernel home range utilization distributions (based on Worton 1989) as grid coverages, with the smoothing factor chosen using least squares cross validation (Silverman 1986). Kernel density plots have been used successfully in numerous tracking studies to quantify habitat use (e.g. Wood *et al.* 2000, BirdLife International 2004, Nicholls *et al.* 2005). The Utilization Distributions (UD) provide probability contours indicating the relative proportion of the distribution within a particular area.

Other data sources

Seabird tracks were overlaid on fisheries, bathymetric and sea height anomaly data. The position and date of trawls for hake, and setting of surface longlines for tuna *Thunnus* spp. and Swordfish *Xiphias gladius*, that took place over the same time period as the birds were tracked, were obtained from vessel logbooks and catch returns. Etopo-2 bathymetry data were obtained from the National Geophysical Data Center (NGDC). Etopo-2 is a worldwide set of 2-minute gridded ocean bathymetry derived from 1978 satellite radar altimetry of the sea surface (Sandwell 1990). Sea height anomaly (SHA) data derived from JASON-1, TOPEX, ERS-2, ENVISAT and GFO altimeters and processed at the Stennis Space Center, were obtained from NOAA (www.aoml.noaa.gov). Maps were generated by interpolating SHA data for an average of a 10-day period corresponding to the overlaid bird track (on average 8.4 days in length, range 2–20, SD 6.9), using Ocean Data View software for visual representation and Arcview 3.2 for analysis. This information was used to investigate whether birds foraging in the open ocean were likely to be targeting positive or negative anomalies, by comparing the gradient in SHA (the first derivative) at PTT locations with a randomly generated set of points. A two-tailed *t*-test for unmatched pairs was used to investigate whether the mean SHA for each location along the bird's track was significantly higher than that of the set of random locations.

Relationships between the distribution of Black-browed and White-capped Albatrosses, fisheries and bathymetry were investigated using Spearman rank correlations and generalised linear models (GLMs) with a Poisson distribution and logarithmic link function (McCullagh & Nelder 1989). Genstat 9 (Genstat Committee 2007) was used for model fitting and the Akaike's Information Criterion was used to guide model selection (Quinn & Keough 2002). The logarithm of the parameter λ of a Poisson distribution was modeled as a linear combination of explanatory variables: e.g. for three explanatory variables, $\log \lambda = a + b_1 x_1 + b_2 x_2 + b_3 x_3$. Variables were summarized by one degree grid square. Initially, the explanatory variables used were number of longline hooks, number of longline sets, number of trawls and

bathymetry. The use of longline sets resulted in better fits to the distribution data than the use of longline hooks, and therefore the latter was subsequently excluded from models.

RESULTS

In 2005, four immature Black-browed and four immature White-capped Albatrosses were tracked for a total of 284 (average per bird 71) and 409 (average 102) days respectively. In 2006, one immature Black-browed, three adult Black-browed and one immature White-capped Albatrosses were tracked for a total of 75 days, 156 (average 10–72) and 35 respectively (Table 1). Birds were tracked for a total distance of 210 400 km (62 000 km immature Black-browed, 94 000 km adult Black-browed and 54 600 km immature White-capped Albatrosses respectively).

Table 1: Summary of satellite-transmitter (PTT) deployments in 2005 and 2006.

	Black-browed (Immature)		Black-browed (Adult)		White-capped (Immature)	
	No of tracks	Average in days (min-max)	No of tracks	Average in days (min-max)	No of tracks	Average in days (min-max)
2005	4	71 (3-123)	0	-	4	96 (9-139)
2006	1	75	3	52 (10-72)	1	35

Whilst in the southeast Atlantic, there was no significant difference in the average distance travelled in 24 hours (p=0.1, 0.3 respectively), or any obvious difference in distribution between adult and immature Black-browed Albatrosses, and data were therefore pooled in further analyses. White-capped Albatrosses travelled less than half the distance per day (average of 127 km.d^{-1}, range 77–168 km.d^{-1}) of Black-browed Albatrosses (average of 303 km.d^{-1}, range 123–505 km.d^{-1}, t=3.65, p=0.006). White-capped Albatrosses also travelled more slowly on average (12.9 km.h^{-1}) than Black-browed Albatrosses (19.8 km.h^{-1}, t=3.82, p=0.003).

White-capped Albatrosses remained almost exclusively (98.9% of time tracked) within 200 nm of the coast, mostly (82.6%) within South Africa's Exclusive Economic Zone (EEZ) and the remainder within Namibian waters (Fig. 1). Time tracked was particularly concentrated on the continental shelf break between 200–1 000 m where they spent 79.1% of their time (55.3% 200–500 m isobath and 23.8% 500-1000 m isobath) (Fig. 2). Two of five tracked White-capped Albatrosses travelled as far north as 20 °S into Namibian waters where they spent 70 of 113 (62.0%) and 8 of 36 days (22.2%) tracked respectively. Two White-capped Albatrosses each made a brief trip outside of the EEZ. One bird travelled 413 km in two days and the other travelled 2 090 km in 10 days before returning to the continental shelf.
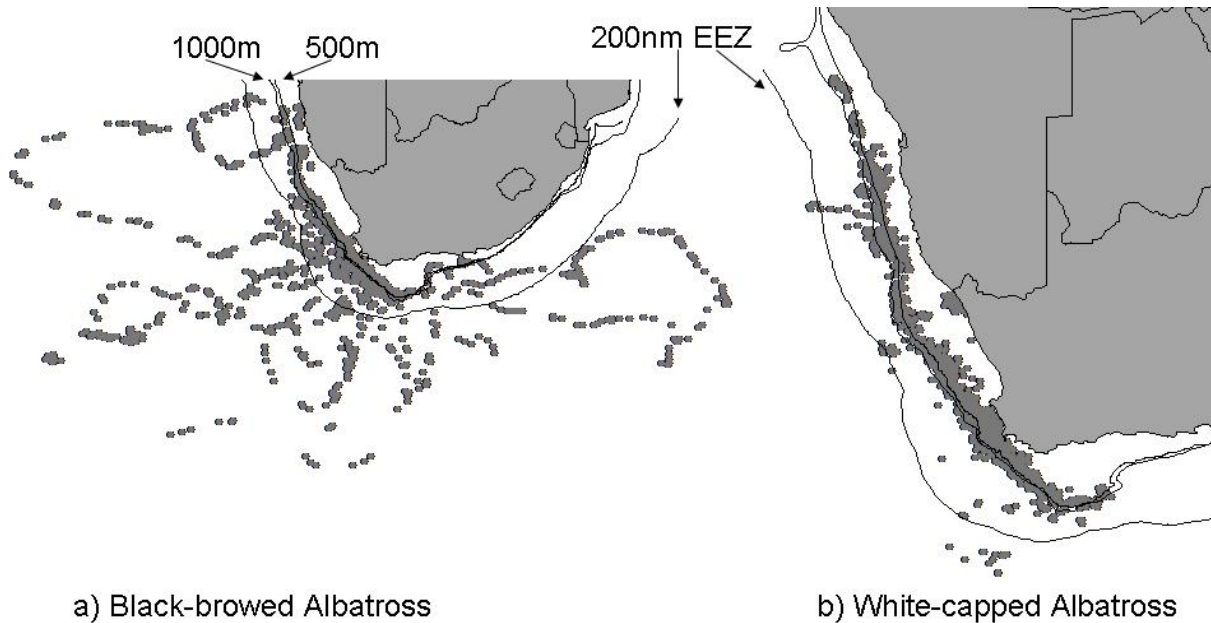
Figure 1: Filtered locations of Black-browed and White-capped Albatrosses tracked for 515 and 418 days respectively using PTTs during July to December in 2005 and June to November in 2006.

All Black-browed Albatrosses tracked for longer than 18 days in South African waters (n=4) made frequent trips into deep, oceanic waters (greater than 3000 m) where they spent 30.6% of their time before returning to forage on the continental shelf, mostly (34.9%) in water between 200–500 m depth (Fig.1 and 2). Oceanic trips lasted on average 8.4 days (range 2–20 days, SD 6.3) covering on average 2 540 km (range 540–5 320 km, SD 1 520) or 340 km.d^{-1} (range 100–575 km.d^{-1}). Between trips they tended to forage on the continental shelf for an average of 15 days (range 4–42 days). One of the immature birds travelled west across the Atlantic Ocean, leaving the southern Benguela on 23 October 2005 and arriving on the continental shelf off northern Argentina on 11 November 2005, having covered a minimum distance of 9 200 km over 19 days and at an average speed of 484 km.d^{-1}. Two of five tracked Black-browed Albatrosses travelled into Namibian waters as far north as 24 °S (20 of 75, 26.7% and 8 of 83, 9.6%) (Fig. 1). Of the three adults tracked, one stayed on the continental shelf for the 10 days for which the device transmitted data. The remaining two birds both foraged southwest of Cape Town (33–37 °S) for 11 and 18 days, then departed from the region on 27 September and 4 October 2006 respectively. Both birds travelled a similar path across the Atlantic Ocean to South Georgia and covered a distance of 7 286 km and 6 444 km in 21 days and 16 days respectively. Black-browed Albatrosses spent almost half (average 50.2%, range 16.2–92.2%, SD 30.8) of the time tracked within 200 nm of the coast compared to an average of 98.91% (range 97.2–100.0%, SD 1.2) in the case of the White-capped Albatross (U=0.0, p=0.004).
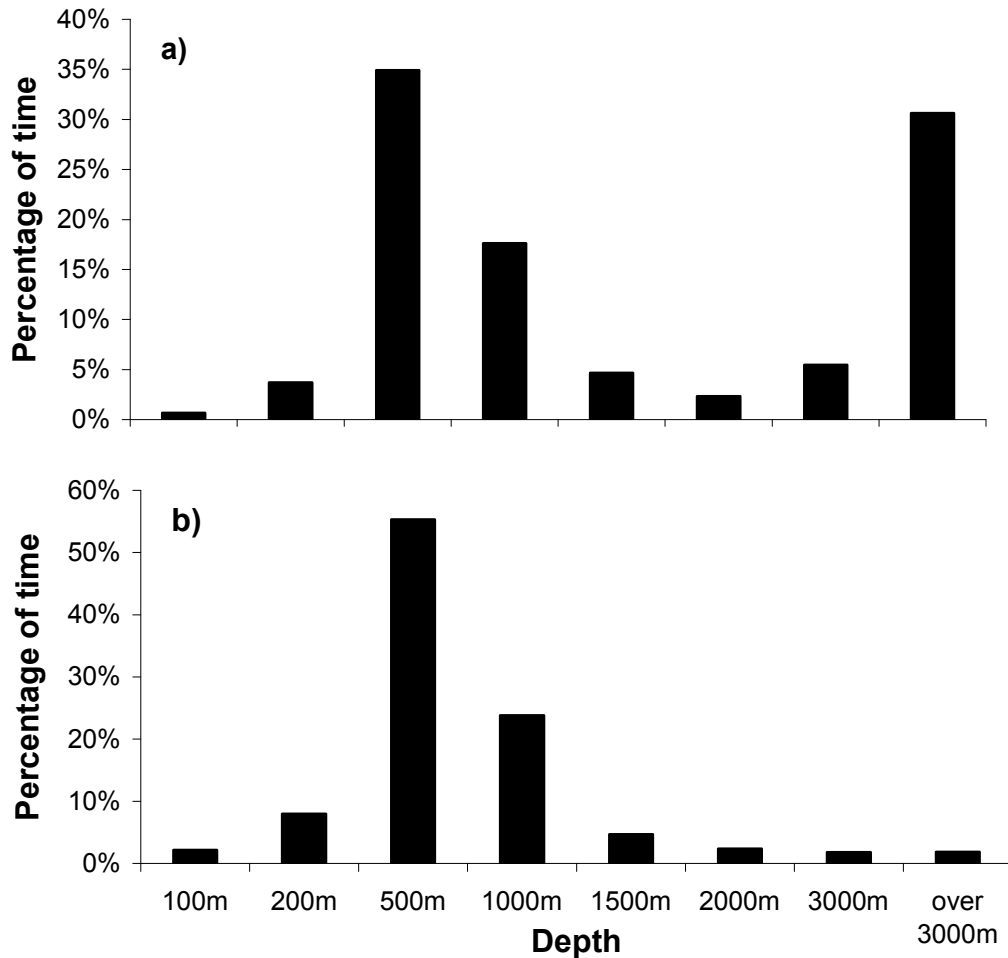
Figure 2: Percentage of time spent at varying water depths for a) Black-browed Albatrosses (excluding time spent commuting to southwest Atlantic) and b) White-capped Albatrosses.

The 95% utilization area of Black-browed Albatrosses in the south east Atlantic (i.e. excluding time spent commuting to southwest Atlantic) was 3.4 times larger than that of White-capped Albatrosses (Fig. 3). White-capped Albatrosses covered an area of 330 000 km^2 (95% utilization area) and a core area (50% utilization area) of 35 500 km^2 whereas Black-browed tracks covered a 95% utilization area of 515 000 km^2 and a core area of 120 000 km^2. Comparing the core area (50% utilization area) between individuals of each species revealed that White-capped Albatrosses were more constrained and on the shelf edge (average 24 400 km^2, 8 420–36 640 km^2, SD 12 280 km^2) compared to Black-browed Albatrosses (average 108 870 km^2, range 28 560–145 960 km^2, SD 54 060 km^2) (Fig. 4). Patterns were similar between individuals.

a) Trawl fishery
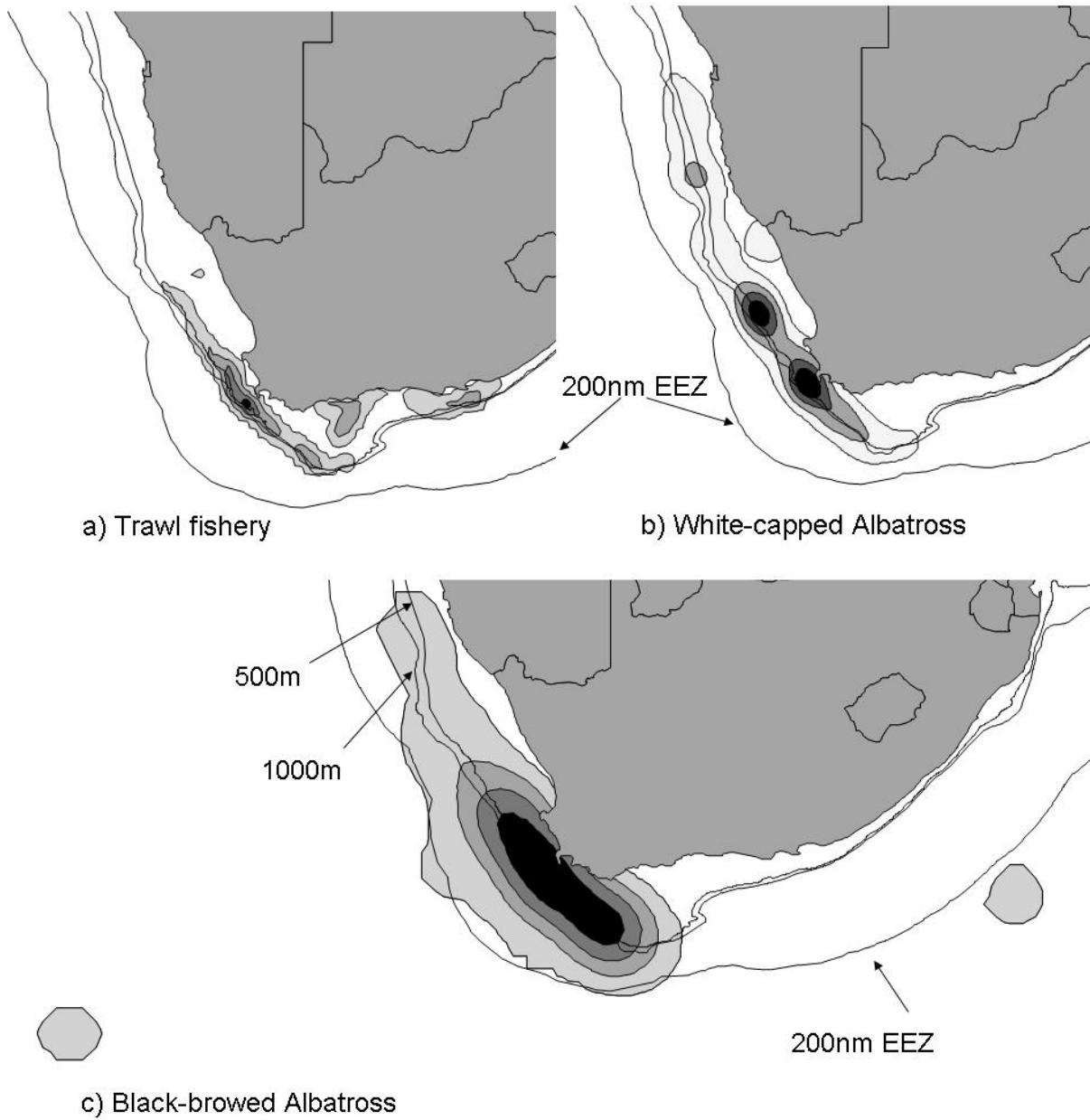
b) White-capped Albatross

c) Black-browed Albatross

Figure 3. Map showing the 25%, 50%, 75% and 95% utilization distribution of (a) the trawl fishery, (b) White-capped Albatrosses and c) Black-browed Albatrosses tracked using satellite-transmitters in the Benguela ecosystem in 2005 and 2006, excluding commuting locations of Black-browed Albatrosses returning to southwest Atlantic Ocean.
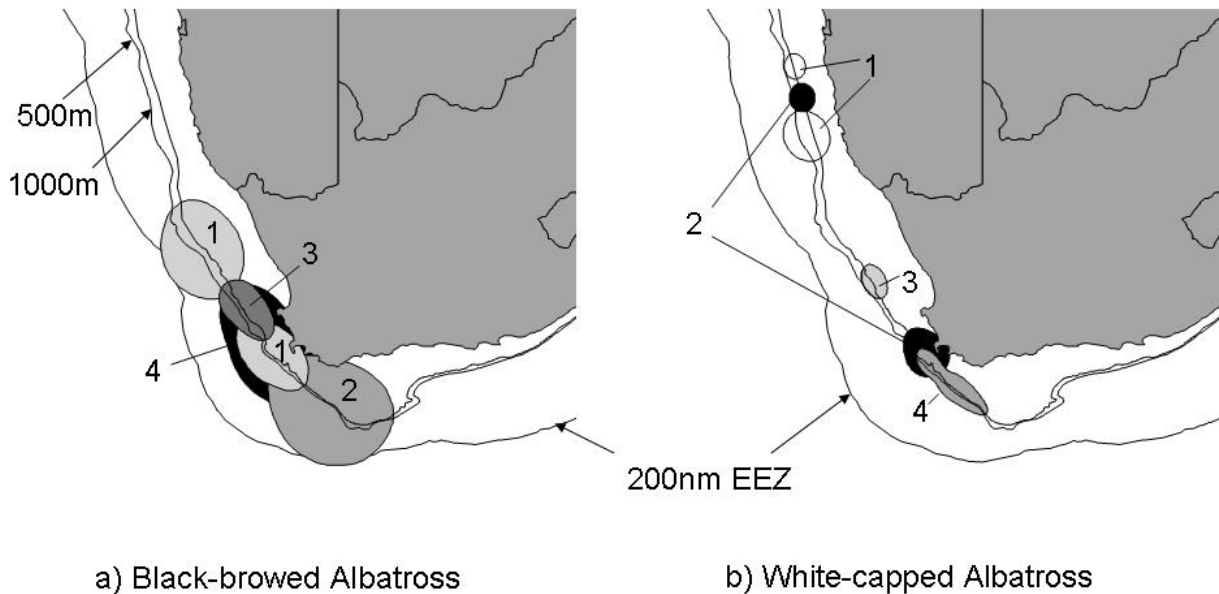
Figure 4: 50% utilization area for a) Black-browed (n=4) and b) White-capped Albatrosses (n=4) excluding time spent commuting and only including birds that foraged for more than 18 days in South African waters.

Overlap with South African fisheries

Excluding time spent commuting to south west Atlantic, Black-browed Albatrosses spent almost half the amount of time on the trawl grounds (average 39.2%, range 16.2–60.0%, SD 18.2) compared to White-capped Albatrosses (average 85.0%, range 71.4–94.3%, SD 29.8) (U=0.0, p=0.004).

Within the South African EEZ, there was a correlation between White-capped Albatross locations and bathymetry and the location of trawl sets, but not the location of longline sets (Table 2). In contrast, Black-browed Albatross locations within the EEZ correlated significantly with the location of longline and trawl sets, but not bathymetry (Table 2). In GLM analysis, bathymetry (explained 39.1% variance), location of trawl sets (explained 16.8% variance) and location of longline sets (explained 4.0% variance) explained much of the variance in the distribution of White-capped Albatrosses (Table 3). Similarly, the location of trawlers (explained 30.8% variance), and bathymetry (explained 27.0% variance) explained much of the variance in the distribution of Black-browed Albatrosses (Table 3). The location of longline sets was also significant, but did not account for any further variance and led to an increase in the AIC, hence any additional explanatory power was negligible.

