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# **The use of fish species in a marine conservation plan for KwaZulu-Natal**

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By

**Philip Haupt**

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**Supervisor: Dr A.T. Lombard**

**Co-supervisors: Dr P.S. Goodman, Dr K. Sink**

**Specialist advisors: Mr B. Q. Mann, Dr E. Lagabrielle, Dr J. Harris**

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## Summary

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# Use of Fish Species in a Marine Conservation Plan for KwaZulu-Natal

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This study formed part of a larger provincial marine systematic conservation plan for KwaZulu-Natal (KZN), South Africa, called SeaPLAN. Owing to budget and time constraints, not all ± 1640 fish species that occur in the region were considered. A method to prioritise species was therefore developed to identify those species which were most at most risk of being excluded by a conservation plan based primarily on habitat representation (i.e. SeaPLAN). The method was based on three underlying principles: (i) species with limited conservation options; (ii) threatened species; and (iii) inherently vulnerable species. From these three principles, seven criteria were defined (e.g. endemic or rare species). Sixty-seven species met the qualifying conditions for these criteria and were consequently included in this study (FishPLAN).

In order to map the distributions of these 67 fish species, the spatial and temporal accuracy of existing marine fish data for KZN was investigated. Only 17% of the data evaluated met the spatial resolution requirements of 1 km<sup>2</sup>, while temporal resolution was high: >99% of the data were collected at daily resolution. A resulting recommendation is that future data collection employ handheld data recording devices (with GPS capability), in order to increase the spatial accuracy of data, minimise human error and improve the efficiency of data flow.

Species life cycle envelopes (SLICES) were developed to capture spatial differences in areas occupied during three life-cycle phases (reproductive, juvenile and feeding). Two distribution modelling techniques were used: Maxent, which uses quantitative data, and CHARMS (cartographic habitat association range models), which uses qualitative range data. A combination of statistical and biological criteria was used to determine the most informative and appropriate model for each species. Species distribution models (SDMs) were constructed for three temporal partitions of the data: annual, summer and winter.

Patterns of species richness developed from the seasonal models showed seasonal differences in patterns that conformed to known seasonal distributions of fish assemblages: richness was higher in southern KZN during winter, while it was higher in northern KZN during summer.

The resulting SDMs were used to develop a conservation plan for fish: conservation targets were set using the minimum recommended baseline of 20% of a species' range, to which biological retention targets (additional proportion of the range) were added, in an attempt to ensure species persistence. The conservation targets were then adjusted using catch per unit effort (CPUE) data to match seasonal abundance of a given species.

Within the existing network of marine protected areas (MPAs), none of the species' targets are met by MPA sanctuary zones (zone As) alone, and all species require greater areas of protection. Three areas, namely offshore of the Tugela River mouth, the reefs offshore of Durban, and Aliwal Shoal, were consistently identified as being important in addition to existing MPAs for conservation of the fish species investigated. The greater efficiency of a seasonal MPA network to protect seasonally varying distributions of biodiversity, suggests that this may be a useful tool to consider in conservation management. The outcome of a conservation plan from this study (FishPLAN) was finally compared with the broader, more inclusive conservation plan, SeaPLAN. This comparison demonstrated how conservation plans based on a single group of species run the risk of identifying areas that are appropriate only for the relevant species, and might fail to conserve biodiversity as a whole.

## **Keywords**

*Fish, species prioritisation, spatio-temporal accuracy, distribution modelling conservation planning, KwaZulu-Natal.*