Table 2: Summary of the results of Spearman's rank correlations investigating the relationship between Black-browed and White-capped Albatrosses, and bathymetry, the presence of longliners and the presence of trawlers.

P-value/t-value	Black-browed Albatross	Bathymetry	Longline	Trawl	White-capped Albatross
Black-browed Albatross	*	-1.93	5.01	3.07	4.30
Bathymetry	0.06	*	-1.56	6.18	2.58
Longline	<0.001	0.12	*	1.60	1.76
Trawl	0.00	<0.001	0.12	*	3.22
White-capped Albatross	<0.001	0.01	0.08	0.00	*

Table 3: Summary of results of generalised linear modelling of Black-browed and White-capped Albatross movements in relation to the presence of trawlers, longliners and bathymetry.
Variance is expressed as a percentage.

Parameter	Black-browed Albatross		White-capped Albatross	
	Variance	p-value	Variance	p-value
Bathymetry	27.0	<.001	39.1	<.001
No of trawls	30.8	<.001	4.0	<.001
No of longline sets	0.0	<.001	16.8	<.001

Relationship with sea surface height

Black-browed Albatrosses undertook return trips into oceanic waters on nine occasions for a total of 76 days during the 370 day study period. On these trips, birds tended to follow the edges of warm anti-cyclonic and cold cyclonic eddies where the SHA gradient was the greatest (see Fig. 5 for typical examples ranging from Namibia to the South African east coast). The mean gradient of each PTT location was significantly higher than that of a set of randomly generated locations on seven of the nine trips (Table 4). Five of these trips were in a clock-wise direction and four were in an anti-clockwise direction.

Table 4: Characteristics of oceanic trips of Black-browed Albatrosses (direction and trip length) and summary of *t*-test results testing the difference between the SHA gradient of birds tracks compared to a set of random locations.

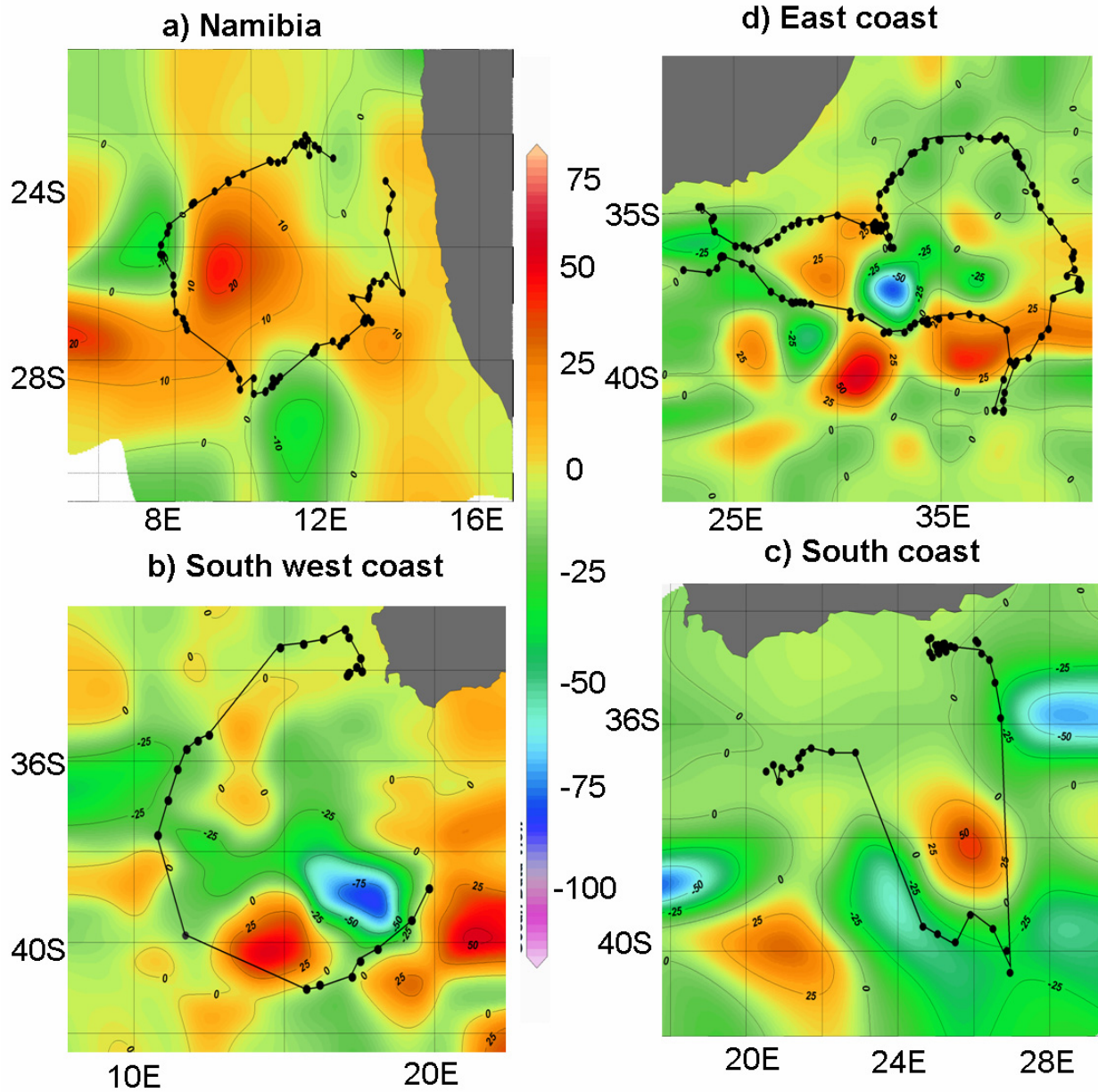| Trip	t-value	p-value	Direction	Trip length	
				Days	km
1	1.94	0.053	anti-clockwise	11	2 599
2	1.3	0.198	anti-clockwise	5	5 322
3	3.37	0.002	anti-clockwise	20	2 281
4	-1.13	0.26	anti-clockwise	7	1 724
5	3.85	<0.001	Clockwise	3	1 450
6	4.76	<0.001	Clockwise	3	4 603
7	4.11	<0.001	Clockwise	16	2 414
8	3.43	0.001	Clockwise	9	504
9	18.73	<0.001	Clockwise	2	1 976

Figure 5/…

Figure 5: Four example Black-browed Albatross tracks overlaid on 10-day average sea-surface height anomalies (SHA) derived from JASON-1, TOPEX, ERS-2, ENVISAT and GFO altimeters in four areas: a) off Namibia and off South Africa's south west coast (b), south coast (c) and east coast (d).

DISCUSSION

Influences on White-capped and Black-browed Albatross distribution

This study shows clear differences in the movements and distribution of White-capped and Black-browed Albatrosses in the southeast Atlantic. In general White-capped Albatrosses spent most of their time on the trawl grounds and all tracked birds largely remained within the EEZ. Black-browed Albatrosses on the other hand made frequent trips into oceanic waters. Of the birds tracked two adults and one juvenile left the southeast Atlantic on 27 September, 4 October and 23 October respectively and travelled across the Atlantic. In the case of the adults they arrived at South Georgia on the 17th October which coincides with dates of breeding commencement in late October (Phillips *et al.* 2005a). The immature bird remained on the continental shelf off Uruguay until its device stopped transmitting data 4 November and was therefore unlikely to attempt breeding (Phillips *et al.* 2005a).

Trawling activity taking place on the continental shelf break off South Africa provides large quantities of high quality food in the form of discards for both these species (Abrams 1983, 1985, Ryan & Moloney 1988). White-capped Albatrosses, which remained almost exclusively along the continental shelf edge, spent most of their time on the hake trawl fishing grounds. Water depth and trawler presence were therefore the strongest predictors of their distribution. There are no published studies on the movements of White-capped Albatrosses at other times of year, but tracking of the closely-related Shy Albatross indicates a similar preference for shelf waters, at least during the breeding season (Hedd *et al.* 2001). The importance of the continental shelf break including areas outside of the trawl grounds as a predictor of White-capped Albatross distribution indicates that they are also likely to be foraging naturally within South African waters. This is supported by a study during the 1980s which found that 60% of the diet of shy-type albatrosses in South African shelf waters comprised of natural prey (Crawford *et al.* 1991).

Black-browed Albatrosses, by comparison, spent relatively little time on the South African trawling grounds, instead feeding mainly outside of the South Africa EEZ. Nevertheless, while they were within the EEZ the presence of trawlers was the strongest predictor of their distribution. Fishery discards and offal were estimated to make up 80% of the diet of Black-browed albatrosses in South African shelf waters during the 1980s (Crawford *et al.* 1991). Therefore, it appears that this species exploits fisheries discards to a large extent while close to the South African coast, whereas it presumably feeds to a much greater extent on natural prey whilst on the return trips to oceanic waters.

Why do Black-browed Albatrosses make frequent trips into the open ocean?

The return trips undertaken by Black-browed Albatrosses into oceanic waters are consistent with the findings of Phillips *et al.* (2005a), who reported Black-browed Albatrosses in the southwest Atlantic to undertake return trips of 5–38 days duration. Return trips in the

southeast Atlantic were typically shorter in duration (2–20 days). A direct relationship might be anticipated between habitat usage and preference (Matthiopoulos 2003) in which case Black-browed Albatrosses may simply 'prefer' to forage in the open ocean, alternating with periods on the continental shelf. However, if accessibility is restricted for any reason, utilisation in theory becomes a function of preference and accessibility (Matthiopoulos 2003). Since trawling on the continental shelf waters of South Africa provides large quantities of predictable food, we might expect both Black-browed and White-capped Albatrosses to be attracted to this area. However, regions with high quality food will also have the highest levels of intra- and inter-specific competition (Ricklefs 1990). It may be that Black-browed Albatrosses in the Benguela make these periodic sorties into oceanic waters to reduce competition, given the likely dominance behind vessels of the considerably larger White-capped Albatrosses. Body size is considered to be an important driver of interference competition in a range of animals (Shoener 1970, Persson 1985, Dickman 1988, Balance *et al.* 1997, Wanless 1998). Others have postulated that size determines dominance hierarchies in foraging albatrosses (Cherel *et al.* 2002, Phillips *et al.* 2005a,b). Wanless (1998) investigated the relationship between size and foraging efficiency and reported that White-capped Albatrosses out-competed Black-browed Albatrosses, particularly for large prey items.

Are Black-browed Albatrosses using oceanographic features in the open ocean?

Physical and biological processes in the ocean affect the distribution and abundance of plankton, which in turn influences the distribution of seabirds and marine mammals (Piontkovski *et al.* 1995, Pakhomov & McQuaid 1996). Seabirds tend to concentrate at physical oceanographic features at different spatial scales, where prey tends to be aggregated (Haney *et al.* 1995, Pakhomov & McQuaid 1996). Results from this study suggest that Black-browed Albatrosses did not use particular bathymetric zones when far from the South African coast, but instead foraged along the margins of meso-scale oceanographic anomalies on their periodic round trips over deep water. Black-browed Albatrosses tended to move on the edge of both colder, cyclonic and warmer, anticyclonic eddies. These eddies not only concentrate organisms at their edges, but also exhibit enhanced nutrient levels, which increase primary productivity, and are thus areas where food is likely to be in considerably higher concentration than elsewhere in the open ocean (Ansorge *et al.* 1999).

Implications for conservation

Both the White-capped and Black-browed Albatross are listed by IUCN as of conservation concern (Near-threatened and Endangered respectively). Given the overlap between White-capped Albatross distribution and trawling activity in the southern Benguela ecosystem it is not surprising that this is the most common albatross killed by that fishery (39% of 18 000 birds killed per year) (Chapter 6). Our results indicate that when Black-browed Albatrosses forage within the South African EEZ, their distribution is also strongly correlated with the

presence of trawlers, and indeed they are the second most common species in bird bycatch (29%). Interestingly, the occurrence of longline fishing activity accounts for far less variance (4% in the case of White-capped Albatrosses and none in the case of Black-browed Albatrosses) compared to that of trawl activities (30.8% and 16.8% respectively). This is probably because far fewer discards are provided by longliners compared to trawlers, so they are generally less attractive to birds except when longlines are being set. However, longliners remain a serious threat: both species ingest baited hooks during setting, and are dragged under and drowned as a result (Ryan *et al.* 2002, Petersen *et al.* 2007, Chapter 1).

Given the importance of the southern Benguela ecosystem for non-breeding adult and juvenile Black-browed and White-capped Albatrosses, and the high level of interaction with trawl and longline fisheries in the region, several mitigation measures have been considered (MCM 2007). These include the introduction of closed areas where fishing is prohibited, and various means of reducing the attractiveness of vessels to seabirds. Given the dramatic changes in marine ecosystems as a result of past fishing activities (Worm *et al.* 2003), including dramatic changes in seabird behaviour and distribution resulting from the presence of fishing vessels (Cooper & Dowle 1976, Ryan & Moloney 1998, this study), the possibility exists that these management actions could place a further burden on these species.

It has been argued that some scavenging seabirds such as Great Skuas in Shetland, and Audouin's Gull *Larus audouinii* and Yellow-legged Gull *L. cachinnans* in the Ebro Delta, depend on fisheries discards for successful reproduction when natural prey abundance is reduced (Furness 1987, Oro *et al.* 1995). However, these are breeding birds with dependent chicks and are therefore tied to a central place. By comparison, the albatrosses in this study are much less constrained in terms of where they can forage and, moreover, will have much lower costs of flight than skuas and larids (Weimerskirch *et al.* 2000). Furthermore, this study presented evidence that Black-browed, and to a lesser extent White-capped Albatrosses targeted natural prey, despite the availability of fisheries discards. Seabirds have evolved in unpredictable, dynamic environments and as a result exhibit highly opportunistic foraging strategies, and catholic dietary tastes. The Black-browed and White-capped Albatrosses in the Benguela Upwelling System are likely to continue adapting their behaviour in response to the changing distribution and abundance of food, much as they would in an entirely natural system. This is supported by evidence from the Ebro Delta, where gulls readily switched their diet back to natural prey after a trawl moratorium was introduced (Oro *et al.* 1995). Moreover, others have argued that fisheries waste is not an essential part of the diet for most seabird populations (Garthe 1997). Given their extreme life histories, seabird populations are particularly sensitive to any reduction in adult survival (Warham 1996). Therefore, management regimes which reduce additional sources of mortality such as those resulting from fisheries interactions are likely to far outweigh the disadvantage of reduced discard and offal availability.

In practical terms, reducing the attractiveness of fishing vessels to seabirds can be achieved effectively in many situations by good offal and discard management (Brothers *et al.* 1999).

Offal and discard management regulations exist for both trawl and longline fisheries in South Africa (MCM 2007). In the case of the trawl fishery, discarding of offal is not permitted during setting when the use of a tori line is not compulsory because of the risk of entanglement with fishing gear. However, discarding offal is not regulated during trawling or between trawls (MCM 2007). It is also possible to equip vessels with macerators that allow finer waste to be discharged which sinks much more rapidly, or holding tanks that allow retention of waste on board, both of which greatly reduce the attractiveness of fishing vessels to larger seabirds in particular (Wanless 1998, MCM 2007). Existing mandatory South African longline fishery regulations require that the discharge of offal must not occur during setting, and during hauling must take place on the opposite side of the vessel from the hauling station where birds may get hooked (MCM 2007). An alternative means of managing fishing in the South African EEZ would be to introduce a system of closed areas. Results from our study, however, suggest that this would be ineffective given the very large foraging ranges and high mobility of the tracked birds.

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Chapter 8

IMPLICATIONS OF NIGHT SETTING FOR
SEABIRDS AND TARGET CATCHES

IMPLICATIONS OF NIGHT SETTING FOR SEABIRDS AND TARGET CATCHES

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ABSTRACT

Catch rates of seabirds in the South African pelagic longline fishery are substantially higher than the internationally and locally accepted target of 0.05/1000 hooks. Although local legislation and fisheries regulations demonstrate the countries resolve to address the issue, substantial numbers of birds are still killed each year. Generalised linear models indicate that the time of setting and moon phase were important indicators of sea bird mortality and therefore by limiting fishing to night setting and/or outside of full moon periods seabird mortality could be substantially reduced. The tuna directed fishery is required to set their lines at night, but not the swordfish directed sector. This decision is based on the premise that Swordfish *Xiphias gladius* catches are highest at dusk. Evaluation of observer data (1998–2005) confirms that Swordfish catch rates are the highest when setting takes place at dusk (6.56/1000 hooks). There was no effect on catch rates of Swordfish or tuna over full moon. Limiting fishing effort during full moon could therefore be considered as an additional management option without a disproportionate effect on the fishery.

INTRODUCTION

Seabird mortality in the South African pelagic longline fishery is almost an order of magnitude higher (0.44/1000 hooks) than the internationally and locally accepted target of 0.05/1000 hooks (Environment Australia 1998, FAO 1999, Cooper & Ryan 2003, Chapter 1). After individual vessels effects, moon phase was the strongest determinant of seabird bycatch catch rates in all generalised linear models (Chapter 1); a finding consistent with other studies (e.g. Klaer & Polacheck 1998). Seabird catch rates were significantly higher (1.07/1000 hooks) during full moon periods compared darker moon phases (0.09/1000 hooks) (Chapter 1). Light conditions also played an important role in explaining the observed bycatch rates in the South African large pelagic longline fishery (Chapter 1), again supported by findings elsewhere (Ashford *et al.* 1995, Klaer & Polacheck 1998, Brothers *et al.* 1999a).

South Africa is a member of the Agreement on the Conservation of Albatrosses and Petrels and consequently has an international obligation to implement management measures to reduce this mortality. Seabirds mitigation measures work by either keeping birds away from baited hooks (e.g. tori lines), reducing the time the hook is available to the birds (e.g. line weighting), making vessels or bait less attractive to the birds (e.g. offal management) or by avoiding peak periods of bird foraging (e.g. night setting) (Brothers *et al*. 1999a).

Peak foraging periods for albatrosses occur predominantly during the day and at night in bright moon conditions, particularly Shy Albatrosses are strongly influenced by moon phase (Harrison *et a*l. 1991, Hedd *et a*l. 2001). Black-browed Albatrosses are also active in the first part of the night i.e. from dusk till mid-night (Weimerskirch & Guionnet 2002). Peak foraging periods for petrels take place at night when competition is at its lowest (Cherel *et al*. 1996). White-chinned Petrels *Procellaria aequinoctialis* are particularly active two hours prior to sunrise (Cherel *et al*. 1996, Barnes *et al*. 1997).

Confining line setting to the hours of darkness between nautical dusk and dawn was first proposed in 1988 (Brothers 1991). Since then, this measure has been widely promoted and prescribed in fisheries regulations globally (Brothers *et al.* 1999b). South African fishery regulations require the tuna directed fishery, but not the swordfish *Xiphias gladius* directed fishery, to set their gear between nautical dusk and dawn (Cooper & Ryan 2003, DEAT 2005). The exclusion of the swordfish directed vessels was motivated by the perception that target catches are greater when setting takes place at dusk (Cooper & Ryan 2003). Despite this, night setting has been made a mandatory requirement in other swordfish fisheries, for example those in New Zealand and Hawaii (Gilman *et al.* 2007). Fishery closure over full moon periods has been advocated as a further measure to address high seabird bycatch in South African waters, but its effect on target catches have not been evaluated. This chapter investigates the validity of exempting Swordfish vessels from setting at night and the implications of fishery closure over full moon periods.

METHODS

See chapter 1 for a full description of methodology. In short, data on bycatch and target catches were collected by fisheries observers in the South African pelagic longline fishery during from 2 256 sets (4.4. million hooks), 1998–2005. Sets were defined as 'day' if the entire setting period took place between nautical dawn and dusk, as 'night' if the entire setting period took place between nautical dusk and dawn and 'dusk' or 'dawn' if it straddled nautical dusk or dawn. Moon phase was divided into eight phases: new moon, waxing crescent, first quarter, waxing gibbous, full moon, waning crescent, last quarter and waning gibbous. An ANOVA was used to test for differences in tuna and Swordfish catch rates between among day, night, dusk and dawn sets, and between moon phases.

RESULTS

Evaluation of observer data (1998–2005) reveal that Swordfish catch rates are significantly higher when setting takes place at dusk (6.56/1000 hooks) compared to during the day (5.54/1000 hooks), at dawn (1.46/1000 hooks), or at night (4.38/1000 hooks) ($F_{3,2244}$=40.8, p<0.001) (Table 1). Tuna catches were the highest when setting took place at dawn (40.35/1000 hooks), compared to during the day (28.51/1000 hooks), at night (24.42/1000 hooks) and at dusk (17.77/1000 hooks) ($F_{3,2244}$=14.4, p<0.001) (Table 1).

Table 1: Average, minimum, maximum and standard deviation of Swordfish and tuna catch rates in the South African pelagic longline fishery, at night, dusk, dawn and during the day.

Time of day	Swordfish catch rate (1000 hooks)				Tuna catch rate (1000 hooks)			
	Average	Min	Max	SD	Average	Min	Max	SD
Dusk	6.56	0.00	111.25	10.02	17.77	0.00	200.00	21.38
Night	4.28	0.00	110.00	8.23	24.41	0.00	892.86	33.57
Dawn	1.46	0.00	23.87	2.16	40.35	0.00	309.09	40.72
Day	5.54	0.00	35.00	8.01	28.51	0.00	174.03	35.61

There was no significant difference or trend in Swordfish catch rate ($F_{7, 2237}$=0.94, p= 0.47) between moon phases in the South African large pelagic fishery (Fig. 1). However, the highest catch rate for Swordfish was recorded at full moon (5.87/1000 hooks) and the lowest at new moon (3.56/1000 hooks). There was a significant difference, but no trend in the tuna catch rate ($F_{7, 2237}$=2.01, p=0.05) between moon phases (Fig. 1).
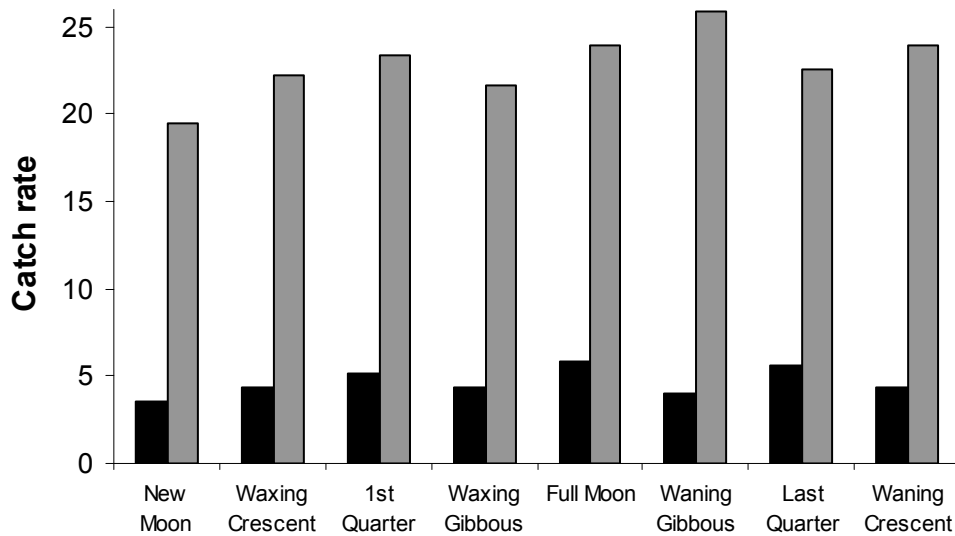
Figure 1: Trends in Swordfish and tuna catch rates per moon phase reported in observer data, 1998 –2005. Grey = tuna catch rates per 1000 hooks, Black= Swordfish catch rates per 1000 hooks.

DISCUSSION

This study shows that time of setting does have a substantial effect on the catch rates of target fish and explains the operational differences observed between tuna and swordfish fleets (Chapter 1). Vessels targeting Swordfish tend to set their gear in the evening (average 20h00) compared to tuna directed vessels who typically set their gear in the early hours (average 01h00) of the morning (Chapter 1). Similar findings were reported in a study conducted in the Hawaiian-based longline fishery which found that in general swordfish sets were characterized by a higher proportion of sets undertaken at night compared to tuna sets (He *et al.* 1997). These operational characteristics correspond well with the diurnal behavior of Swordfish, which are typically distributed in deep (>500 m) water during the day and near-surface waters at night (Carey & Robison 1981). Bigeye Tuna *Thunnus obesus* occupy depths >200 m during the day and move up to shallower waters at night and therefore tend to get caught on deeper set gear during the day or early morning (Grudinin 1989, Holland *et al*. 1990, Boggs 1992).

Managing fisheries in an ecosystems approach means balancing diverse objectives. This is a particular challenge in a developing country where poverty alleviation and job creation are national priorities. However, it is the role of the government to protect public goods, like seabirds. Based on observed seabird bycatch rates, it is recommended that South African fisheries regulations be amended to include night setting as a compulsory measure for all sectors. As a minimum day and dawn setting should be prohibited. Setting lines at night will reduce mortality, but not eliminate it and thus the need to use mitigation measures such as tori lines especially on bright moon nights is essential. This study also suggests that by limiting fishing during full moon periods seabird bycatch may be reduced. Although,

Swordfish catch rates have been reported to increase over full moon phases elsewhere (He *et al.* 1997, Bigelow *et al.* 1999), this finding was not supported by fisheries observer data in South Africa and thus should not have a disproportionate economic impact on the fishery. Closure of the fishery during full-moon could therefore be considered. This measure is likely to be more practical for swordfish directed vessels because their trips are typically 10-14 days compared to tuna directed vessels whose trips are typically of 3 month duration. For these vessels, reducing fishing effort over full moon would have economic impacts on the fishery.

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Chapter 9

GEAR CONFIGURATIONS, LINE SINK RATES AND SEABIRD BYCATCH IN PELAGIC LONGLINE FISHERIES

GEAR CONFIGURATIONS, LINE SINK RATES AND SEABIRD BYCATCH IN PELAGIC LONGLINE FISHERIES

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ABSTRACT

Optimal line sink rates of 0.3 m.s^{-1} are a requirement of the South African longline fishery, yet gear configurations to achieve this sink rate have not been established. Five gear configurations were investigated: the American longline system using no weighted swivel, 60 g and 120 g weighted swivels, the use of a wire trace and the Asian pelagic longline system. None of these weighting regimes achieved 0.3 m.s^{-1} consistently. The fastest line sink rates were achieved by the addition of a 120 g weighted swivel (0.35 m.s^{-1} to 10 m) which is an improvement from 60 g (0.24m.s^{-1} to 10 m) currently used by the South African Swordfish directed fishery. The gear used by the Asian longline fleet targeting tunas, sank at a rate of 0.19 m.s^{-1} to 10 m, which is likely to contribute to the larger seabird bycatch rates in this sector.

INTRODUCTION

Albatrosses are relatively shallow divers (0.3–12.4 m) (Prince *et al.* 1994). Therefore maximising the rate at which longlines sink, reduces the time the hook is within the reach of the birds, reducing seabird mortality. Some petrels however, can dive considerably deeper e.g. Sooty Shearwater *Puffinus griseus* can dive to a maximum depth of 67 m (Weimerskirch & Sagar 1996). For these birds maximizing the line sink rate will be less effective.

Various 'line weighting' regimes have been investigated and proposed for demersal (Robertson *et al.* 2003, Moreno *et al.* 2006, Chapter 12) and pelagic (Brothers *et al.* 2001, Anderson & Mcardle 2002) longline fisheries. Although gear configurations differ among fisheries, a line sink rate of 0.3 m.s^{-1} is recommended (Brothers *et al.* 2001, Anderson & Mcardle 2002, CCAMLR 2007). This sink rate allows hooks to reach a depth of 10 m while under the aerial coverage of a well constructed tori line (100–150 m) if the line is set at 7-8 knots (12.9–14.8 km.hr^{-1}).

The South African fishery uses three main gear configurations, associated with the swordfish, tuna and shark directed fisheries. The swordfish fishery typically uses the American longline system, comprising a monofilament mainline and dropperlines (Watson & Kersetter 2006) (Fig. 1). On average five dropperlines spaced approximately 40 m apart are attached between buoys. Dropper lines are attached to the mainline with a snap-on clip (tuna clip) and typically consist of an 8 fathom (14.6 m) upper section, a 60 g weighted swivel, a lightstick, 2 fathom (3.7 m) of lower section and a hook. The gear used by the shark fishery comprises an upper section of 5 fathom (9.3 m) leadcore (sukiyama), 1.5 fathom (2.8 m) monofilament, a 0.6 m wire trace and a hook. The tuna fishery were all Asian flagged vessels which typically do not include a weighted swivel, but frequently include five sections consisting of monofilament, lead core (sukiyama) and braided core. Branch lines are on average 40 m long (Watson & Kersetter 2006).
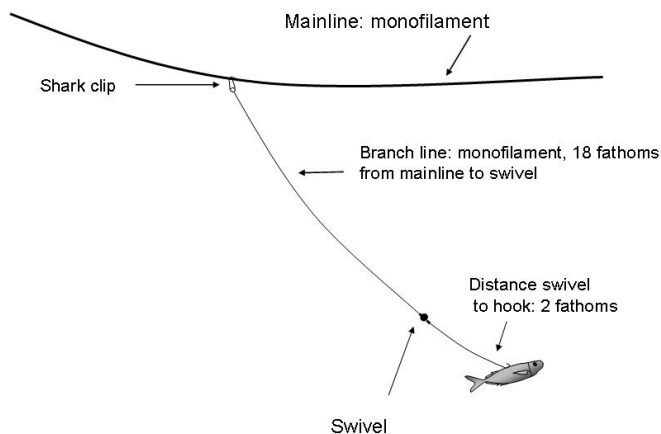

Figure 1: Typical branch line configuration of the American longline fishing technique.

METHODS

TDR deployment and calculation of line sink rates

Time-depth recorders (TDRs), manufactured by Wildlife Computers (Mk9) were deployed to collect line sink rates. The Mk9 TDR is 66.5 mm in length and 17 mm in height and width and weighs 30 g. It measures depth (0.5 m increments to 2 000 m, accuracy of 1% to 1 000 m), temperature (0.05 °C increment, range-40–60 °C), light intensity and wet/dry conditions. It has a 15 mb non-volatile flash memory. The TDR was attached 1 m from the hook. Before deployment, the TDR loggers were soaked for a minimum of 30 min in seawater to stabilize their temperature to ambient water temperature, preventing fluctuations of readings at deployment. Swordfish directed longlines were set by casting the branch line by hand out of the prop wash. The tuna longline gear, on the other hand was set using a line setter. In both cases this deployment technique is typical for the sector. The internal clocks of the TDRs were synchronized with the observer's watch before deployment to determine exact water entry times. These were verified using the wet-dry sensor which provided an independent reference for time of immersion. The data were downloaded after each set and the loggers were redeployed.

Line sink rates were calculated as depth of hook over time and were calculated to depths of 2 m, 5 m, 10 m and 15 m. In order to eliminate most seabird bycatch, the line should reach a minimum depth of 10 m i.e. a depth in excess of the average diving depth of the majority of vulnerable seabirds, in the time it is under the protection of the tori line (Huin 1994, Prince *et al.* 1994, Hedd *et al.* 1997). Since the average setting speed is 7–8 knots (13.0–14.8 km.h^{-1}) and the tori line has an aerial coverage of 100 m (Melvin *et al.* 2004), the line, in ideal conditions, should be protected by the tori line for approximately 26 s. The depth after 26 s is therefore also calculated.

Effect of gear configurations

Data were collected from swordfish, tuna and shark directed vessels. Three weighting regimes were compared for the swordfish fishery: (a) a non-weighted swivel was used on the branch line, b) one 60 g lead weighted swivel was added to the branch line 2 m above the hook and c) two 60 g weighted swivels were used in tandem on the branch line 2 m above the hook. For tuna and shark directed fisheries, line sink rates achieved with typical gear configurations were investigated. A one-way ANOVA was used to test differences in sink rates. Seabird bycatch was also recorded for each set.

RESULTS

Sink rates for 311 datasets were collected: 248 from the swordfish fishery (82 on unweighted branch lines, 83 with 60 g weights and 83 with 120 g weights), 53 from the tuna fishery and 10 from the shark fishery (Fig. 2 and Table 1). There was a significant difference in line sink rates between all groups to 2 m, 5 m, 10 m and depth at 26 s   ($F_{4,\,306}$=26.61, 45.63, 29.5 and 35.27; p<0.001, <0.001, <0.001 and <0.001 respectively) (Fig. 2). Line sink rates increased from the slowest recorded on unweighted branch lines, followed by gear typical of the tuna directed fishery, 60g weights and gear typical of the shark fishery i.e. use of a wire trace. The fastest average line sink rates were recorded in the swordfish directed fishery using a 120 g swivel, which was the only treatment which achieved the target rate of at least 0.3 m.s^{-1}. There was, however, large variation in line sink rates for all groups and standard deviations around the mean sink rates were substantial (Fig. 3 and Table 2). There were sets which reached the target rate at 30 s in all treatments (Table 3).

Table 1: Summary of line sink rate data collection on swordfish, tuna and shark directed vessels using time depth recorders deployed on commercial fishing vessels.

Vessel flag	Target	Line setter	Hooks	Birds	Catch rate (1000 hooks)	No. of datasets	No. of sets	No weight	60g	120g
South African	Swordfish	No	13848	0	0.00	33	11	11	11	11
South African	Swordfish	No	18184	0	0.00	30	10	10	10	10
South African	Swordfish	No	9550	0	0.00	15	5	5	5	5
South African	Swordfish	No	22200	4	0.18	38	13	12	13	13
South African	Swordfish	No	14730	0	0.00	33	11	11	11	11
South African	Swordfish	No	10190	16	1.57	24	8	8	8	8
South African	Swordfish	No	11050	9	0.81	24	8	8	8	8
Namibian	Swordfish	No	7150	0	0.00	6	2	2	2	2
Namibian	Swordfish	No	30770	1	0.03	45	15	15	15	15
South African	Sharks	No	10830	0	0.00	10	10	0	0	0
Asian	Tuna	Yes	16780	10	0.60	6	6	0	0	0
Asian	Tuna	Yes	103730	50	0.48	44	44	0	0	0
Asian	Tuna	Yes	6000	4	0.67	3	3	0	0	0
			275012	94	0.34	311	136	67	68	68

Table 2: Summary of line sink rates to 2 m, 5 m, 10 m and depth after 26 s, maximum depth of gear and seabird bycatch rates per experimental group.

|	Fishery	Gear configuration	2 m	5 m	10 m	26 s	max depth	Bird mortality (Catch rate)
Average	Tuna	Multi-section dropper	0.15	0.17	0.19	4.1	131	0.58
min	Tuna	Multi-section dropper	0.05	0.05	0.09	0.5	73	0.34
max	Tuna	Multi-section dropper	0.40	0.36	0.36	9.0	189	0.67
std dev	Tuna	Multi-section dropper	0.06	0.06	0.06	1.7	33	0.09
Average	Shark	Wire trace	0.29	0.35	0.28	7.6	33	0.00
min	Shark	Wire trace	0.12	0.20	0.18	5.0	23	0.00
max	Shark	Wire trace	0.40	0.50	0.42	10.0	51	0.00
std dev	Shark	Wire trace	0.11	0.12	0.10	2.0	10	0.00
Average	Swordfish	60 g swivel	0.24	0.27	0.24	6.6	51	0.29
min	Swordfish	60 g swivel	0.07	0.10	0.05	1.0	27	0.00
max	Swordfish	60 g swivel	0.67	0.63	0.59	13.5	120	1.57
std dev	Swordfish	60 g swivel	0.11	0.11	0.12	2.7	21	0.55
Average	Swordfish	120 g swivel	0.30	0.37	0.35	8.8		
min	Swordfish	120 g swivel	0.09	0.12	0.06	3.0		
max	Swordfish	120 g swivel	0.67	0.83	0.71	15.5		
std dev	Swordfish	120 g swivel	0.14	0.16	0.18	3.2		Same sets as
Average	Swordfish	No swivel	0.14	0.15	0.14	3.6		60 g swivel
min	Swordfish	No swivel	0.02	0.01	0.02	0.0		
max	Swordfish	No swivel	0.40	0.45	0.48	16.5		
std dev	Swordfish	No swivel	0.10	0.10	0.11	2.9		
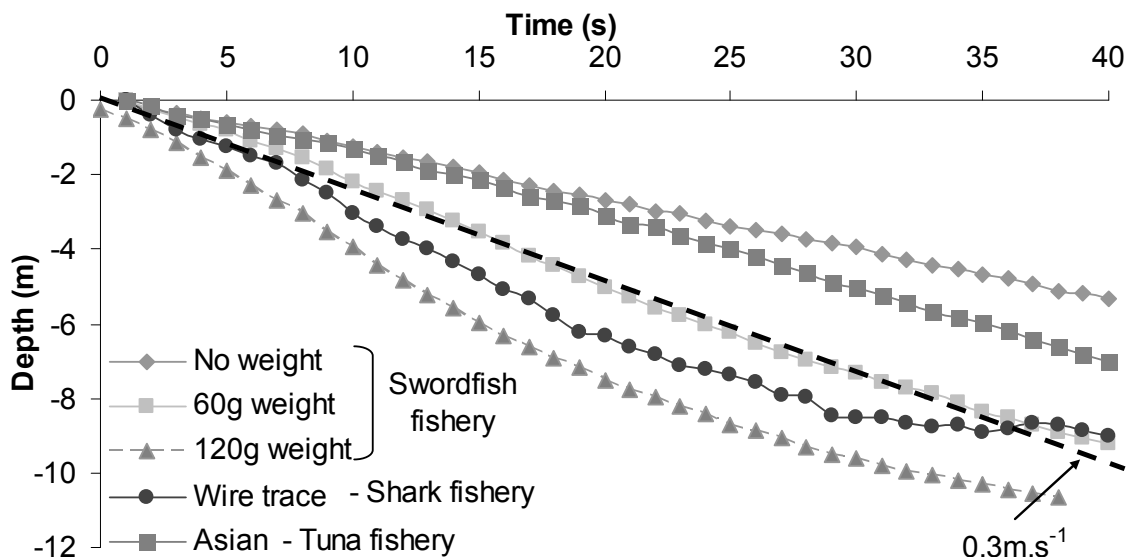
Figure 2: Average line sink rates for the American longline system with no weighted swivel, a 60 g or 120 g weighted swivels and a wire trace, and the Asian longline system.
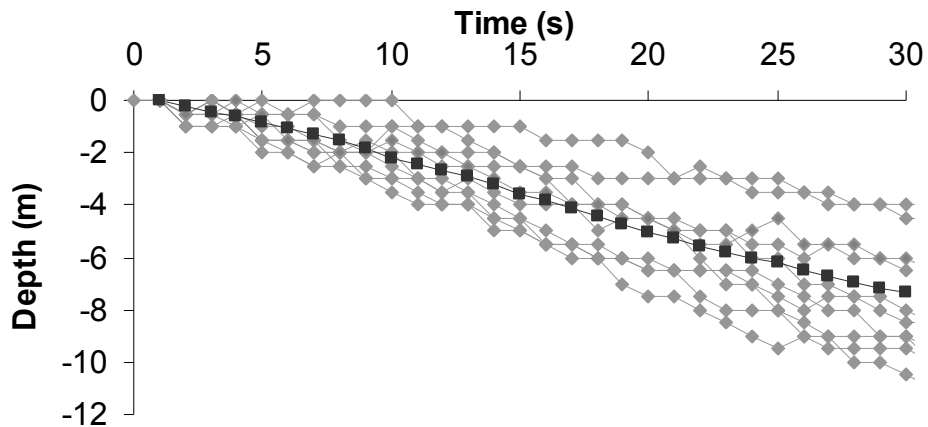
Figure 3: The mean line sink rate profile for 60 g weighting of American longline gear (typical of swordfish directed fishery) (black) and a random sample (n=10) (grey) of the 83 line trajectories sampled.

Table 3: Percentage of lines reaching 10 m after 20 s, 30 s, 40 s and 50 s for each fishery and gear configuration.

Fishery	Gear configuration	20s	30s	40s	50s
Tuna	Multi-section dropper	0%	2%	17%	34%
Shark	Wire trace	0%	30%	40%	50%
Swordfish	No swivel	1%	9%	12%	17%
Swordfish	60 g swivel	6%	18%	43%	59%
Swordfish	120 g swivel	25%	46%	65%	72%

The average maximum depth attained by Asian fishing gear type was 131 m. A line setter was used on all sets at an average speed of 12.6 (10.7–13.0) knots or 23.3 (19.8–24.1) km.hr^{-1}, which is faster than the longline was set (average speed of 7.1 (4–9.3) knots or 13.1 (7.4–17.2) km.hr^{-1}). Line setters were not used in the swordfish or shark directed vessels which achieved an average maximum depth of 51 m (Table 2).