# **General introduction to the use of fish species in a marine conservation plan for KwaZulu- Natal**

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The Food and Agriculture Organization (FAO) report on the State of the World Fisheries and Aquaculture has shown that more than 50 % of the world's marine fish stocks are fully exploited, and 28 % are overexploited or depleted (Garibaldi et al. 2008). The urgency of immediate and effective action to conserve rapidly declining natural resources and biodiversity has been emphasized by a vast number of scientists from wide ranging fields at several international conferences (e.g. Convention of Biological Diversity (CBD), 1992; Earth Summit 1992; World Summit for Sustainable Development, Johannesburg 2002; World Parks Congress (WPC), Durban 2003; IPCC, Kyoto 1997). The outcomes of these international conservation conferences have mandated increased global conservation effort (Dirzo & Raven 2003; UNEP 2004; Garibaldi et al. 2008). In particular, the CBD set the Millennium Development Goals (MDGs) charging countries which are signatory to the treaty with the development of National Biodiversity Strategy Action Plans (NBSAPs) (CBD Article 6, UNEP 2004), where the goals were set to increase protected areas to represent 20% of all habitat types. Currently less than 1% of the marine environment under South Africa's jurisdiction, known as the Exclusive Economic Zone (EEZ), is under any formal protection (Lombard et al. 2004). South African initiatives like the National Spatial Biodiversity Assessment (NSBA) (Lombard et al. 2004) and National Protected Area Expansion Strategy (NPAES) (DEAT & SANBI 2008) have since been launched to formulate strategies to achieve national MDGs.

Marine ecosystems in South Africa have been degraded as a result of multiple factors such as, water abstraction practises from agriculture and forestry, pollution, coastal erosion, and over exploitation of marine resources (Southern African Development Community 2008). The effects of overfishing have received a lot of attention (Yeld 1992; Sink et al. 1994; Mann

2000; Chale-Matsua 2001; Anonymous 2001). Stock declines, changes in marine fish community structures and reductions in average sizes have been observed nationally in South Africa since the 1970's (Mann 2000). The KwaZulu-Natal (KZN) Province of South Africa has the highest national density of shore anglers (Brouwer et al. 1997) but also has several over-exploited endemic fish species with complex life history patterns (Garratt 1985; Garratt et al. 1994; Mann et al. 2006). Many of these species also exhibit aggregative spawning behaviour (Mann 2000; Heemstra & Heemstra 2004). Consequently, many KZN fish species have a low resilience to fishing pressure and require increased protection (see Mann 2000; Mann et al. 2006). Furthermore, marine ecosystems and the fisheries that they support are vulnerable to global climate change, and its effects (e.g. sea level rise) pose direct risks to humans and economic development (Atkinson & Clark 2005; Southern African Development Community 2008). Marine Protected Areas (MPAs) have been shown to be effective to mitigate negative impacts like fish stock declines and changes in community composition (Roberts et al. 2001; Gell et al. 2003). MPAs have also been suggested as mitigating measures for climate change (Secretariat of the Convention on Biological Diversity 2009). Although South Africa has been benefiting from MPAs since 1964 (Robinson & de Graaff 1994), the *ad hoc* allocation process of these areas is questionable in terms of its efficacy in protecting biodiversity (Salm et al. 2000; Gell et al. 2003). This stems from the allocation and proclamation being a difficult process of pragmatic, economic and humanitarian considerations, often resulting in proclamation of sub-optimal habitat owing to socio-economic considerations outweighing ecological priorities (Robinson & de Graaff 1994).

The competition between resource exploitation and the limited resources available to conservation therefore necessitates optimal conservation resource allocation to satisfy spatially explicit and goal directed conservation (Margules & Pressey 2000; Leslie 2005). The hasty development and integration of technology (Geographical Information Systems (GIS), remote sensing data, conservation planning software e.g. C-Plan, Marxan, Zonation) and statistical modelling techniques (general linear regressions, general additive models, bioclimatic envelopes and Maxent models) have fuelled the development of the systematic approach to conservation planning (Simberloff 1997; Goodchild 1999). Systematic conservation planning is employed by most developed and developing nations today in an attempt to increase their marine protected area estates (Cowling & Pressey 2003; Cowling et al. 2003c; Leslie 2005).

This approach has been increasingly used in bioregional and provincial conservation plans (both terrestrial and marine) in South Africa, for example, the Cape Action Plan (CAPE) (Cowling et al. 2003b), the Succulent Karoo Ecosystem Plan (SKEP) (Driver et al. 2003),

and the Subtropical Thicket Ecosystem Plan (STEP) (Cowling et al. 2003a). Although still a youthful approach in the marine environment, it is emerging as a useful conservation tool, for example, the Prince Edward Island marine reserve proposal (Lombard et al. 2007), and establishment of “no-take” reserves on Australia’s Great Barrier Reef (Fernandes et al. 2005). Systematic conservation planning in the marine environment has come a long way since applications (e.g. McAllister et al. 1994).