Significantly (U=1379.0, p=0.003) more birds were killed by the tuna directed sector (64 of 94 or 68.1% on 53 sets) at an average catch rate of 0.58/1000 hooks compared to the swordfish directed sector where seabirds were caught at an average rate of 0.29/1000 hooks (Table 1). Overall, more birds were caught when the gear sank slower than at 0.3 m.s^{-1} or faster (Fig. 4).
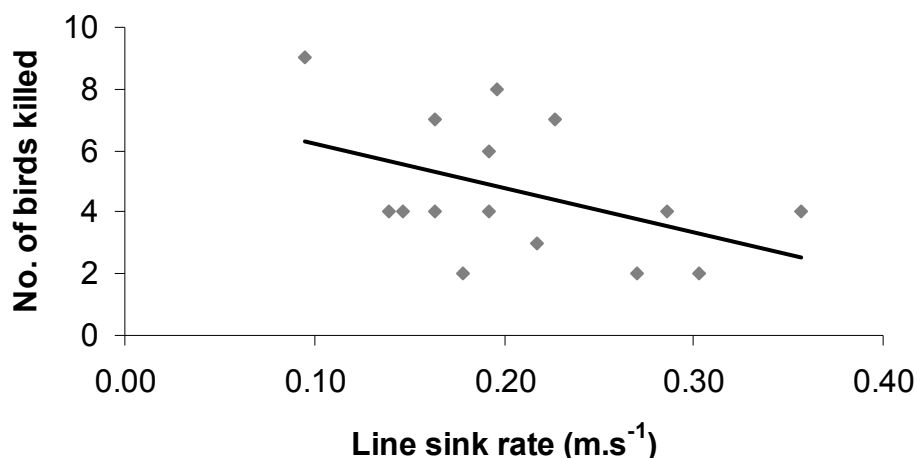
Figure 4: Bird mortality (sets where more than one bird was caught) plotted as a function of line sink rate.

DISCUSSION

The three species most commonly caught by the large pelagic longline fishery are the White-chinned Petrel, shy (*sensu lato*) albatross *Thalassarche steadi/cauta* and the Black-browed Albatross *T. melanophrys* (Chapter 1) which dive to average depths of 6.0 m (maximum 12.8 m) (Huin 1994), 1.9 m (maximum 7.4 m) (Hedd *et al.* 1997) and 2.5 m (maximum 4.5 m) respectively (Prince *et al.* 1994). Therefore once the hook reaches 10 m (equivalent to a sinking rate of 0.3 m.s^{-1} and assuming at least 100 m protection from a tori line and an 8 knot setting speed) they would be out of the reach of the species most at risk. No weighting regime achieved this sink rate consistently. A number of factors influence the rate at which the gear sinks. Firstly, heavier gear generally sinks more quickly, although this relationship is not linear. For the swordfish fishery, the proportion of sets which achieved the desired rate of 0.3 m.s^{-1} increased from 12% to 43% with the addition of a 60 g lead swivel. Adding another 60 g only increased the proportion by a further 12 %. In a similar study in New Zealand, adding a 60 g swivel within 1–2 m of the hook attained the line sink rate of 0.45 m.s^{-1} (Anderson & Mcardle 2002). Secondly, the location of the weight is also an important determinant of sink rate (Santani & Uozumi 1998, Brothers *et al.* 2001, Anderson & Mcardle 2002). Brothers *et al.* (2001) found that the heavier the weight, and the closer it is to the hook, the more rapidly the hook will sink. They attained sink rates of 0.3–0.6 m.s^{-1} using either an 80 g weight within 3 m of the hook, or a 40 g weight at the hook (Brothers *et al.* 2001). Pemberton *et al.* (1995) reported that by adding a 20 g weight at or near the hook, a sink rate of 0.5 m.s^{-1} can be achieved.

Other studies conducted in pelagic and demersal longline fisheries have reported that currents and turbulence also affect line sink rates (Brothers *et al.* 1999a), as can setting

speed and the use of a line setter (Løkkeborg & Robertson 2002, Cooper & Ryan 2003, Løkkeborg 2003, Robertson *et al*. in press). The faster the setting speed, the more tension or resistance is likely to be on the line, slowing sink rates (Robertson *et al*. in press). The use of a line setter can increase the line sink rate if it deploys the gear faster than speed the line is set at because this decreases tension on the line (Løkkeborg & Robertson 2002). In an experiment in the North Atlantic, longlines set with a line setter reached a depth of 3 m 15% faster then lines set without a line setter, although beyond this depth sink rates were similar (Løkkeborg & Robertson 2002). In this study, only tuna directed vessels used a line setter. This is likely to be the reason tuna directed gear achieved an average maximum depth of 131 m compared to 51 m achieved by swordfish directed vessels which did not use a line setter. This depth is far deeper than the length of the branch line which is 40 m in the tuna directed fishery and 20 m in the Swordfish directed fishery. Despite using a line setter, tuna gear was still too light to achieve optimal line sink rates. During line setting a considerable proportion of hooks were within the known diving range of a number of seabirds frequenting these vessels and may account for the high catch rates reported in this fishery (Huin 1994, Prince *et al.* 1994, Hedd *et al.* 1997). Future experiments to improve the line sink rate of Asian pelagic longline gear will be imperative to decreasing seabird mortality in South African waters since most birds (88%) were caught by this fleet at a rate of 0.51/1000 hooks compared to the South African swordfish fleet which caught seabirds at a rate of 0.23/1000 hooks (Chapter 1). Resistance to changing traditional gear configurations is likely to result from the notion that altering fishing gear configuration may affect the lofting of gear in the water and consequently tuna catches. Traditional gear in many cases includes one or two short (2 m) sections of lead core line in the branchline. An experiment investigating the effect of a longer section of lead core (i.e. 20 m) line positioned 2 m from the hook could be conducted. A pelagic longline can be up to 100 km in length and thus the catchability throughout the line may not be comparable. To overcome this one could alternate treatments and conduct a matched pairs analysis with the line sink rate and CPUE of target and seabirds as the response variables.

The number of birds caught decreased with increasing line sink rates. Few published studies demonstrate that increasing the line sink rate decreases seabird bycatch (Brothers *et al.* 1999a, Løkkeborg 2003). Even when optimal line sink rates are achieved, fishing gear is still within the reach of the birds for the short time the gear is sinking. Therefore the use of a tori line is important to prevent birds from reaching the fishing gear as it sinks. However, even with an effective tori line with 100 m protection, a large proportion of the hooks were still within the reach of the birds, indicating that no one mitigation measure is likely to solve the seabird bycatch issue and that it is important that a combination of measures be used.

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Chapter 10

THE USE OF CIRCLE HOOKS TO REDUCE TURTLE BYCATCH AND THEIR EFFECT ON OTHER VULNERABLE SPECIES

THE USE OF CIRCLE HOOKS TO REDUCE TURTLE BYCATCH AND THEIR EFFECT ON OTHER VULNERABLE SPECIES

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ABSTRACT

Catch rates of turtles in the South African pelagic longline fishery are lower than those reported in pelagic longline fisheries elsewhere, but small numbers of three endangered species and one critically endangered species are caught. No mitigation measures have been implemented to prevent turtle bycatch in this fishery. The use of circle hooks has been implemented elsewhere to reduce turtle capture and mortality. This study tests the effect of using circle hooks on catch rates of target and non-target species by alternating typical 'J' hooks with 18/0 circle hooks (n=11 400 hooks). Too few turtles were caught to demonstrate a benefit, but observations suggest that circle hooks may increase shark catches. Bait type also significantly effected catch rates of sharks and explained more variance than hook type.

INTRODUCTION

Turtle mortality in the South African pelagic longline fishery is low (average 0.04/1000 hooks) compared to catch rates (0.2–6.5/1000 hooks) in pelagic longline fisheries elsewhere (Lewison *et al.* 2004, Witzell 1999, Chapter 2). The South African pelagic longline fishery, nevertheless, catches at least three Endangered species and one Critically Endangered species of turtle and is likely to be responsible for the lower than expected recovery rates of turtle, especially Leatherback Turtles *Dermochelys coriacea*, populations breeding on the north coast of KwaZulu-Natal (Hughes 1974, Hughes 2001, Wright 2004, Chapter 2).

Amongst suggested mitigation measures, such as deeper sets, reducing the soak time, the use of time-area closures and voluntary measures such as moving to a new area after catching a turtle, is the replacement of 'J' hooks with circle hooks (Løkkeborg 2004, Watson *et al.* 2005, Gilman *et al*. 2006, Kersetter & Graves 2006, Read 2007). Hard-shelled turtles are most often caught by ingesting a hook, whereas soft-shelled turtles (e.g. Leatherback) tend to get entangled in gear or hooked in the body (Watson *et al.* 2005). The wider the hook, the less likely a turtle will swallow it. When swallowed, circle hooks tend to hook turtles in the mouth, rather than being deeply swallowed as typically occurs with 'J' hooks (Trumble *et al*. 2002, Cooke & Suski 2004). If a turtle is still alive when gear is retrieved, it is more likely to survive if it is hooked in the mouth rather than if it is hooked more deeply (Watson *et al*. 2005).

The effect of circle hooks on catches of target fish has been varied (Largarcha *et al.* 2005, Read 2007). In the Ecuadorian Mahi-Mahi *Coryphaena hippurus* fishery, circle hooks reduced fish catches to such a degree that their use was considered impractical (Largarcha *et al.* 2005). Whereas catch rates for Swordfish in the Hawaiian longline fishery increased by 16% with the use of circle hooks (Gilman *et al.* 2007). Thus circle hooks need to be tested under local conditions before they are implemented as a way to reduce turtle mortality (Read 2007).

METHODS

The effect of circle hooks on turtle bycatch and their impact on catch rates of other species was investigated in the South African Pelagic longline fishery by alternating 18/0 circle and 'J' shaped hooks on 34 sets (11 416 observed hooks) during four trips undertaken between April 2006 and September 2007. The effect of bait type (fish versus squid bait) was also investigated. Catch per hook, status (dead or alive) and hooking location were recorded for all species caught. Differences in catch rates between circle and 'J' hooks were investigated using matched pairs *t*-test. A two-way ANOVA was used to investigate the potential combined effect of bait and hook type and a generalised linear model (Genstat Committee 2007) to investigate the magnitude of bait and hook type effects.

RESULTS

Only one turtle, a Loggerhead *Caretta caretta* was caught during the study. It was caught on a J hook and swallowed the hook. A further fourteen species were caught during the study: three species of tuna (Bigeye *Thunnus obesus*, Yellowfin *T. albacares* and Longfin *T. alalunga*), Swordfish *Xiphias gladius*, two other species of fish (*Escolar* spp. and Angelfish *Brama brama*), five species of sharks (Blue *Prionace glauca*, Short-finned Mako *Isurus oxyrinchus*, Thresher *Alopias vulpinus*, Bronze Whaler *Carcharhinus brachyurus* and Smooth Hammerhead *Sphyrna zygaena*), three seabirds (shy-type albatross *Thalassarche cauta/steadi*, Black-browed Albatross *T. melanophrys* and White-chinned Petrel *Procellaria aequinoctialis*) (Table 1). There was no significant difference between catches on the 'J' hook and circle hook for all species except the Short-finned Mako Sharks (29.8/1000 hooks (n=170) on circle hooks versus 13.3/1000 hooks (n=76) on 'J' hooks, p=0.01) and total sharks (41.5/1000 hooks (n=243) on circle hooks versus 23.0/1000 hooks (n=131) on 'J' hooks, p=0.01) (Table 1 and 2).

Bait type affected catch rates of Blue (p=0.01) and Short-finned Mako (p<0.001) Sharks and explained more variance than hook type (9.2% compared to 0% for Blue Sharks and 50% compared to 2.5% in Short-finned Mako Sharks) (Table 2). Short-finned Mako sharks were predominantly caught on fish bait (48.4/1000 hooks) compared to squid bait (2.97/1000 hooks). The opposite was true for Blue Sharks which were predominantly caught on Squid bait (16.91/1000 hooks) rather than fish bait (16.9/1000 hooks) (Table 2).

Sharks (n=368) were caught on circle hooks in the mouth on 94.9% of occasions (the remaining 1.7% swallowed the hook and 3.4% were hooked in the body) whereas sharks caught on 'J' hooks swallowed the hook on 73.9% of occasions (26.1% were hooked in the mouth). 8.6% of sharks were dead when hauled on 'J' hooks compared to 1.7% of sharks on circle hooks.

Table 1/…

Table 1: Numbers of fish, sharks, seabirds and turtles caught on 'J' (n=5 708 hooks) and circle hooks (n=5 708 hooks) during experimental trials. Differences in catch rates per set were tested with paired sample t-test (n=34).

Sample	*t*-value	p-value	'J' hook	Circle hook
Swordfish	0.7	0.49	10	12
Bigeye Tuna	0	1.00	6	6
Longfin Tuna	1.35	0.19	20	35
Yellowfin Tuna	-0.72	0.48	9	6
Angelfish	1	0.33	1	0
Escolar	-1	0.33	0	2
Total tuna	1.14	0.26	35	47
Blue Shark	1.2	0.24	55	67
Short-finned Mako Shark	-2.68	0.01	76	170
Smooth Hammerhead	1	0.33	0	1
Thresher Shark	-1.82	0.08	0	3
Bronze Whaler	-1.45	0.16	0	2
Total sharks	2.87	0.01	131	243
Black-browed Albatross	-1	0.33	1	0
shy-type albatrosses	1	0.33	0	1
White-chinned Petrel	-1	0.33	1	0
Total birds	-0.44	0.66	2	1
Loggerhead Turtle	-1.44	0.16	1	0

Table 2: Summary of ANOVA results of catch rates of Short-finned Mako and Blue sharks on 'J' versus circle hooks and fish versus squid bait.

Bait type	Short-finned Mako Sharks				Blue Sharks			
	'J'	Circle	p-value	Total	'J'	Circle	p-value	Total
Fish	29.95	66.75	<0.001	48.35	0.00	3.42	0.62	1.71
Squid	1.78	4.15	<0.001	2.97	16.32	17.50	0.62	16.91
Total	13.31	29.78	0.01	21.55	9.64	11.74	0.66	10.69
p-value				<0.001				0.01

DISCUSSION

The fact that only one turtle was caught in this study is not surprising, given the low catch rates in the South African pelagic longline fishery and relatively small sample size. Preliminary mitigation studies conducted elsewhere have shown that using large 18/0 circle hooks significantly reduces turtle bycatch (Watson *et al.* 2005, Kerstetter & Graves 2006, Gilman *et al.* 2007, Read 2007). Circle hooks and mackerel bait were found to significantly reduce both Loggerhead (92% decrease) and Leatherback (67% decrease) Turtle interactions when compared with 'J' hooks and squid bait (Watson *et al.* 2005). Moreover, circle hooks significantly reduced the rate of hook ingestion by Loggerheads, reducing post-hooking mortality. Gilman *et al.* (2007) reported that the capture rates of Leatherback and Loggerhead Turtles significantly declined with the use of circle hooks in the Hawaiin longline fishery for Swordfish by 83% and 90%, respectively.

Other comparisons of 'J' versus circle hook catches of target fish have produced mixed results. Watson *et al.* (2005) reported an increase in the catch per unit effort (CPUE) of directed fisheries (Watson *et al.* 2005). Kersetter and Graves (2006) reported that catch rates for most species were not significantly different between hook types, although circle hooks generally increased tuna catch rates and decreased Swordfish catches. Hoey and Moore (1999) also suggested that the combination of circle hooks and dead bait would resulted in increased tuna catch rates. In the Hawaiian longline fishery for Swordfish, catch rates increased by 16%, but combined tuna species and combined Mahi Mahi, Opah *Lampris guttatus*, and Wahoo *Acanthocybium solandri* catch rates declined by 50% and 34%, respectively (Gilman *et al.* 2007). Although the sample size was limited, my study shows no significant difference between tuna and Swordfish catches on 'J' and circle hooks.

Given the inherent vulnerability of sharks to increased mortality, the significantly higher catches of sharks, and Short-finned Mako Sharks in particular, may reduce the desirability of implementing circle hooks as a mitigation measure in this fishery. Similar findings were reported in the Southwestern Indian Ocean where catch rates of Short-finned Mako Sharks increased from 2.2/1000 hooks on 'J' shaped hooks to 2.7/1000 hooks on circle hooks using mackerel bait in both cases (Ariz *et al.* 2005). However, the effect of bait type was greater than hook type, with catch rates decreasing to 0.8/1000 hooks using squid (Ariz *et al.* 2005), supporting the similar finding in the present study. Conflicting results were reported in the Hawaiian longline fishery where catch rates of sharks were 36% lower with the use of circle hooks and fish bait (Gilman *et al.* 2007) and caution must be voiced since the present study is based on a limited sample size. The natural diet of both Blue and Short-finned Mako Sharks includes squid and bony fishes (Stillwell & Kohler 1982, Kohler 1987).

Higher catch rates on circle hooks may be the result of larger retention of sharks because of the location of hooking (in the jaw) rather than an overall higher catch rate. Numerous studies have reported a significant reduction in the proportion of turtles and sharks that

swallowed circle hooks compared to being hooked in the mouth, which may increase the likelihood of survival (Yamaguchi 1989, Falterman & Graves 2002, Watson *et al.* 2005, Gilman *et al.* 2007, this study). Fish versus squid bait may reduce turtle catches and also contribute to addressing concerns over the sustainability of Blue Shark catches, but may negatively affect Short-finned Mako sharks. It is recommended that this be investigated further in the South African large pelagic fishery before implementation is considered.

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Chapter 11

*IS A 5% FIN TO TRUNK RATIO UNIVERSALLY
APPROPRIATE AS A DISINCENTIVE
TO CATCH SHARKS?*

IS A 5% FIN TO TRUNK RATIO UNIVERSALLY APPROPRIATE AS A DISINCENTIVE TO CATCH SHARKS?

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ABSTRACT

Vulnerable life-histories, global under-reporting, worldwide stock status concerns and local evidence of unsustainable fishing make it important for pelagic sharks to be managed in a precautionary manner. The practice of shark finning is considered wasteful. Full utilization of shark catch is encouraged by South African legislation which requires fishers to retain shark trunks in an 8% fin to trunk ratio. This is higher than the globally accepted norm of 5%. It was found that South African operators tend to maximize the flesh remaining on shark fins resulting in fin to trunk ratios of 15.91% and 7.83% for Blue *Prionace glauca* and Short-finned Mako Sharks *Isurus oxyrinchus* respectively. Thus a fin to trunk ratio of 8% may be appropriate for the South African fishery. However, given the re-introduction of Asian vessels into the fleet since 2005 the validity of a higher fin to trunk ratio for this fleet should be investigated.

INTRODUCTION

Shark fisheries are widely accepted as being unsustainable because most sharks have life-history strategies that make them inherently vulnerable to overexploitation (Hoenig & Gruber 1990, Musick *et al.* 2000, Barker & Schluessel 2005). These life-history implications in conjunction with poor quality data, global under-reporting, illegal trade, few data on shark biology and stock status concerns make it important for pelagic sharks to be managed with a precautionary management approach. Given the observed decreasing CPUE and a decreasing length frequency for Blue *Prionace glauca* and Short-finned Mako *Isurus oxyrinchus* Sharks in the South African pelagic longline fishery for tunas and Swordfish *Xiphias gladius*, management options for at least these two species should be considered (Chapter 3).

Shark finning, the retention of shark fins and discarding the remainder of the carcasses at sea, is considered wasteful and full utilization of catch is promoted (FAO 1995, Prestowitz 1996, Shivji *et al.* 2002, Lack & Sant 2006). This may be best achieved by regulating that the carcass and fins, if cut off, are landed together, as is required in South Africa (Lack & Sant 2006, DEAT 2007). The retention of the trunk, which is often of very low commercial value, takes up scarce hold capacity, incurs refrigeration costs and the possibility of contamination of higher valued fish by the high ammonia content of the meat (Lack & Sant 2006). These factors provide a disincentive to retain whole sharks with only the higher-valued fins retained. Current practice worldwide is to regulate with a 5% fin to dressed weight ratio (ICCAT 2004, IATTC 2005, IOTC 2005, Cortés & Neer 2006, SEAFO 2006). The appropriateness of this regulation has been under debate at the various Regional Fisheries Management Organisations (RFMOs) because of differences between species and processing techniques between fishers and fleets (ICCAT 2004, IOTC 2005). South African regulations deviate from the 5% internationally accepted norm, with a fin:trunk ratio of 8%. This study investigates the validity of this regulation to inform the management of this sector and relevant RFMOs.

METHODS

On selected shark and tuna-directed longline vessels, scientific Observers employed by Capricorn Fisheries Monitoring cc[1] brought in whole (wet) specimens of blue and mako sharks for processing ashore. This facilitated accurate weighing and cutting of fins identical to that normally conducted at sea.. Measurements (to the nearest cm), weights (to the nearest 100 g) and sex were recorded on shore to ensure accuracy. South African crew undertook the processing to simulate processing at sea.

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[1] Observer service provider organization based in Cape Town

The fins for export by the South African directed shark fishery include the dorsal fin, both pectoral fins, ventral flaps and the caudal fin. The dorsal fin is the most valuable and buyers will not accept any excess flesh. Therefore it is cut with a straight cut. Pectoral fins, on the other hand, are cut in a half moon shape to maximise the flesh on the fin and hence increase the weight. Anal fins are prepared by first removing a piece of flesh including the anal fin and the claspers in a male. Processing of Blue Sharks frequently involve the removal of the belly flaps. The caudal fin is cut at the pre-caudal pit and therefore includes a considerable amount of flesh.

RESULTS

Average fin to dressed weight ratios differed significantly between Blue (15.9%) and Short-finned Mako (7.8%) sharks (t_{21}=19.7, p<0.001) (Table 1). Average fin to round weight ratios also differed between Blue (6.8%) and Short-finned Mako Sharks (5.0%) (t_{21}=5.17, p<0.001) (Table 1). There was a good correlation between dressed weight and fin weight for both Blue (R^2=0.97) and Short-finned Mako (R^2=0.97) Sharks (Fig. 1). This relationship was not as evident for round weight versus fin weight (Blue R^2=0.60 and Short-finned Mako R^2=0.54 Sharks).

Table 1: Summary (average, standard deviation, range and Coefficient r^2) of Blue and Short-finned Mako Shark dressed weight to total, dorsal, pectoral and anal fin weights in the South African pelagic longline sector.

Blue Shark(n=5)	Average	Std deviation	Minimum	Maximum	Coefficient r2
Total Fin	15.9%	1.4%	14.6%	17.8%	0.97
Dorsal Fin	1.4%	0.3%	1.0%	1.6%	0.93
Pectoral Fin	3.1%	0.4%	2.8%	3.8%	0.89
Anal Fin	2.3%	0.5%	1.7%	2.7%	0.81
Short-finned Mako Shark (n=18)					
Total Fin	7.8%	0.6%	6.6%	8.9%	0.96
Dorsal Fin	0.8%	0.1%	0.6%	0.9%	0.97
Pectoral Fin	1.6%	0.3%	1.2%	2.1%	0.86
Anal Fin	0.6%	0.1%	0.4%	0.9%	0.88

Blue Shark

$$y = 0.135x + 0.243$$

(Fin weight vs Trunk weight)

Blue Shark

$$y = 0.040x + 0.656$$

(Fin weight vs Total weight)

Short-finned Mako Shark

$$y = 0.067x + 0.183$$

(Fin weight vs Trunk weight)

Short-finned Mako Shark

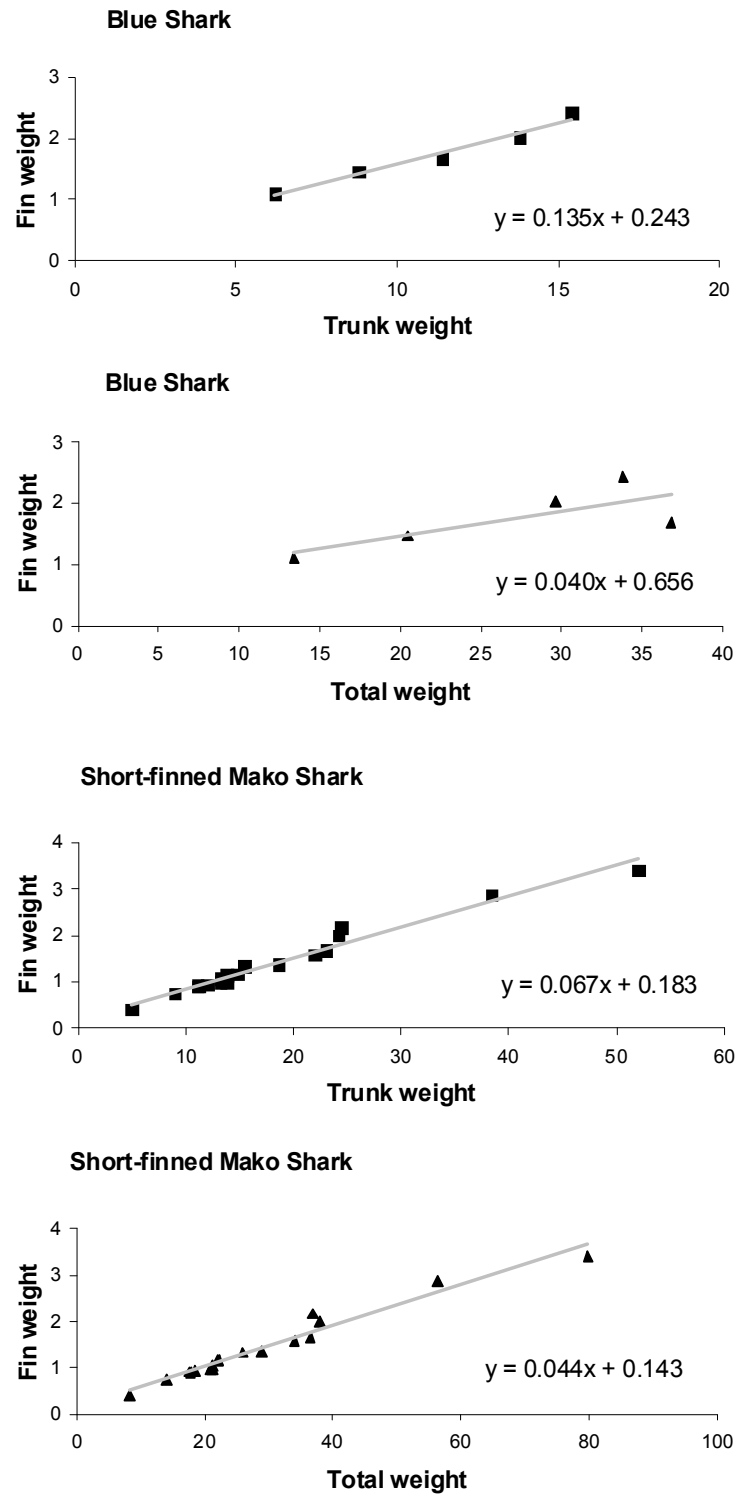$$y = 0.044x + 0.143$$

(Fin weight vs Total weight)

Figure 1: Graphical representation of the relationship between fin weight (kg), and trunk and total weights (kg) for Blue and Short-finned Mako sharks.

DISCUSSION

Fin to trunk ratios exceeded the 5% recommendation for both species, highlighting differences in on-board processing South African operators maximize the flesh remaining on the fins, remove the belly flaps and freeze fins rather than dry them (typical of Asian fleets). These will all contribute to the higher fin to trunk ratio observed in this fleet. These findings are similar to the higher fin to trunk ratios reported in the Spanish pelagic longline fishery (5.8% for Short-finned Mako and 14.7% for Blue Sharks (Mejuto & Garcia-Cortés 2004). However, in a review of data collected by the US National Marine Fisheries Service and the University of Florida Commercial Shark Fishery Program on fin to trunk ratios, the World Conservation Union concluded, that the use of 5% as a target figure in shark fishery management plans already allows considerable flexibility for species-specific variation in fin to dressed weight and should not be exceeded (IUCN 2003). Furthermore, this conclusion was supported by Cortés and Neer (2006) who suggested that if species specific management was not feasible, the available data suggest that an aggregated 5% ratio was not inappropriate when using the primary fin set. Therefore, although this study supports a higher fin to trunk ratio for the South African domestic operators, it is unlikely to be relevant for other fleets. Therefore given the re-introduction of Asian vessels into the South African pelagic longline fishery, the validity of a higher fin to trunk ratio for this fleet should be investigated prior to implementation.

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Chapter 12

LINE SINK RATES AND IMPLICATIONS FOR SEABIRDS, TARGET AND NON-TARGET FISH SPECIES IN THE SOUTH AFRICAN DEMERSAL LONGLINE FISHERY

LINE SINK RATES AND IMPLICATIONS FOR SEABIRDS, TARGET AND NON-TARGET FISH SPECIES IN THE SOUTH AFRICAN DEMERSAL LONGLINE FISHERY

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ABSTRACT

Although longline fishing gear is regarded as relatively selective, it is now evident that vulnerable species such as seabirds, turtles and sharks are caught incidentally. The demersal longline fishery operating in the Benguela Current Upwelling System off South Africa is no different. Endangered seabirds, endemic sharks and skates and a collapsed stock of Kingklip *Genypterus capensis* are amongst the bycatch in this fishery. Increasing line sink rates to 0.3 m/s by adding weights to longlines has been proposed and implemented to reduce seabird bycatch, but gear configurations to achieve this sink rate have not been established for the hake fishery, nor has the effect of increased weighting been tested on target and non-target species. For various weighting regimes (4, 6, 8 kg weights spaced at 40, 50 and 60 fathoms). No significant difference was found in the sink rate to 2 m, 5 m, 10 m and 15 m for dropper lines. However, there was a significant difference in the sink rate for the portion of the line near the weight. 4 kg weights sank significantly slower than the 6 kg weights which in turn sank slower than the 8 kg weight. There was no significant difference in the catch rate of hake *Merluccius* spp. between the dropper (84.73/1000 hooks) and the weight (100.53/1000 hooks), but there was a significant increase in the catch rate of Kingklip, the three most commonly caught demersal sharks (Short-spine Spiny Dogfish *Squalus mitsukurii*, Yellow-spotted Catshark *Scyliorhinus capensis* and Izak Catshark *Holohalaelurys regani*) and the most commonly caught skate (Biscuit Skate *Raja straeleni*) near weights compared to near droppers. Thus while hake catches are unlikely to be reduced by increased weighting, other vulnerable species of fish, shark and skate may be affected. Given that relatively few birds are caught in this fishery off South Africa, the increased impact on non-target fish species may outweigh the potential benefits of increased weighting on reduced seabird bycatch.

INTRODUCTION

Longline fishing is regarded as a relatively selective fishing technique (Lokkeborg & Bjordal 1992), but more recently it has become evident that it results in a suite of vulnerable species being caught incidentally (Brothers *et al.* 1991, Bergin 1997, Croxall & Gales 1998, Gales 1998, Witzell 1999, Myers & Worm 2003, Lewison *et al.* 2004, Barker & Schluessel 2005). The species most severely affected by incidental mortality in demersal longline fishing operations include a number of seabirds (Moreno *et al.* 1996, 2006, Bergin 1997, Croxall & Gales 1998, Gales 1998, Nel *et al*. 2002Favero *et al.* 2003, Delord *et al.* 2005, Chapter 4), sharks and skates (Myers & Worm 2003, Barker & Schluessel 2005, Lack & Sant 2006, Chapter 5). In general, long-lived species with low fecundity and low recruitment rates are more likely to be adversely affected than short-lived species with high fecundity and high reproductive capacity to recover (Warham 1996, Musick *et al.* 2000). Conservative life-history traits render these species especially vulnerable to increased adult mortality (Warham 1996, Musick *et al.* 2000). Species with small global populations and limited geographical distribution are especially vulnerable.

The Benguela Upwelling System, off the west and south coasts of South Africa, contains a high degree of endemism especially amongst the skates and demersal sharks (Compagno *et al.* 1991). Of 1 240 species of chondrichthyans worldwide, 223 species occur in South African waters, at least 59 of which are endemic to the Benguela (Compagno *et al.* 1989, Compagno 2000). At least 30 of these species have been recorded as bycatch in trawl and longline fisheries, and the sustainability of these catches has been questioned (Smale & Compagno 1997, Walmsley 2007, Chapter 5). Furthermore, the Benguela Upwelling System, along with the commercial fisheries it supports, provides rich foraging for a range of pelagic seabirds which are vulnerable to mortality in longline fishing (Abrams 1985, Barnes *et al.* 1997). Their incidental mortality in longline fisheries has been well documented in South African waters (Barnes *et al*. 1997, Ryan & Boix-Hinzen 1998, Ryan *et al.* 2002, Chapter 1 and 4) and globally (Brothers *et al.* 1991, Bergin 1997, Croxall & Gales 1998). Mounting evidence suggests that longline mortality is the leading cause of observed decreases amongst many albatross and petrel populations (Gales 1998, BirdLife International 2007).

In longline fisheries, seabirds are caught and drowned when they snatch baited hooks as the lines are set (Brothers *et al.* 1999). Albatrosses are relatively shallow divers, 0.3–12.4 m (Prince *et al.* 1994), although some petrels can dive considerably deeper (e.g. Sooty Shearwaters *Puffinus griseus* dive up to 67 m; Weimerskirch & Sagar 1996). White-chinned Petrels *Procellaria aequinoctalis*, the most commonly caught seabird by this fishery (Barnes *et al*. 1997), dive to depths of 14 m (Huin 1994). Demersal fisheries operating in the Benguela typically set their gear to depths of 100 m to 600 m (Japp 1993), far beyond the reach of vulnerable seabirds. It is only during the short period while the line sinks that seabirds are caught. By increasing the rate at which the line sinks, the time when hooks are within the reach of birds is reduced and mortality minimised. An optimal line sink rate of

0.3 m.s^{-1} has been proposed (Robertson 2000) and implemented (CCAMLR 2005) as a mitigation measure to reduce seabird bycatch. Although required in the South African hake fishery regulations (Cooper & Ryan 2003), gear configurations to achieve this sink rate have not been established, nor has the effect of increasing the mass or reducing the spacing of weights been tested on target and non-target species. Such trials have been conducted elsewhere (Robertson 2000, Robertson *et al.* 2003, 2006, in press, Moreno *et al.* 2006, Seco Pon *et al.* 2007) and mortality rates of White-chinned Petrels have been shown to be up to 13 times lower by vessels using appropriate weighting regimes in the Patagonian Toothfish *Dissostichus eleginoides* fishery around Kerguelen Island (Delord *et al*. 2005). None of these studies evaluated the modified Spanish system gear configuration used in the demersal longline fishery operating in the Benguela Upwelling System.

The demersal longline fishery which targets Cape hakes *Merluccius capensis* and *M. paradoxus* in South African waters has the added complication that it overlaps substantially with a collapsed stock of Kingklip *Genypterus capensis* (Japp 1993). The South African fishing fleet predominantly uses the demersal double longline technique developed by the Spanish, which can be varied to target either Kingklip or hakes (Japp & Wissema 1999). Hake display a diurnal migratory pattern (Botha 1980). They occur on or just above the seabed during the day and move up through the water column to middle depths at night. Demersal longliners targeting hake therefore tend to use relatively buoyant gear which they set after midnight to catch fish returning to the seabed in the early morning (Japp 1993). In contrast, heavier gear which lies on the sea floor is used when targeting Kingklip or toothfish which occur on the seabed.

The South African hake longline fishery typically sets lines up to 30 km long (Japp 1993). Each line has an anchor line at either end, and two lines, a top and bottom line, between the anchor lines. The top line is more buoyant than the bottom line and is attached to the bottom line by dropper lines at approximately 54 m intervals (Fig. 1). The sections of the bottom line between droppers are referred to as 'pots' because of their storing method. Hooks (approximately 15 000, 1.5–2 m apart), weights and floats are attached to the bottom line (Japp 1993). Weights, typically 6 kg, are spaced approximately 108 m or 60 fathoms apart (Fig. 1). This weighting regime falls short of that currently required by CCAMLR (Commission for the Conservation of Antarctic Marine Living Resources) for demersal longlining in the convention area (6 kg weights spaced 20 m apart or 8.5 kg weights spaced 40 m apart), a guideline which has been used for fisheries regulations in South Africa (CCAMLR 2005, Brothers 1995, Agnew *et al.* 2000). Because the South African fishery targets hake and not Patagonian Toothfish, the appropriateness of this measure has been questioned. The possibility exists that by decreasing the spacing between weights or increasing the mass of weights a higher proportion of the line will lie closer to the seabed and thus increases the bycatch of Kingklip and other non-target species such as demersal sharks and skates.

Increasing longline sink rates may reduce seabird mortality, but there is a need to recognize and explore potential impacts on other non-target species. Evaluating the effect of line sink rates on seabirds and ignoring the remaining ecosystem may result in simply shifting the conservation concern to other vulnerable species or reducing the fisheries ability to catch their target species. This study investigates the effect on target and non-target catch rates using two methods of optimising line sink rates: (a) increasing the mass of the weights, (b) decreasing the spacing of weights.
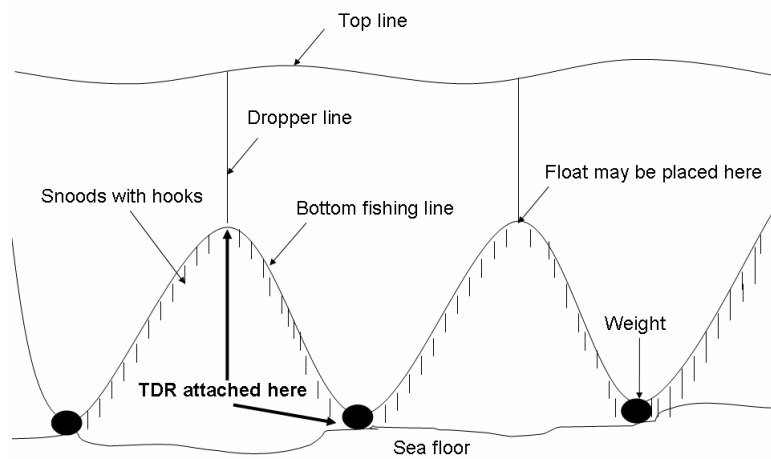

Figure 1: Typical gear configuration used by the South African demersal hake longline fishery, showing the TDR attachment points used in this study.

METHODS

Trials were conducted on 11 voyages with a total of 49 sets on four commercial demersal longliners targeting hakes from March 2005 to October 2007. Fishing took place on the continental shelf at depths from 205 to 415 m along the west and south coasts of South Africa. 'ANCORA' hooks baited with either Sardine *Sardinops sagax* or Mackerel *Trachurus trachurus capensis* were used. Approximately 13 000 hooks over 22 km (two lines of 11 km each) were deployed per set. Lines were set at night beginning between 01h30–03h30 (approximately 45–60 min duration) and hauled from 08h00 for approximately eight hours.

TDR deployment and calculation of line sink rates

Time-depth recorders (TDRs), manufactured by Wildlife Computers (Mk9) were deployed to collect hook depth over time or line sink rates. The Mk9 TDR measures depth, temperature, light intensity and wet/dry conditions. TDR loggers were attached to the fishing gear by splicing them into the multi-filament fishing line either at the weight or dropper or float (Fig. 1). Full details of deployment and description can be found in chapter 9. Line sink rates were

calculated as depth of hook over time to depths of 2 m, 5 m and 10 m. Depth after 26 s was also calculated because this is the average time the line is be protected by a well-constructed tori line (Melvin *et al*. 2004).

The effect of mass and spacing on line sink rates

To test the effect of mass on the line sink rate, three different concrete weights (4 kg, 6 kg and 8 kg, weighing 2.5, 4 and 6 kg respectively in sea water) were used and their sink rates measured using TDRs. It was not practical to test concrete weights with a mass of greater than 8 kg as space was limited on board these vessels. In each case the weight (either 4, 6 or 8 kg) was placed on 10 consecutive attachment points. One TDR was attached to the fifth weight and another to the adjacent dropper (which may or may not have had a float attached to it) (Fig. 1). To test the effect of reducing the spacing between the weights, two intervals, 40 fathoms and 50 fathoms (1 fathom = 1.829 m), were compared to the conventional spacing regime of a 6 kg weight at 60 fathom intervals.

Line sink rates to depths of 2 m, 5 m, 10 m and 15 m were modelled using multiple linear regression (Quinn & Keough 2002). Genstat 9 was used for model fitting (Genstat Committee 2007). Explanatory variables investigated included season, area, vessel name, TDR position (i.e. weight, dropper or float), mass of weight (4, 6 and 8 kg) and spacing between weights (60, 50 and 40 fathoms) as well as interactions between position, mass and spacing.

The effect of weighting on catch composition

Catch composition was recorded for every hook for approximately 10 min intervals (approximately the length of time it takes to haul the line between five weights and droppers/floats) every 20 min for the entire line. The catches of 10 hooks on either side of the weight, dropper or float (15 cm) were compared to see whether changing the spacing regime may impact on species composition. An ANOVA, matched pairs *t*-test and one sample *t*-test were used to investigate the difference in the catch composition between the weight, float and dropper locations.

RESULTS

A total of 169 time-depth datasets were collected under various regimes (weight: n=86, dropper: n=68, float: n=15) (Table 1).

The effect of mass and spacing on line sink rates

There were no differences in line sink rates between vessels or areas fished, so these terms were excluded from further analyses. All models selected were significant (p<0.001), accounting for 36.2% (sink rate to 2 m), 79.6% (5 m), 75.7% (10 m) and 70.9% (15 m) and

85.0% (depth after 26 s) of the variance and included three variables: location (dropper or weight), mass of weight (4, 6 or 8 kg) and spacing (40, 50 or 60 fathoms). In all cases the line adjacent to the weight sank significantly faster (0.41 m.s^{-1} to 10 m) than the dropper (0.14 m.s^{-1} to 10 m) (Table 1, Fig. 2). There was no significant difference in the sink rate between a 'float' and a 'dropper' and therefore dropper and float data were pooled for further analysis. This may be an artefact of a small sample size which requires further investigation. There was substantial variation in line sink rates especially within the first 2 m (Table 2, Fig 3).