Ezemvelo KZN Wildlife (EKZNW) is currently developing a fine-scale systematic conservation plan, SeaPLAN, for the marine environment of the KZN province, South Africa. The key objective of SeaPLAN is to conserve biodiversity including biotic and abiotic processes, habitats and species (Lagabrielle et al. 2010). The study area is KZN’s EEZ which stretches along the shoreline from the Mozambique border in the north, southwards to the Umtamvuna River, and 200 nautical miles offshore.

Conservation planning in the marine environment is inherently hampered by data deficiencies, particularly in offshore areas. Benthic habitats are poorly mapped, abiotic and biotic processes are not clearly understood in time and space, species distributions are poorly mapped, and factors governing their distributions are different from terrestrial species (Norse 1993; Robinson & de Graaff 1994). Not all biodiversity can be included in a conservation plan, owing to the time required to collect species and habitat data. Consequently, the data used in the conservation plan should attempt to include as diverse species and habitat data as possible, in an attempt to represent as much of the variety in a given area as possible (Ferrier et al. 2000; Possingham et al. 2005). The systematic process of conservation planning frequently uses species or habitats as surrogates to represent unmapped biodiversity (these species, habitats, or even ecological processes, are referred to as biodiversity features). In the ocean, fish species are useful surrogates for biodiversity as they are well studied and inhabit a vast array of different kinds of habitat, ranging from coastal reefs systems, to canyons, to the offshore pelagic zone (see Heemstra & Heemstra 2004). I contributed the marine fish species component to SeaPLAN. Time and resource constraints limit the size of this ‘sample of biodiversity’ of the ca 1431 marine fish species in KZN (Junor 1992) to a mapable number. To identify which species will be used as biodiversity features in the conservation plan, I ask the question: *How does one identify appropriate species for a marine conservation plan?*

Conservation efforts often focus on threatened species (e.g. IUCN), rare species (Kattan 1992), endemic species (Wilson et al. 2006) or combinations thereof (Musick 1999). As yet there is no clear method on which to base species selection for conservation planning (Marris 2007). In Chapter One, I investigate previous methods used and develop a set of criteria to select species that meet the objectives of SeaPLAN.

Conservation planning requires information on the distribution of biodiversity features (in this case, marine fish species), within the planning region, but cannot wait for complete knowledge given the ongoing decline of marine systems (Grantham et al. 2009; WRI 2005). Instead, conservation planning must often rely on predicted species distributions (Wilson et al. 2005).

Predicted species distributions require species occurrence and environmental data (Guisan & Zimmermann 2000). Ongoing marine fish data collection programmes from various organisations (Marine and Coastal Management (MCM), Oceanographic Research Institute (ORI), South African Institute for Aquatic Biodiversity (SAIAB)) are available. Recent assessments of data collated by online repositories, for example, Oceanographic Biodiversity Information Facility (OBIS), have highlighted the need for careful evaluation of point data prior to using it in distribution models or basing management decisions on it (Robertson 2008; Robertson et al. 2010). In Chapter Two I assess the spatial and temporal (spatio-temporal) accuracy of some of the data for South African marine fish species to determine its usefulness for distribution modelling and conservation planning.