Difference in sink rates of the droppers associated with a 4 kg (0.12 m.s^{-1}), 6 kg (0.15 m.s^{-1}) or 8 kg (0.13 m.s^{-1}) weights were not significant (Table 1, Fig. 4). The 4 kg weight sank significantly more slowly (0.31 m.s^{-1}) than the 6 kg weight (0.39 m.s^{-1}), which in turn sank more slowly than the 8 kg weight (0.56 m.s^{-1}) (Table 1). Reducing the spacing between weights had no significant effect on sink rate of weights, droppers or floats (Table 1).

Table 1: Sample size and average sink rates (m.s^{-1}) for all experimental groups (4, 6 and 8 kg; 40, 50 and 60 ftm) to 2 m, 5 m, 10 m, 15 m, depth after 26 s and time to 10 m.

| TDR position		Sample size				Average sink rate m.s^{-1}					
Location	Mass	40 ftm	50 ftm	60 ftm	Total	2 m	5 m	10 m	15 m	Depth (m) at 26 s	Time (s) to 10 m
Dropper	4 kg			18	18	0.14	0.10	0.12	0.12	1.6	83
	6 kg	11	10	20	41	0.15	0.11	0.15	0.14	1.3	66
	8 kg			9	9	0.07	0.11	0.13	0.14	1.7	76
Sub-total		11	10	47	68	0.14	0.11	0.14	0.13	1.6	71
Float	6 kg	3	10	2	15	0.12	0.08	0.11	0.11	0.5	90
Sub-total		3	10	2	15	0.31	0.08	0.11	0.11	0.5	90
Weight	4 kg			15	15	0.56	0.43	0.31	0.25	8.6	32
	6 kg	14	20	20	54	0.68	0.53	0.39	0.32	9.3	26
	8 kg			17	17	0.79	0.66	0.56	0.59	13.0	18
Sub-total		14	20	52	86	0.68	0.54	0.41	0.36	10.0	25
Grand Total		28	40	101	169	0.43	0.32	0.27	0.25	5.8	36
Depth at 26 s		6.54	6.11	6.05							
Time to 10m		62	66	55							
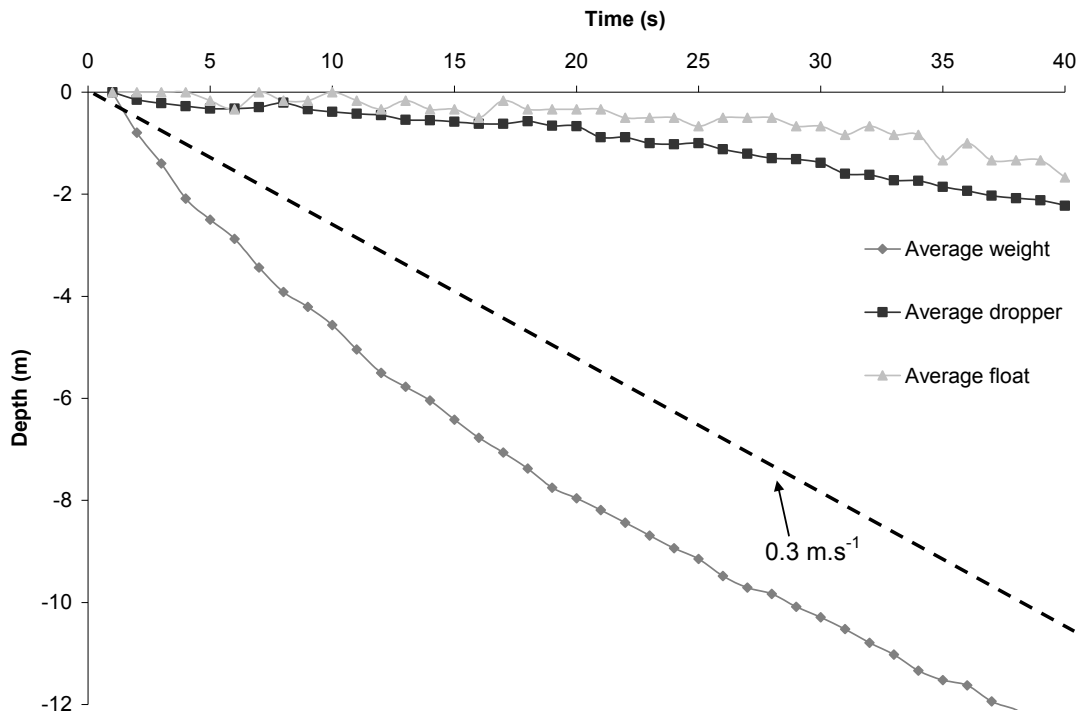
Figure 2: Average line sink rates achieved by weights (6 kg) droppers and floats in the South African demersal longline fishery. The dashed line represents 0.3 m.s^{-1} recommended in permit conditions for this fishery.
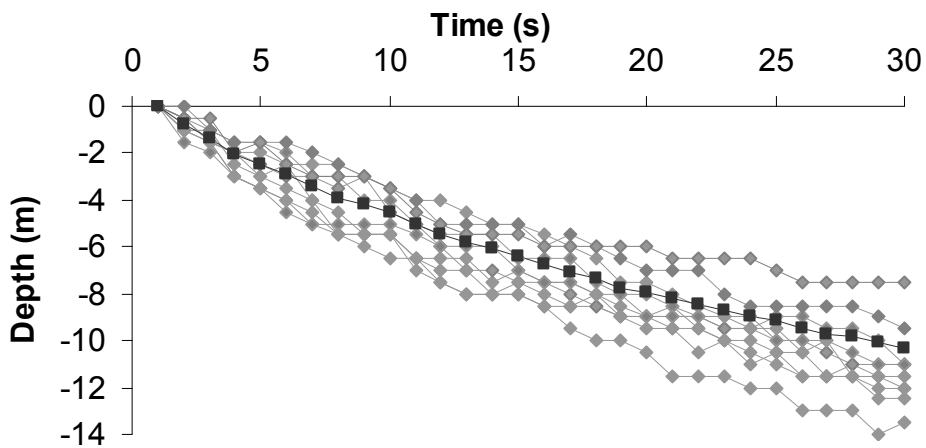

Figure 3: The mean line for 6 kg weighting (black) and a random sample (n=10) (grey) of the 41 individual line trajectories.

Table 2: Summary of standard deviation of sink rates to 2 m, 5 m, 10 m and 15 m, for 4 kg, 6 kg and 8 kg weights and associated droppers.

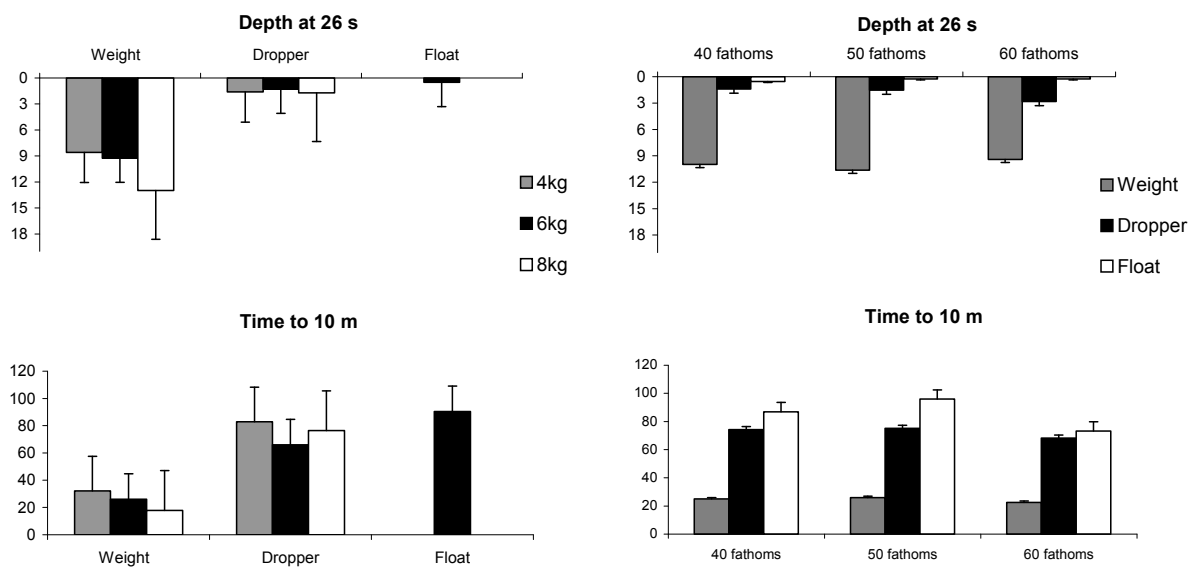| Section of line	Mass	n	2 m	5 m	10 m	15 m
Dropper	4 kg	18	0.22	0.05	0.04	0.03
Dropper	6 kg	56	0.57	0.04	0.07	0.03
Dropper	8 kg	9	0.02	0.02	0.03	0.03
Weight	4 kg	15	0.40	0.23	0.13	0.08
Weight	6 kg	54	0.28	0.15	0.10	0.08
Weight	8 kg	17	0.28	0.18	0.11	0.22

Figure 4: Average depth at 26 s and time to 10 m described by position, mass of weight and spacing.

The effect of weighting on catch composition

There was no consistent difference in catch composition between the dropper and float locations (p=0.43). Their data were therefore pooled. There was no significant difference in the catch rate of hake between the dropper (84.7/1000 hooks) and the weight (100.5/1000 hooks) for either the matched pairs *t*-test (t_{48}=−1.63, p=0.11) or the one-sample *t*-test of the relative change between the catch rate on the dropper and the weight (t_{48}=−2.17, p=0.04)

(Table 2, Fig. 5). However, more Kingklip were caught near weights (1.97/1000 hooks) than near droppers (0.39/1000 hooks) ($t_{48}=-2.55$, p=0.01 and $t_{48}=05.27$, p<0.001) (Table 2, Fig. 5).

All three of the most commonly caught demersal shark species (Short-spine Spiny Dogfish *Squalus mitsukurii*, Yellow-spotted Catshark *Scyliorhinus capensis* and Izak Catshark *Holohalaelurus regani*) and the most commonly caught skate (Biscuit Skate *Raja straelenii*) were caught at higher catch rates (17.35, 3.11, 3.86 and 4.17/1000 hooks respectively) near weights compared to near droppers (9.28, 0.78, 0.81 and 1.04/1000 hooks respectively; ($t_{48}=-3.56$, -4.04, -1.51 and -3.57, p<0.001, <0.001, 0.14, <0.001 respectively, representing a relative increase of 1.87, 4.00, 4.76 and 3.99 times that on the dropper ($t_{48}=-5.99$, -4.02, -4.55, -4.74, for all p<0.001).
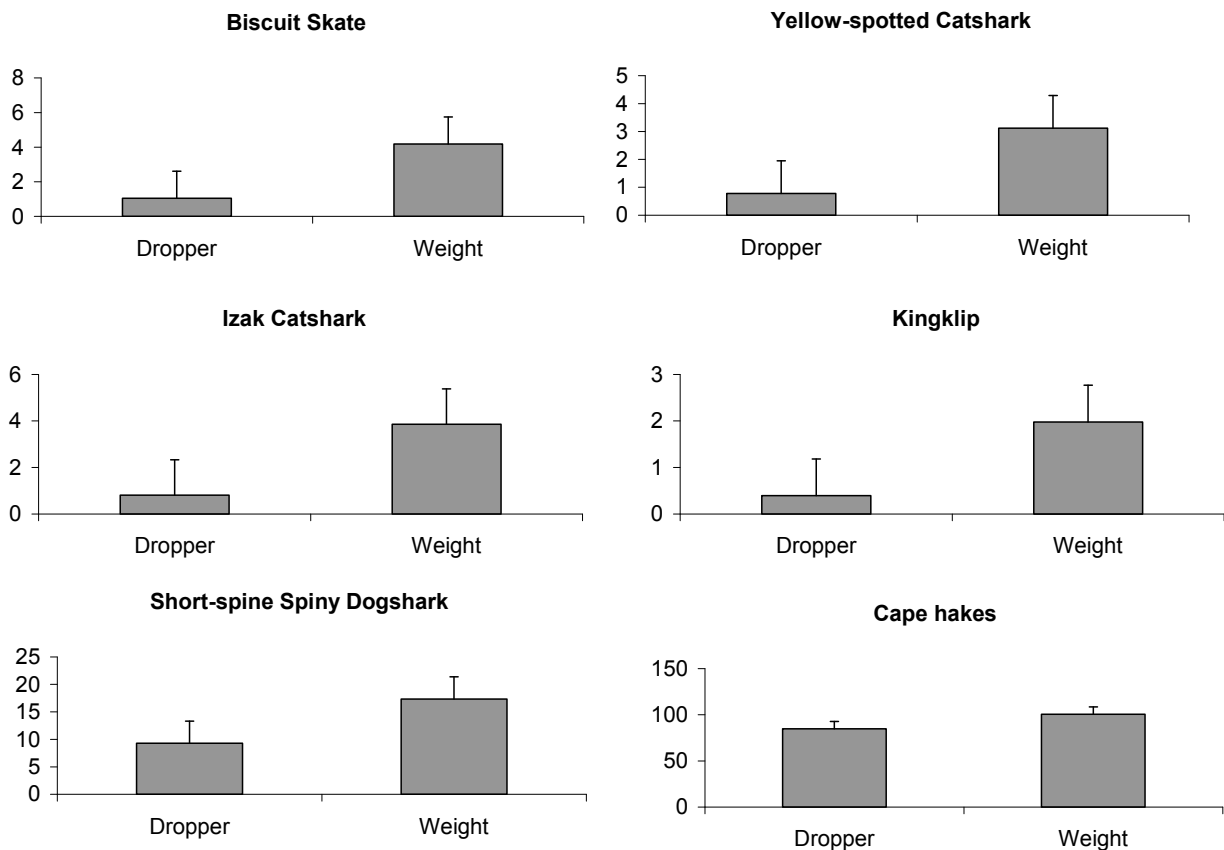

Figure 5: Average catch rates on the dropper and weighted portions of the line for six species namely; Biscuit Skate, Yellow-spotted Catshark, Izak Catshark, Kingklip, Short-spine Spiny Dogshark and Cape hakes.

DISCUSSION

The effect of mass and spacing on line sink rates

The most striking finding of this study was the marked difference in line sink rates between sections of the line close to weights and those close to droppers. The line appeared to behave the same for all models (sink rate to 5 m, 10 m, 15 m and depth at 26 s) except in the case of the sink rate to 2 m for which less variance could be explained and more variation was observed. Sections of the line near the dropper and floats remained within 2 m of the surface for 30–35 s. This is most likely due to the affect of turbulence caused by the propeller in the initial 2 m (Robertson 2000, Robertson *et al.* 2003, Moreno *et al.* 2006). This means that baits are easily accessible at a distance of 123–143 m astern the vessel if set at 8 knot (4.1 m.s^{-1}), which is a real danger to surface feeding seabirds like albatrosses. Seabird mortality could be reduced by decreasing the number of hooks near the floats as described by Seco Pon *et al.* 2007.

Hooks close to weights sank at least at the recommended rate of 0.3 m.s^{-1}, whereas those near droppers sank much more slowly, seldom attaining more than 0.13 m.s^{-1} to 10 m. Increasing line weighting increased sink rates significantly, but hooks close to droppers still sank at <0.15 m.s^{-1}. Surprisingly, reducing the spacing between weights had no effect on average sink rate. This is contrary to the findings of Robertson *et al.* (2007), who found that the distance between weights, mass of weights and setting speed all affected the sink rate of Spanish system longlines. Robertson *et al.* (2007) also found that increasing the mass of weights increased the sink rate independently of changes in setting speed and distance between the weights. When comparing the line sink rates recorded for various setting speeds (6–10 knots) and distance between weights (30–50 m), the distance between weights was the more important and shortening the distances between weights resulted in a faster sink rate (Robertson 2000, Robertson *et al.* 2007). It is possible that the distance between the weights in the present study was too great to affect the average sink rate. At present, only the weighted portion of the line sinks at the desired rate of 0.3 m.s^{-1}. Increasing the mass of the weight to 8 kg and the spacing to 40 fathoms were insufficient to achieve optimal line sink rates throughout the line.

The effect of weighting on catch composition

Results from this study indicate that although further reducing the spacing between weights and thus increasing the total number of weights on the line is unlikely to have an effect on the target species, it is likely to increase catches of Kingklip, the three of the most commonly caught demersal shark species (Short-spine Spiny Dogfish, Yellow-spotted Catshark and Izak Catshark) and the most commonly caught skate (Biscuit Skate), because catch rates of these species were 2–5 times greater close to the weight than the dropper. All these species occur on the seafloor (Payne 1977, Compagno *et al.* 1991, Ebert *et al.* 1991, 1996,

Compagno 2000, Richardson *et al.* 2000), and thus are more likely to be caught on hooks set near weights. A similar finding was reported in the longline fishery for Hake *M. merluccius* in southern Portugal where catch rates of sharks (e.g. Black-mouth Catshark *Galeus melastomus*) were significantly higher near the weight, whereas the catch of hake was higher on the mid portion of the line (Coelho *et al.* 2003). This study found that shark catches could be reduced by reducing the numbers of hooks close to the weight (Coelho *et al.* 2003). This measure could be considered for the South African fishery since this level of capture may have undesirable consequences for these species which are known to be either overexploited (in the case of Kingklip) (Japp 1993) or vulnerable because of limited geographical distribution and a K-selected life history (in the case of the sharks and skate). This could be investigated by conducting an experiment which alternates a section of line with 10 hooks removed on either side of the weight with a section where hooks were not removed and catches compared. It is conceivable that kingklip or sharks could move up the line following the bait. Such an experiment would help resolve this question. The loss of the total number of hooks could be compensated for by adding further sections to the line.

Conclusions

It is questionable whether it is practical to increase the mass of concrete weights further because of the limited space available on these vessels (Robertson *et al.* in press). To test greater masses one would need to use lead or steel weights which have a lower surface to volume ratio (Daugherty *et al.* 1989). Concrete weights are cheap and easy to manufacture (M. Pimento pers. comm.). However they require a large volume to obtain a mass of 8 kg or greater and thus utilize a large amount of space. This high mass to volume ratio also results in a reduced mass in the water (Daugherty *et al.* 1989). For instance a concrete block weighing 8 kg in air only weighs 6 kg in seawater. They also erode quickly as they knock against rocks etc. on the seabed (Robertson *et al.* 2007). A comparative study using steel or lead weights is recommended. These materials are superior because they do not degrade as easily and thus retain their mass (Robertson *et al.* 2007).

Integrated weighted line has been tested in Patagonian Toothfish (Robertson *et al.* 2003, 2006, Delord *et al.* 2005) fisheries elsewhere and found to increase line sink rates. Robertson *et al.* (2006) suggested the use of integrated weighted line (line with strands of lead woven into the rope) which sank at a rate of 0.2 m.s^{-1} to 2 m and 0.24 m.s^{-1} to 20 m and decreased White-chinned Petrel mortality by 98.7% and 93.5%, in 2002 and 2003, respectively. However integrated weighted line is likely to increase the portion of the line on the seafloor, probably increasing the bycatch of vulnerable fish, shark and skate species and is therefore unlikely to be a solution to increasing sink rates in this fishery. A compromise may be to place a weight and a float on a 10 m line at the point of dropper line attachment (G. Robertson pers. comm.). The line would sink rapidly to 10 m, out of the reach of the most vulnerable birds, while the float ultimately keeps the line off the seabed, out of the depth range of vulnerable fish, sharks and skates.

This study demonstrates the necessity to evaluate the ecosystem as a whole and not focus on a single species or taxon group when managing a fishery. The relative costs and benefits need to be weighed up and trade-offs made in order to conserve the most vulnerable species without compromising other such species and the economic viability of a fishery. By reducing the economic viability of a fishery there will be reduced incentive to comply and thus the measure will not be effective. This study indicates that, while the target species is unlikely to be affected by increased weighting, other vulnerable species of fish, shark and skate may be affected and need to be borne in mind for future trials.