Although species distribution modelling techniques have progressed to a high level of sophistication (see Elith et al. 2006 for a review), all results are still subject to data quality. Given the varying data quality, the following question is posed: *How can we best use the available data in South Africa to model the marine fish species distributions?*

I explore how literature descriptions of species distributions and their associations with broad-scale habitat information (e.g. bathymetry, reefs, and coral reefs) can be used (with Boolean multiplication) to generate cartographic habitat association range models (CHARMs). I then incorporate expert advice on the species preferred ranges to refine the CHARMs. I also explore the use of information on differences between the distribution ranges of a given species during different phases of its life cycle. I to complement and refine the distribution models (I refer to these as Species Life Cycle Envelopes, or SLICEs). The point-based data, evaluated in Chapter Two, are used in conjunction with remotely-sensed satellite data for oceanographic variables (e.g. sea surface temperature) to construct probability of occurrence models using Maxent software. In Chapter Three I develop and use a dichotomous key to support decision making with regards to the data and distribution modelling techniques available and apply it to the fish species selected for the conservation plan, SeaPLAN. Different suites of species are known to occupy KZN waters during summer and winter (van der Elst 1988), and their seasonal distribution ranges were modelled to identify differences in seasonal distributions.



Systematic conservation planning also requires that explicit quantitative and operational targets be set for biodiversity features (Margules & Pressey 2000). These targets are usually set as a percentage of known (or inferred) distribution (e.g. 20%) to be included within MPAs. A conservation assessment is then undertaken, in which the overlap between existing MPAs and biodiversity features is calculated. The amount of each feature present in an MPA tells us which biodiversity features have their targets met in MPAs, and which ones are underrepresented in MPAs (and by how much). In order to meet these currently unmet targets, conservation planning software is often used to delineate these additional areas (Cowling et al. 2003b; Fernandes et al. 2005; Lombard et al. 2007). GIS software is used to divide the planning domain into planning units (e.g. identically sized grid cells or hexagons) and measure the presence (or abundance) of the biodiversity features in each of these units. In Chapter Four I assess the conservation status of the species based on the best set of distribution models developed in Chapter Three.

The existing MPAs in KZN are zoned A, B, and C, imposing different regulations within the respective zones (see Table 1). MPA zones A are no-take marine sanctuaries where no fishing, harvesting or other activities harmful to the ecosystem are allowed. MPA zones B are controlled zones that allow some activities, for example, game fishing is allowed but not bottom fishing. MPA zones C allow extractive use, for example, fishing and removal of marine invertebrates, but not pollution or land transformation. Conservation targets for all species can therefore only be met in A zones, while B zones do not contribute to game fish targets, and C zones do not contribute to any marine fish species' targets. The planning software then chooses planning units to add to the existing reserve system so that all targets can be met by a new, expanded reserve system. Objectives for reserve design or configuration are set in terms of software parameters, for example: (1) the total additional area required must be minimised, or (2) new planning units must be clumped or adjacent to existing reserves, etc. The result of this exercise informs a conservation plan (Knight et al. 2006; Moilanen et al. 2009). I examine conservation-planning outputs for three 'starting point' scenarios (areas with their protection status predetermined): (i) all current MPA zones A are considered as contributing to target achievement (i.e. used as starting points); then (ii) zones A and B are used as starting points (here B zones are theoretically rezoned to A zones); and (iii) zones A, B and C are used as starting points (i.e. both B and C zones are theoretically rezoned to A zones).

**Table 1. Summary of restrictions in the three marine protected area (MPA) zones in KwaZulu-Natal (KZN), and IUCN\* equivalent categories.**

<i>MPA ZONE</i>	<i>KZN Wildlife Category</i>	<i>Summarised restrictions</i>	<i>Applicable IUCN categories</i>
A	Sanctuary zone	No fishing or any other extractive uses. No actions that result in ecosystem damage.	Ia
B	Restricted zone	Restricted extractive use, e.g. game fishing, but no bottom fishing.	II and III
C	Controlled zone	Controlled fishing, and other extractive uses. Fishing for several species is allowed. No activities with broad scale ecosystem impacts, e.g. pollution, dredging.	III, IV and V

\*International Union for Conservation of Nature (see Dudley 2008).