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Synthesis and way forward
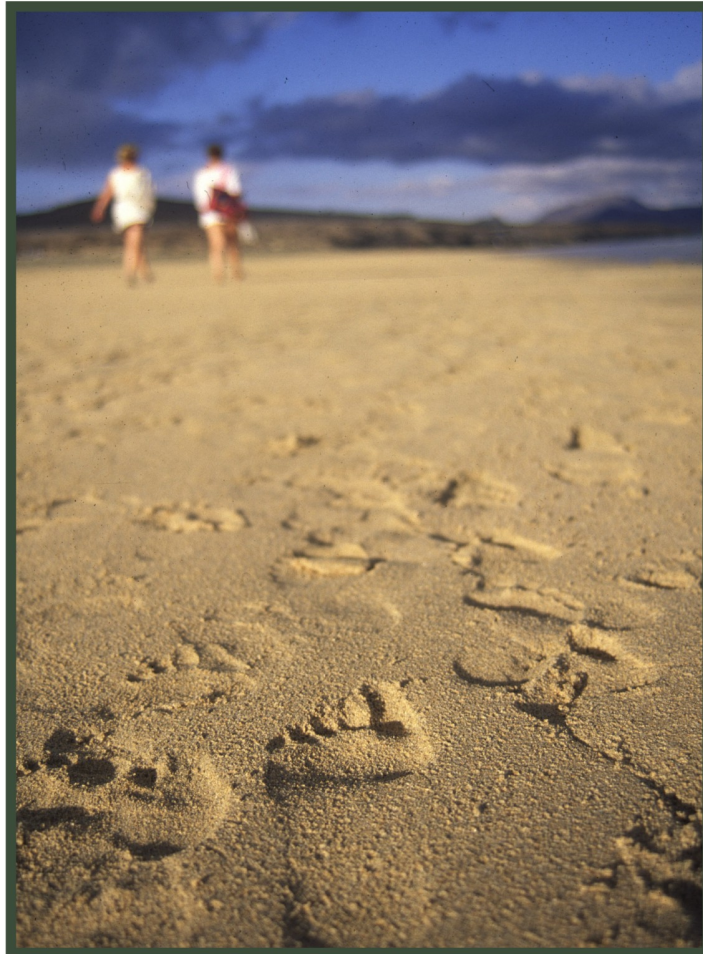
4. SYNTHESIS: OVERLAPS, INTERACTIONS AND TRADE OFFS

IMPERFECT REALITIES

This work relied heavily on fisheries observer data that were not always representative or accurate. Fisheries information on non-target species was frequently poorly collected, recorded and maintained. Although every effort was made to verify information, including cross checking against original reports, interviewing observers and ground truthing data, some of the information used in this thesis has short-comings. The largest bias was towards data collected during shorter, inshore trips. South African flagged vessels are required to carry observers on one in five trips. This had cost implications for the fishery because a crew member was replaced with an observer. Observer trips therefore tended to be of shorter duration and fishing took place closer inshore than on trips without observers. This was not the case for the Asian flagged large pelagic fleet after 2004 because this fishery had 100% observer coverage. However, prior to 2004 very few observers were placed on Asian vessels due to logistical constraints (Ryan *et al*. 2002). Catch rates calculated from observed sets, where compliance to regulations was likely to have been higher than unobserved sets, has the implication that catch rates were likely to be higher on unobserved sets (Gales *et al*. 1998) and may have resulted in an underestimation of total bycatch.

A further limitation of these data is the relatively small sample size (7–10% observer coverage). Nevertheless, these data sets incorporated information collected over 6 000 sea days and observations from 4 843 longline sets and 1 171 trawls, a substantial improvement on previous estimates. Another limitation is the poor species identification, which was illustrated when birds were returned to port and species identification could be confirmed (Chapter 1). Only 60% of birds returned to port were correctly identified to species, and 71% correctly identified to the grouping 'albatross', 'petrel' or 'shearwater'. Although the same verification could not be made for turtles or sharks, similar misidentifications are likely to have occurred. South African fisheries observers were not formally required to record bycatch data until 2002, and only received formal identification training since 2004.

Finally, fisheries observers frequently worked under arduous conditions and were expected to collect large volumes of data regarding fishing operations, target catches and non-target catches. Non-target species were often discarded, frequently without being hauled on board. Such catches were easily overlooked by fisheries observers, whose priority was to collect information on target stocks. Animals dislodged from the line prior to hauling also were not recorded and thus not incorporated in estimates of bycatch. Thus although the data used in this report has limitations they are most likely to result in conservative estimates of total bycatch.

EXTRAPOLATING ESTIMATES: ARE THE NUMBERS CREDIBLE?

There is no standard method of estimating total bycatch from observer data (Lewison & Crowder 2003). There is a large proportion of sets where no vulnerable bycatch species are caught and this frequently complicates extrapolations of bycatch (Lewison & Crowder 2003). Pennington (1983) overcame this problem by only using sets where bycatch took place. Categorical and Regression Tree Analysis (CART) has also been used to estimate seabird bycatch (NMFS 2000). This method uses continuous predictor variables to identify non-overlapping groups (Breiman *et al*. 1984). A number of studies (Klaer & Polacheck 1997, NPFMC 2001, Stehn *et al*. 2001, Lewison & Crowder 2003) used a methodology of calculating total catch for each species by summing the extrapolated catch stratified by various strata such as area, target catch and fleet based on observed catch rates and raised to total fishing effort.

The Pennington method was not used in this study because there were adequate samples of sets with bycatch. The CART method was not used because the data were so complex that the algorithm frequently failed to converge and results were not readily interpretable. A method of calculating total catch for each species by summing the extrapolated catch by area, season and flag was finally chosen as the most appropriate for the data set, as done for example by Lewison and Crowder (2003). Generalised linear models performed adequately for all analyses and were used to identify significant predictors of bycatch which were in turn used to identify appropriate strata for stratification. Explanatory variables were included in stratifications with increasing complexity. There was surprisingly little difference between complex stratifications and simple extrapolation of mean rate for seabirds given the importance of flag and season (Table 1). However, for turtles and sharks there was a decrease with increasing complexity in extrapolation technique indicating the importance of taking these factors into consideration when extrapolating total mortality for these species (Table 1). Extrapolations of total seabird mortality in South African longline fisheries have decreased dramatically from previous estimates of approximately 25 000 killed per year in 2000 (Ryan *et al*. 2002) compared to estimates calculated in this report of 6 000 killed in the same year based on a larger sample size. Because bycatch data have a large proportion of zeros, small sample sizes are inherently noisy and trends are likely to be masked by the variability in the data. Thus it is suspected that a similar pattern may occur in the Namibian hake fishery, where once more data are collected the estimate for seabird bycatch may decrease from the present estimate of 16 700 birds per year. Nevertheless the precautionary approach justifies immediate action.

In summary, total estimates of bycatch in this study are likely to be underestimates and contain an unquantified degree of uncertainty. They are however, based on the best available information. We present an approximation of mortality levels that may be used as a reference point against which we could measure actions to reduce this largely unnecessary mortality.

Table 1: Summary of estimated total seabird, turtle and shark mortality in the South African large pelagic longline fishery using a basic extrapolation of a single catch rate applied to the total effort, an extrapolation by flag and an extrapolation stratified by 5° grid cell, season and flag, 1998–2005.

Species group	Year	Basic	Extrapolation by flag			Extrapolation by flag, area and season		
		Total	Asian	SA	Total	Asian	SA	Total
Seabirds	Average	2 870	2 649	306	2 955	2 591	299	2 890
	min	1 073	0	44	561	0	44	547
	max	5 574	5 807	611	6 047	5 885	597	6 221
Turtles	Average	566	197	63	260	33	156	189
	min	23	0	0	19	0	19	68
	max	1246	899	308	1 030	74	335	344
Sharks	Average	73 457	17 318	21 883	39 202	27 375	15 729	43 104
	min	15 245	0	2 550	2 550	0	2 773	20 318
	max	164 500	53 125	50 679	70 020	70 000	29 675	77 324

ARE CATCHES SUSTAINABLE?

A decreasing seabird catch rate was observed in the pelagic and demersal longline fishery. Some of the apparent decrease in catch rate is likely to reflect reduced numbers of birds at sea, as a result of ongoing population decreases in several key species (BirdLife International 2007, Chapter 1 and 4). Seabird mortality in the demersal trawl fishery was substantial and given the conservation status of species caught, is cause for concern (Chapter 6).

Total turtle mortality in the pelagic longline fishery is considerably lower than reported in other pelagic longline fisheries (reviewed in Table 5, Chapter 2). Nevertheless, all the species caught are Endangered and in the case of the Leatherback Turtle *Dermochelys coriacea*, Critically Endangered. Loggerhead and Leatherback Turtles populations have, since 1980, failed to sustain population growth rates expected from a recovering population, especially the Leatherback population. This study highlights the significant overlap of fishing effort and Leatherback Turtle movements, suggesting that at sea mortality may be the cause of lower than expected recovery rates.

The catch per unit effort (CPUE) of Blue Sharks and Short-finned Mako Sharks decreased from 2001 and 2000 respectively (Chapter 3). A decreasing CPUE is generally indicative of a decreasing population (Seber 1982). This was accompanied by a decrease in length frequency for both species, 2002–2007, which is further evidence of an over-fished population.

Two of the four most commonly caught demersal shark and skate species in the demersal longline fishery show a decreasing abundance index: *Scyliorhinus capensis* decreased by 44% on the west coast of South Africa and by 50% on the south coast and that of *Raja straeleni* decreased by 69% on the west coast and by 65% on the south coast from 2000 to 2007.   Management measures to reduce this level of mortality are recommended (Chapter 5).

HOLISTIC ASSESSMENT OF MITIGATION MEASURES

It is important that management measures to reduce bycatch do not unintentionally result in an impact on other vulnerable species. It is thus necessary to evaluate the ecosystem as a whole and not to focus on a single species or taxonomic group. The short-comings of taking a limited view are highlighted in a number of examples in this report where a single-species group approach may have resulted in detrimental effects on another group of vulnerable species. For example, various weighting regimes were investigated in the demersal longline fishery to increase line sink rates as a mechanism to reduce seabird mortality. We found that although catches of hake were unlikely to be reduced by increased weighting, other vulnerable species of fish, shark and skate may be affected (Chapter 12). Given that relatively few birds were caught in this fishery off South Africa, the increased impact on non-target fish species may outweigh the potential benefits of increased weighting on reduced seabird bycatch. In Chapter 10 the use of circle hooks was considered to reduce turtle capture in the pelagic longline fishery. Too few turtles were caught to demonstrate a benefit, but observations suggested that circle hooks may have increased shark catches. However, bait type also significantly affected catch rates of sharks and explained more variance than hook type, so this may simply have been an artefact of a small sample size. Implementing circle hooks could be counter-productive if reducing shark catches was a management objective of this fishery.

The relative costs and benefits of mitigation options need to be weighed up and trade-offs made in order to conserve the most vulnerable species without compromising other species. However it is equally important to consider the economic viability of a fishery. By reducing the economic viability of a fishery there will be reduced incentive to comply, thus reducing the efficacy of a measure. An example here is the effect of limiting setting in the Swordfish directed fishery to between nautical dusk and dawn. Chapter 8 confirmed that Swordfish catch rates were the highest when setting took place at dusk. However, the time of setting is an important indicator of sea bird mortality. Setting lines at night is a widely implemented management measure to reduce the capture of albatrosses, but this should be understood in the context of the fishery.   Understanding these interactions assists fisheries managers to make the appropriate compromises.

THE USE OF TIME AND AREA CLOSURES FOR MANAGING VULNERABLE BYCATCH

Closed areas or marine protected areas have not been widely used as a fishery management tool mainly because fisheries managers have tended to focus their attention on abundant species with high biological productivity. However, closed areas are an important management tool for conserving less abundant species with low biological productivity such as seabirds, turtles and sharks (Halpern & Warner 2002, Worm *et al*. 2003, Walker 2004, Alpine & Hobday 2007). The use of marine protected areas is increasingly important in fisheries management and for the conservation of marine ecosystems (Halpern & Warner 2002, Gell & Roberts 2003, Hilborn *et al*. 2004, Roberts *et al*. 2005). However, their application in open ocean systems remains limited (Mills and Carlton 1998, Worm *et al*. 2003, Alpine & Hobday 2007). The question remains whether we can use marine protected areas to reduce bycatch? This was investigated by evaluating the spatial and temporal overlaps of the catches of seabird, turtles and sharks in the pelagic longline fishery.

Temporal overlaps

There was little overlap in the seasonal distribution of catch rates among seabirds, turtles and sharks off southern Africa (Fig. 1). Overall, seabird catch rates were the greatest in the austral winter (May–October) (Chapter 1), turtle catch rates over the summer months (January–June, all year for the Critically Endangered Leatherback Turtle *Dermochelys coriacea*) (IUCN 2007) (Chapter 2) and shark catch rates in the latter half of the year (June–December) (Chapter 3). A single closed season is therefore unlikely to adequately protect these groups of species, although this does not and should not eliminate the possibility of a closed season to protect a specific species or group of species, especially if it is particularly endangered or vulnerable at certain stages of its life history. This would have the added advantage of protecting other vulnerable groups incidentally.
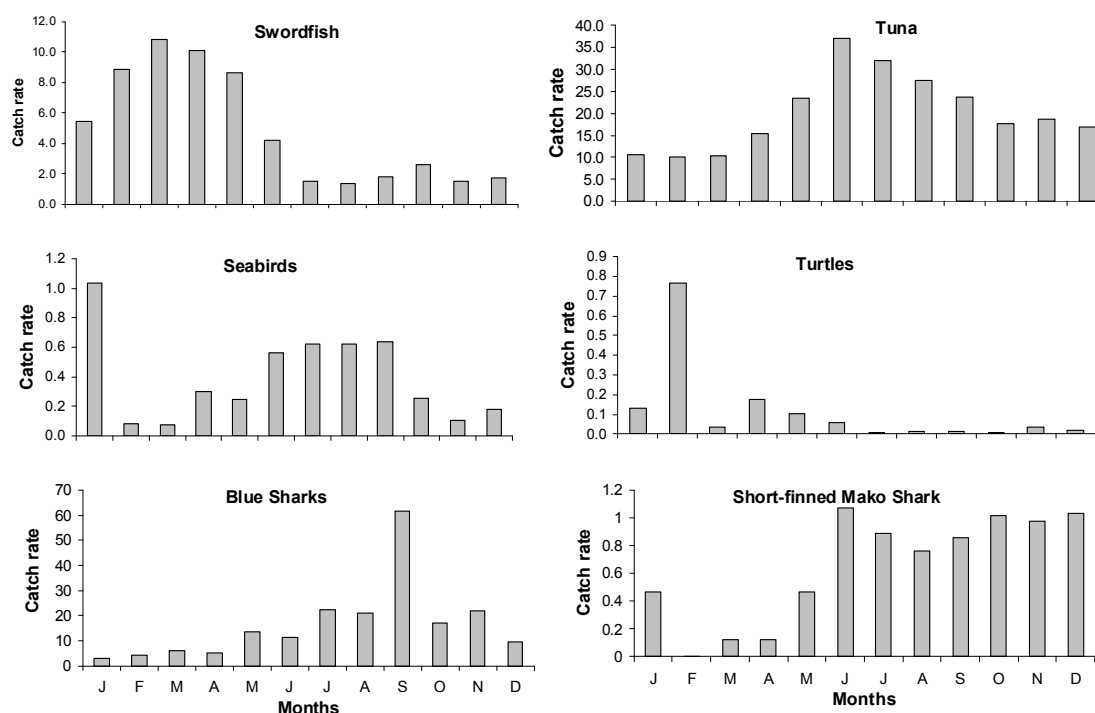
Figure 1/…

Figure 1: Average monthly catch rates (per 1000 hooks) of Swordfish, tuna, seabirds, turtles, Blue and Short-finned Mako Sharks, 1998–2005 in the pelagic longline fishery.

Spatial overlaps

Seabird catch rates were highest on the Agulhas Bank, off Cape Point and on the continental shelf edge in water depths of between 500 m and 1 000 m (Fig. 2, Chapter 1). Loggerhead Turtles *Caretta caretta* were predominantly caught on the Walvis Ridge in the south eastern Atlantic, whereas Leatherback Turtles were caught throughout the region and revealed no statistically significant trend (Chapter 2). In the case of Blue *Prionace glauca* and Short-finned Mako Sharks *Isurus oxyrinchus* catches were the highest along the west coast, in the vicinity of Cape Point, and on the Agulhas Bank (Fig. 2). Blue Sharks were predominantly caught in deeper water (average depth of 1 000 m) compared to Short-finned Mako Sharks which were predominantly caught further inshore in water on average 200 m deep (Fig. 2, Chapter 3).
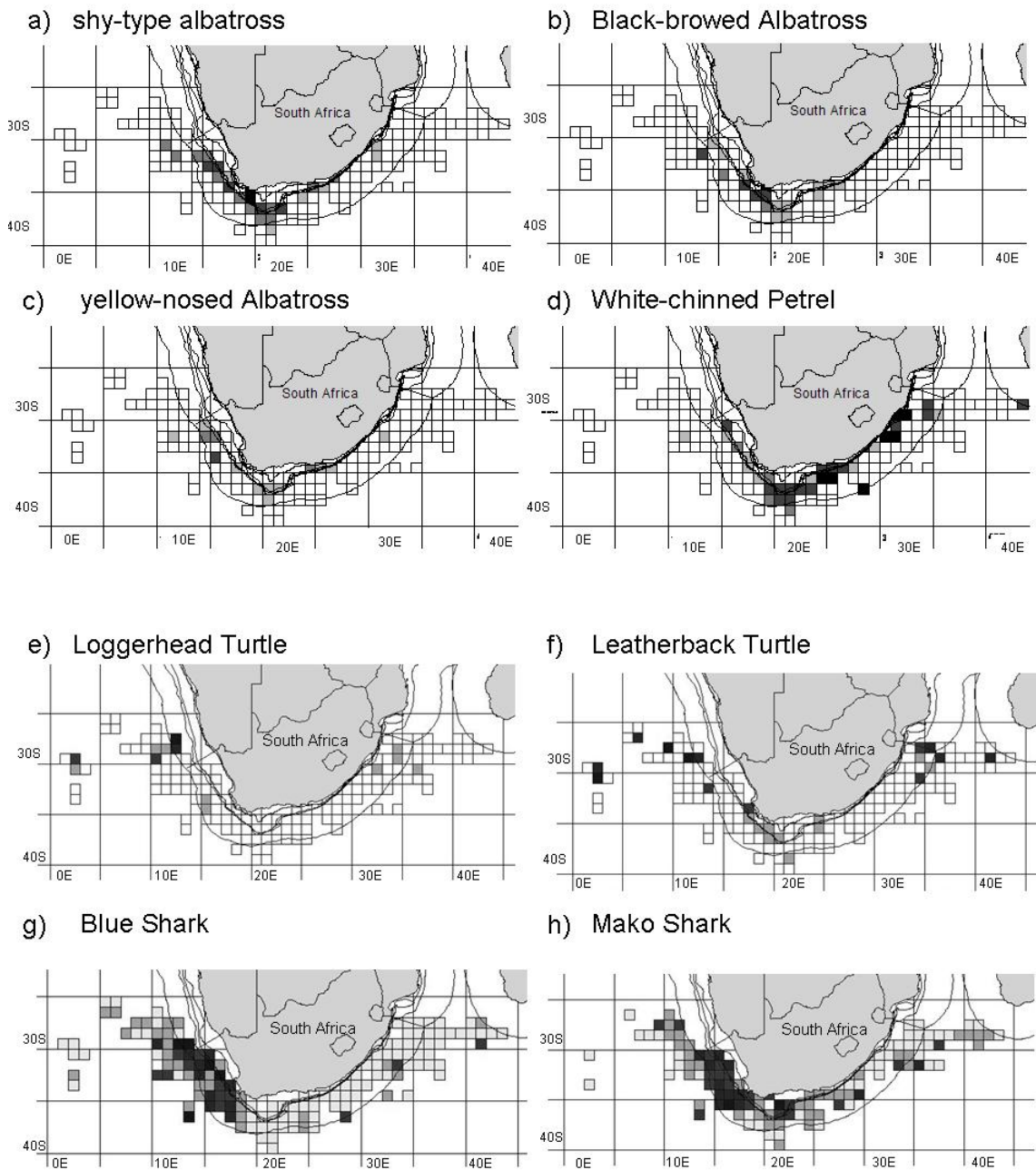
Figure 2/…

Figure 2: Spatial distribution of bycatch catch rates summarized by 1° grid cell, 1998–2005.

Using MARXAN, conservation planning software we identified areas that through closure of fishing could reduce the bycatch of seabirds, turtles and sharks with the least impact on the commercial catches of tunas *Thunnus* spp and Swordfish *Xiphias gladius*. Conservation targets were set to incorporate 10%, 30% and 50% of average annual total catch for Black-browed Albatrosses, shy-type albatrosses, White-chinned Petrels, Loggerhead Turtles, Leatherback Turtles, Blue Sharks and Short-finned Mako Sharks using observer data, 1998–

2005. MARXAN (Ball & Possingham 2000, Possingham *et al*. 2000) provided a near optimal solution to the minimum set problem by minimizing a multivariate objective function. The objective function incorporated a penalty for areas where target catches were highest and investigated the effect of mimimising the boundary length or reducing fragmentation of areas (Steward & Possingham 2005). The analysis was confined to South Africa's Exclusive Economic Zone (EEZ) which was divided into 65 1° grid cell planning units. MARXAN was run with one million iterations and runs were repeated 1 000 times to find both the best solution and the summed solution. The best solution identified a set of planning units that achieved conservation targets while minimizing the cost, which in this case minimized the loss of commercial catch to the industry. The summed solution was the selection frequency of a particular planning unit across all 1 000 runs and maybe used as an indication of conservation value. A value of 798, for example, indicated that it was selected 798 out of 1 000 runs.