In Chapter Four conservation targets are set for marine fish species using baseline and biological retention targets. Baseline targets are the minimum target (ca 20 %) that has been recommended to viably conserve biodiversity (WSSD 2002; Agardy et al. 2003; Svancara et al. 2005). Targets are to some extent area dependent, and therefore species with small ranges may not necessarily be viably conserved by the baseline target. Retention targets are added to the baseline target for species that require more than the minimum target of their ranges conserved. Abundance of especially migratory species in KZN is highly seasonal, e.g. shad (*Pomatomus saltarix*). Conservation targets play an important role in the assessment and in guiding network selection to meet currently unmet targets. Consequently, targets were adjusted to seasonal abundance of the same suite of fish species for which no temporal variation was taken into account. I run a conservation plan based on the conservation status assessment for the three temporal divisions of the data (annual, summer and winter distributions). Conservation resources are often limited and it is therefore important that the spatial distribution of protected areas is efficient in the conservation of biodiversity (Hobday & Hartmann 2006; Game et al. 2009). In Chapter Four I ask: *How differently would conservation planning software allocate spatial marine protected areas if seasonal variations of fish distribution patterns and abundance are considered?*

I explore the use of temporal variations in the distribution and abundance of biodiversity features to guide seasonal reserve selection. The differing levels (i) to (iii) of starting point protection described above are used in combination with the three temporal data sets to create nine scenarios which are used to explore the impact of MPA zonation and fish seasonality on target achievement.

The shortcomings of using a single group of species as a biodiversity surrogate has been pointed out by Beger et al. (2007) for Indo-Pacific coral reef species. The present study focussed only on a single group of species, while SeaPLAN included a far wider selection of biodiversity, habitats and processes. Results of this study are therefore compared with those of SeaPLAN to identify similarities and differences.

Two themes of this study, prioritising species for conservation planning and modelling their ranges using CHARMs, were previously used in SeaPLAN. An important aspect of conservation planning is that its progress is monitored and that plans be updated as new information becomes available (Grantham et al. 2010). This study investigated and developed the data preparation (e.g. Maxent species distribution models using point locality data), and introduced some novel ideas (e.g. temporal differences in biodiversity distribution and target achievement), which will in turn feed back into SeaPLAN. The lessons learned during this study are discussed, and recommendations are made of how the aims of the various chapters could be better achieved and applied to conservation plans.

## **Study Area**

The study area is confined to the marine environment of the KZN Province in South Africa (Figure 1). The area defined as South Africa's responsibility by the United Nations Convention on the Law of the Sea is known as the Exclusive Economic Zone (EEZ). The EEZ starts from the shore and extends out to 200 nm (370.4 km) offshore and forms the outer perimeter of the study area. The northern boundary of the study area is at Kosi Bay (Mozambique border), and the Umtamvuna river mouth (Eastern Cape Province boundary) in the south. The KZN coastline is ca 640 km long and the EEZ covers 233 747 km<sup>2</sup> (126213 nm<sup>2</sup>) (Lagabrielle et al. 2010).

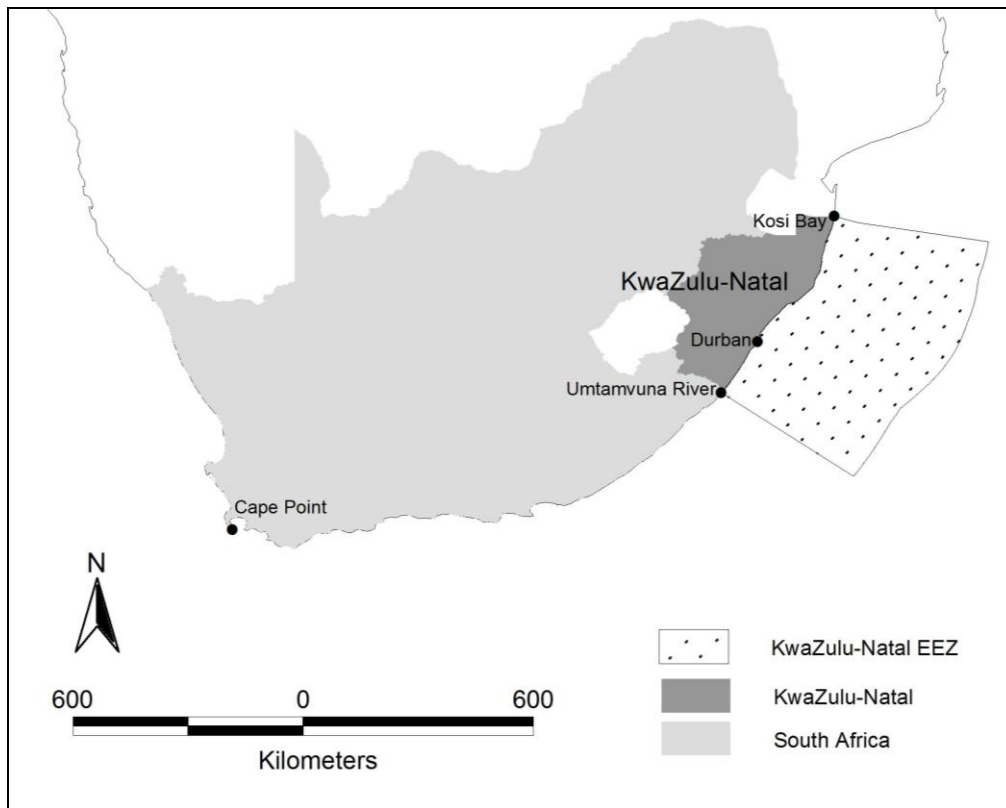


Figure 1. The KwaZulu-Natal Exclusive Economic Zone extends from the shoreline to 200 nm offshore, from the Mozambique border (marked Kosi Bay) in the north, to the Eastern Cape boundary (marked Umtamvuna River) in the south.

## Coastal oceanography of KwaZulu-Natal

The KZN coastline is bordered and strongly influenced by the Agulhas current, one of the world's major western boundary currents (Lutjeharms 2006). The narrow KZN continental shelf slopes steeply for most of the coastline, the 200 m depth contour being within 20 km (Lutjeharms 2006). The shelf widens and slopes more gently in the Natal Bight area (Durban to Richards Bay) inducing variability in the otherwise stable and fast flowing Agulhas current (Lutjeharms & van Ballegooyen 1984), which has a considerable effect on the adjoining shelf circulation (e.g. Natal Pulse) (Lutjeharms 2006). Nearshore counter currents are observed from time to time, and are thought to be generated by strong local winds, and during cold fronts that travel up the coast from the Cape (Lutjeharms 2006).

Seasonal changes in ocean climatology result in turnover of seasonally dynamic fish assemblages (van der Elst 1988). During the summer when sea surface temperatures are warmer (ca 28° C), tropical species extend their feeding ranges further south from the tropical regions, e.g. hammerhead sharks (*Sphyrna mokarran*), yellow fin tuna (*Thunnus albacares*), and king mackerel (*Scomberomorus commerson*). Lower water temperatures (ca 23° C) during winter allow more temperate water species, several of which are endemic to Southern Africa, to migrate into KZN from the south to complete reproductive phases of their life cycles, e.g. sardines (*Sardinops sagax*). They are in turn followed by species such as copper sharks (*Carcharhinus brachyurus*) who exploit the rich feeding grounds.

The KZN EEZ includes two distinct nearshore bioregions namely, the Delagoa and Agulhas bioregions (Figure 2, Chapter 1), which meet at Cape Vidal (Sink et al. 2010). The water clarity and its high temperatures in northern KZN allow for coral reefs to grow, as far south as Cape Vidal, but also at Aliwal Shoal further south (Schleyer 2008). Water temperatures are lower south of Richards Bay where the Agulhas current veers further offshore. From here, the waters are more turbid and the substratum changes to rock reef and sand, and consequently the fish community composition changes. The coastline is broken by several estuaries, that supply rich feeding grounds to species, e.g. bull sharks (*Carcharhinus leucas*), spawning grounds for seabream (*Acanthopagrus vagus*) and juvenile nurseries for dusky kob (*Argyrosomus japonicus*) (Whitfield 1998). The two bioregions in KZN's waters connect South Africa's temperate marine fish-endemism to the diversity of more tropical latitudes, and play an important role in the species that were included in the study, and finally, the areas that are important for conservation.