There was considerable overlap of all three groups and fishing effort on the continental shelf edge, especially on the west coast and in the Cape Point and Agulhas Bank regions. Similarly the areas selected by MARXAN analysis converged near Cape Point with the exception of the 10% target (Fig. 4). The best solution protected areas included 12–34% of the EEZ and 13–37% of fishing grounds when little limitation was placed on the boundary length and 9–28% of the EEZ and 10–30% of fishing grounds when boundary length was minimized (Table 2). Similar proportions were found in a study off the east of Australia where protected area requirements ranged from 7 to 26% (Alpine & Hobday 2007). Limiting the boundary length and reducing fragmentation has implications for protected area management and allows for the most efficient use of resources for monitoring, compliance and surveillance (Alpine & Hobday 2007).

Table 2: MARXAN outputs expressed as a percentage of the South Africa Exclusive Economic Zone and the large pelagic longline fishing area based on observer data, 1998–2005.

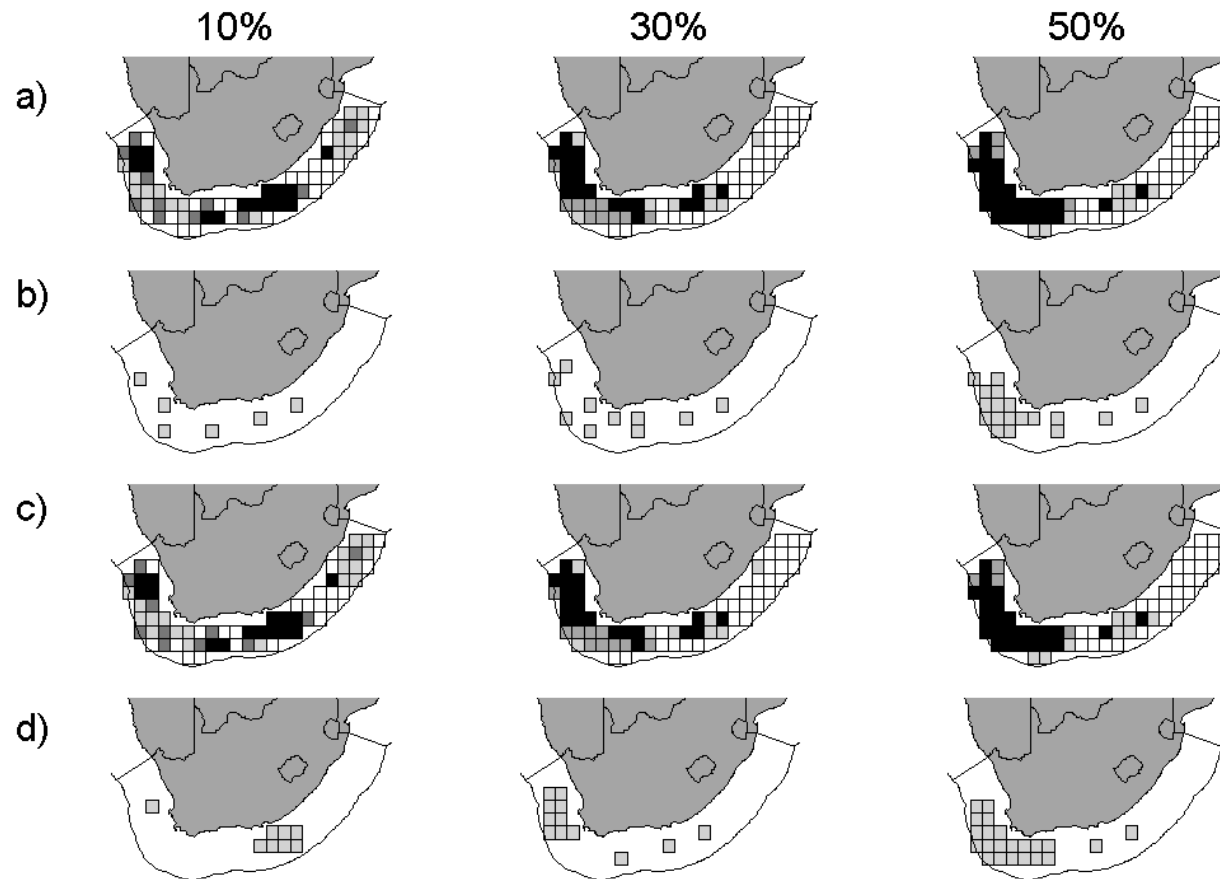| % of EEZ	Conservation target		
	10%	**30%**	**50%**
No limit on boundary length	9%	15%	28%
Boundary length minimised	12%	18%	34%
% of Fishing area			
No limit on boundary length	10%	17%	30%
Boundary length minimised	13%	20%	37%
Relative increase with increasing conservation target			
No limit on boundary length	-	6%	12%
Boundary length minimised	-	6%	15%

Figure 3: Maps depicting MARXAN outputs of 10%, 30% and 50% of average seabird, turtle and shark catches per 1° grid cell indicating a) summed solution (without minimizing boundary length) b) best solution (without minimizing boundary length) c) summed solution (limiting boundary length) d) best solution (limiting boundary length). The shading in a) and c) indicate the relative importance of grid cells, where important increases from light to dark.

If implemented, the 30% and 50% best solution areas in the vicinity of Cape Point would have economic consequences for the large pelagic longline fishery for at least two reasons. Firstly, Cape Town is the most frequently used port for this fishery and implementation would therefore increase their commuting distance to fishing grounds. Secondly, this area is not only important fishing grounds for the large pelagic fishery, but also for a number of other fisheries including demersal longline and trawl fisheries for hake. These fisheries are also known to catch a similar suite of species, although because of the depth range they operate in, no turtles are caught and mainly demersal, rather than pelagic sharks are caught (Barnes *et al*. 1997, Walmsley *et al*. 2007, Chapter 5 and 6). The 10% area on the other hand is likely to be more economically viable although it begs the question of whether only protecting 10% of the area where mortality occurs is adequate.

It is important to bear in mind that the data used in this analysis are unlikely to be representative of the entire EEZ and are positively biased towards fishing areas. This short-coming is probably minor as only five of 65 planning units contained no data. However, important areas for all three bycatch groups may exist outside of fishing areas and may not be captured by this analysis. The spatial distribution of some species, especially seabirds, is likely to be altered by the distribution of fishing effort. Indeed, the presence of fisheries has been identified as an important determinant of seabird distribution at sea (Wahl & Heinemann 1979, Ryan & Moloney 1988, Furness *et al*. 1992, Garthe 1997, Votier *et al*. 2004). The result of satellite tracking and GLM analysis in South African waters suggested that the presence of trawlers was a strong predictor of both shy-type and Black-browed Albatross distribution (Chapter 7). This raises questions about the validity of closed areas alone for the protection of these two species and scavenging seabirds in general, which may simply follow fishing vessels out of a closed area. This illustrates that displacing fishing effort may not reduce the capture of all vulnerable species (Botsford *et al*. 2003).

Although seabirds, turtles, sharks, and pelagic fish (e.g. tunas and Swordfish) are highly mobile groups of species, they are not dispersed randomly or uniformly throughout the ocean and aggregate in areas of high productivity (Piontkovski *et al*. 1995, Pakhomov & McQuaid 1996, Ansorge *et al*. 1999, Polivina *et al*. 2004, Chapter 7). It is this behaviour that makes species, in this case tuna and Swordfish, exploitable at economically viable levels (Brill & Lutcavage 2001, Norse 2006). Similar concerns have been voiced elsewhere about the usefulness of closed areas to protect pelagic species in general, based on the fact that open ocean systems are highly dynamic and adequate protection of its biological and physical components would require vast areas to be closed to fishing which may have undesirable economic consequences (Angel 1993, Horwood *et al*. 1998, Boersma & Parrish 1999, Botsford *et al*. 2003, Alpine & Hobday 2007). Alpine and Hobday (2007) examined this perception by modelling the oceanic region off eastern Australia and considered a set of biological, physical and social attributes. This study concluded that the area requirements of oceanic protected areas should not be an impediment to their consideration and implementation (Alpine & Hobday 2007).

In conclusion, the use of spatial management holds promise for the protection and management of vulnerable bycatch species and is likely to have far-reaching benefits, such as wider ecosystem protection. Notwithstanding this, it is unlikely that a single measure will curb the magnitude of mortality currently experienced in this fishery, at least in the short term. It should therefore be seen as one of a suite of measures available to fisheries managers to address the bycatch of vulnerable species. This study also demonstrates how decision support software such as MARXAN can be used to identify possible candidate areas for closures. Improving our ability to monitor and predict the movement of vulnerable species will increase our ability to use protected areas for their management in the future.

ECOSYSTEM BASED MANAGEMENT: A BALANCING ACT

Resource managers are often faced with the problem of competing interests (Hildén 1997). Balancing the apparently conflicting objectives of job creation and poverty alleviation, against the commitment to implement an Ecosystems Approach to Fisheries Management (EAF), is a key challenge facing southern African fisheries managers. Ultimately an EAF aims to ensure the health of the ecosystem, which underpins fishing production and livelihoods, by catering for human and the ecosystem well-being collectively and therefore ultimately contributes to job security (Reykjavik Declaration 2001, FAO 2003, Pikitch *et al*. 2004, Shannon *et al*. 2004, Morishita 2008). Implementing an EAF in southern African longline and trawl fisheries involves the integration of social, economic, and ecological goals. This is only achievable if one set of goals does not dominate at the expense of the others and win-win situations are sought (Meffe *et al*. 2002). These win-win situations require compromise. In the past, many failures in resource management have been blamed on the inability to balance social, economic, and ecological objectives (Keough & Blahna 2006). The long-term benefits of successfully implementing integrative, collaborative ecosystem management outweigh the short-term difficulties associated with such efforts.

Win-win is first prize: a seabird example

In my opinion, the best example of a win-win situation is that of the incidental mortality of seabirds. Generally fishers do not want to catch seabirds as no market exists for them and there is considerable public concern about the issue. Many seabirds affected by fisheries, are opportunistic scavengers and as such are adapted to foraging around fishing vessels and frequently steal bait without getting hooked. Seabirds returned to port by South African observers have been recorded to have up to 16 pieces of bait in their stomachs, suggesting that they potentially are stealing large quantities of bait from longlines (Nel *et al*. 2002). Each hook set without bait is one less opportunity to catch fish and therefore it is in the economic interest for fishers to limit the availability of baited hooks to seabirds. Good data exist both in terms of understanding the causal factors of seabird mortality as well as good population trend information for seabirds on breeding islands. Coupled with this, cost-effective, practical

mitigation measures have been identified and demonstrated that good compliance can virtually eliminate seabird bycatch and, by implication, increase the economic viability of the fishery.

The successful reduction of seabird mortality through the use of mitigating measures has been demonstrated by CCAMLR (Kock 2001, Nel *et al*. 2002). Sadly, this is not the general rule. Mitigation measures to reduce seabird bycatch have been conditions of South African longline fishing permits for over a decade; yet substantial numbers of birds continue to be caught, especially in the pelagic longline sector (Chapter 1). This is largely because of poor compliance, a phenomenon not limited to South Africa.

Addressing poor compliance

The single biggest challenge facing conservationists and fisheries managers to overcome and reduce bycatch in longline and trawl fisheries in South African waters is addressing the low compliance to mitigation measures. This issue can be addressed through a suite of measures:

a) Education and awareness

Low compliance is frequently the result of a lack of understanding of the life history characteristics of seabird populations (Bergin 1997, Robertson 1998, Gilman 2001). Fishers, who are accustomed to catching less vulnerable species, perceive the relatively low catch rates of seabirds as insignificant (Robertson 1998). Seabirds are opportunistic scavengers attracted to fishing vessels as they discard fisheries waste (Brothers *et al*. 1991, 1999a,b, Bergin 1997), often in large numbers, creating the impression to fishers that seabirds are plentiful. For effective implementation of mitigation measures it is essential to educate fishers (Bergin 1997) about the fact that seabird populations are indeed declining at unsustainable rates due to their K-selected life history traits (Warham 1996, Croxall & Gales 1998, Gales 1998).

Compliance with the use of tori lines improved dramatically from virtually non-existent to approximately 51% in the longline fishery partly as a result of an education programme launched in South Africa in 2004. Lessons learnt from this programme were that fishers did not respond well to the classroom environment and the most effective learning took place at sea. At first impression this was not considered time effective because trips were at least 10 days in duration. However, experience has shown that fishers communicate frequently by radio and this proved to be a useful means of communication with multiple vessels.

b) The role of observers

Observers can play a key role in monitoring compliance. For example, observers in South African fisheries report that tori lines were only used on 51% of sets in the pelagic longline fishery and 9% in the demersal longline fishery. This has the implication that tori lines are less likely to be used when observers are not present. Compliance improved to 73% of sets in 2005 when improved observer coverage was the result of a condition placed on joint-venture vessels operating in the fleet. Similar improvements in compliance with improved observer coverage have been reported elsewhere (Gales *et al*. 1998). Even though observers are not onboard to bring about compliance, their mere presence is likely to have an effect. Increasing observer coverage is therefore likely to be important to further increase compliance.

Observers can also play a role in educating fishers and demonstrating mitigation measures at sea. Since data collected by observers are used to assess bycatch levels it is imperative that observers are adequately trained to ensure correct species identification. This was highlighted as a short-coming in this study and should be addressed in future initiatives.

c) Appropriate mitigation

Another factor resulting in low compliance is poorly defined or impractical mitigation methods. Mitigation methods should describe the method clearly and concisely. In general, fisheries regulations should be defined by gear configuration rather than by the desired outcome. This facilitates both accurate implementation and enforcement. For example, foreign flagged vessels which do not add weights to their lines are often in breach of the regulation to achieve a line sink rate of 0.3 m.s^{-1}. A compliance officer cannot enforce this regulation without the use of a time depth recorder to calculate the line sink rate. To facilitate enforcement, this permit condition should rather be defined by the gear configuration tested to achieve a desired line sink rate, for example 60–120 g weight placed 2 m from the hook (Brothers *et al*. 2001, Chapter 9). At present this is not the case for the South African pelagic longline fishery and this requires addressing.

Alternatively, vessels could be required to demonstrate that they meet the desired line sink rate prior to entering the fishery. A precedent for this exists in CCAMLR fisheries (conservation measure 24-02 (CCAMLR 2007) which require vessels to demonstrate a sink rate of 0.3 m.s^{-1} prior to commencing fishing). Line sink rate trials conducted in this report highlight that although particular gear configurations will on average achieve desired sink rates, very high variability around the average is observed. This has practical implications for a management measure requiring fishers to demonstrate line

sink rates prior to entering a fishery and needs to be borne in mind when developing protocols to implement this measure.

The main challenge increasing implementation of tori lines is decreasing the chances of entanglement. This could be achieved by designing a purpose made drogue which could guide the tori line away from longline gear while still keeping seabirds away from baited hooks.

d) Social influences and resistance to change

Ultimately, implementation of mitigation measures is dependent on the fishing industry and relies on a willing skipper on every boat, because enforcement officers cannot be everywhere all the time. Social influences such as peer pressure and resistance to change need to be understood and overcome (Dent & Goldberg 1999). The current situation in both pelagic and demersal longline fisheries of South Africa is that only a few individuals comply with all regulations. The influence of peer pressure is acting against compliance – it is not "cool" to comply. People often do things simply because other people do, so-called band-wagoning (Goidel & Shields 1994). As more individuals begin to comply, for whatever reason, the probability of other individuals doing so is increased (Goidel & Shields 1994). Peer pressure can also be used to encourage others to comply. Efforts should be made to identify "champions" within the fishing industry. In my experience, a 'champion' within the industry is likely to have more influence over his peers than a fisheries manager or conservationist.

Mitigation measures usually involve changes in gear configuration and/or fishing operations. Humans are resistant to change and fishers are no exception (Folger & Skarlicki 1999). Brothers *et al*. (1999a) reported that fishers resist changing their gear configuration. For example, they perceive that increasing the weighting of their gear may affect the characteristic of the line sinking, the attractiveness of the bait to the target species, the ability to retain the hook at the target depth, an increased cost and the possibility of negatively affecting crew safety. Resistance is an inevitable response to any major change (Folger & Skarlicki 1999). If management does not understand, accept and make an effort to overcome resistance, it can undermine even the most well-intentioned idea. A manager's ability to implement a new regulation that involves changed behaviour depends in part on how effectively they create and maintain a climate that minimises resistant behaviour (Coetsee 1999). In South Africa, we have attempted to overcome this resistance by spending time at sea on longline and trawl vessels. Time at sea demonstrates to fishers the willingness of managers or conservationists to understand the realities of life at sea and in doing so builds trust and 'street credibility'. This also allows for mitigation measures to be demonstrated under local conditions which ensures their practicability and further encourages implementation and compliance.

e) Incentives

Adhering to regulations is a combination of enforcement and voluntary compliance (Brothers *et al*. 1999a). The latter is essential because not all vessels can be inspected at all times. Thus the fate of seabirds, turtles and sharks is essentially in the hands of the fishing industry. To encourage voluntary compliance, skippers should be made aware of the conservation status of these animals. They also need to be included in the decision making processes to ensure that mitigation and management measures implemented are practical and cost-effective and have the support of the fishing industry from the start. Incentives for compliance should include increased access to rights and quota allocations for those fishing responsibly. Eco-labels such as the Marine Stewardship Council and the Sustainable Seafood Initiative can also encourage voluntary compliance (May 2003, Jacquet & Pauly 2007). As consumers become more aware of the threats to the world's oceans, so they begin to use their discretion when making the choice of which products to purchase (May 2003, Jacquet & Pauly 2007). Fishing industries which act responsibly are more likely to secure a market advantage than those who do not. The South African demersal trawl fishery has Marine Stewardship Council certification. Addressing seabird bycatch was a condition placed on this certification. The results of this study (Chapter 6) highlighted the need for mitigation. Tori lines and offal management requirements were implemented in the trawl fishery in 2006 and compliance was estimated to be 80% during the day within the two year (B. Watkins pers. comm.). This is a substantial improvement from the situation with the longline fishery where virtually no compliance for the first ten years was observed, and is largely attributable to the Marine Stewardship Council certification highlighting the role eco-labels and market driven forces can play in implementing solutions.

f) Enforcement

Some might argue that according to game theory, individuals act to maximize their own benefit, without concern for the effect on others. It especially pays to cheat if others cheat. Equally, if everyone (fishers, fisheries managers and conservationists) co-operates everyone wins. The tragedy of the commons is that unless the cost (of losing vulnerable species or payment of a fine) is incurred directly by the individual, the benefit (in this case business as usual) will outweigh and persist (Hardin 1968). It is therefore important that the cost of non-compliance and risk of being caught be sufficiently high to outweigh the benefit. At present this is not the case in southern Africa where penalties implemented for breaking bycatch related regulations are insignificant and fall short of acting as an incentive to comply. Penalties need to be brought in line with the commercial interests of the fishery to act as a disincentive. This alone is likely to greatly facilitate implementation of mitigation measures.

A reality check

The harsh reality is that although win-win situations should be sought, they occur infrequently. Even for seabirds, where there are cost-effective, practical solutions that operate in the economic interest of the fishery, seabirds continue to be killed and their populations continue to decrease (BirdLife International 2007). In South Africa, BirdLife and WWF have been running an education programme for fishers, observers and compliance officers since 2004. They have also been responsible for amending the wording of permit conditions to ensure it is easily understandable to those needing to implement them, have facilitated workshops to include skippers in the decision-making process, have provided tori lines free of charge and been to sea on numerous longline and trawl vessels to demonstrate mitigation measures. This campaign has seen an increase in compliance of the use of tori lines on longlines from almost negligible to approximately 50% in the pelagic longline fishery in the first two years, but little further increase in compliance since.

Every effort should be made to adequately understand a problem, find cost-effective, practical solutions, educate fishers, include them in decision making processes and ensure adequate incentives. Experience has shown that these will only take compliance so far, after which effective enforcement is necessary. A further option is the setting of upper precautionary catch limits, beyond which fishing should stop. Such limits should ideally be placed per vessel rather than across a fleet. In all cases 'vessel' was the best predictor of mortality. Bycatch is not evenly distributed throughout the fleet, with a handful of vessels responsible for killing the majority of seabirds, turtles and sharks, a finding consistent with Klaer and Polacheck (1998). Setting upper precautionary catch limits can therefore act to eliminate problem vessels. It can also act as an incentive for skippers to comply, because a vessel complying with regulations is unlikely to reach an appropriately set upper precautionary catch limit and thus likely to continue fishing unhindered by the limit.

In conclusion, the warning bells are not new: scientists have been sounding them for over a decade. This report has gone some way to further our understanding and identify possible solutions. What is now needed to save these species from extinction is action.

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