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## Chapter 1

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# Selecting Fish Species for Marine Conservation Planning

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## Abstract

Several criteria have been proposed to identify species for conservation planning. As yet, no consensus has been reached at an international level. This lack of consensus also underpins many conservation-planning efforts that strive for complete species lists. The different objectives of individual conservation plans require different types of species lists, e.g. the IUCN's threatened species list is relevant for conserving only threatened species (and not, for example, endemic species). Systematic conservation planning attempts to include as representative a sample of biodiversity as possible, within the resource limitations of the project, and therefore should include species that are the least likely to receive any form of protection if not explicitly mapped (for example, species that may not be well-represented by the use of habitat surrogates). In this study, I defined three underlying principles to identify seven criteria that would prioritise marine fish for the KwaZulu-Natal provincial spatial conservation plan, SeaPLAN. The three principles included species with limited conservation options, threatened species, and species that are inherently vulnerable to extinction. The seven criteria were developed to include all biodiversity, and not to focus only on harvested species. These criteria included endemic species; species of conservation concern; species with life history vulnerability; highly resident species; estuarine-dependent

species; rare species; and species dependent on specialised habitats. Two hundred and eighty of the ca 1430 species that have been recorded in KwaZulu-Natal satisfied the conditions of at least one criterion. This was too large a number of species to include within the time constraints of the project. Consequently, species were retained only if they satisfied the qualifying conditions of at least two criteria, of which one was either endemic species, or species of conservation concern. By this method, 67 species were selected for conservation planning. Most (22 %) of the species were seabreams (Sparidae), 8% were gobies, 6% were kobs (Scaenidae), and 5% were rockcods (Serranidae). The lack of information on specialised habitat dependency limited the number of species that qualified for this criterion. Criteria were defined to best meet the objectives of the project (i.e. biodiversity representation), but needed to be appropriate and practical for the project and its time and budget constraints. Prioritising species for similar conservation projects can therefore be based on similar principles, but different project objectives may require different use or combinations of the criteria defined here.

## Introduction

SeaPLAN is a fine-scale systematic conservation plan that aims to conserve all marine biodiversity in KwaZulu-Natal (KZN), including species, habitats and ecological processes. Conservation planning requires that the spatial distribution of biodiversity features such as species, habitats, and processes, be mapped to make spatially explicit management recommendations (Margules & Pressey 2000). Habitats in the marine environment include features such as coral reefs, rock reefs, mud banks and canyons, and processes are typically features that are not fixed in time and space, such as eddies and chlorophyll fronts.

The urgency of conservation planning requires decisions on the biodiversity features that should be prioritized, as not all can be included in the conservation plan (Marris 2007). In SeaPLAN the species selection was limited to oysters, turtles, a selection of marine fish and mammals, as there were too many species to include (e.g. ca 1431 marine fish species alone (Junor & Coke 1992)) and distribution data are typically sparse. This study reports on the marine fish species used for SeaPLAN. Only species confined to the KZN Exclusive Economic Zone (EEZ) were eligible for consideration, but the ca 1431 species were too many for a single project to deal with. Consequently, a method was developed to prioritise a subset of these species for conservation planning.

In conservation planning environmental parameters like broad habitat types are often used as surrogates for unmapped and unknown species distributions (Faith et al. 2004; Begger et

al. 2007). However, the use of environmental parameters as surrogates is appropriate only if there is good congruence between the surrogate's distribution and the species' distribution (Lombard et al. 2003; Dalleau et al. 2010). Species that occur at finer scales than the scale used to map the habitats or ecological processes in a conservation plan will not necessarily benefit from protecting portions of these broader features (biodiversity surrogates). Owing to the risk that some species may not be represented by a map of broad-scale habitats (e.g. species restricted to fine-scale or highly localised and specialised habitats) or processes (e.g. species that concentrate at chlorophyll fronts), it is recommended that broad habitats maps and process maps as well as species distribution maps be used as data input for conservation plans (particularly maps of those species most at risk of 'falling through the broad habitat and process net' (henceforth referred to as 'falling through the net') (Noss 1983, 1987; Cowling & Pressey 2003; Brooks et al. 2004).

Under the assumption that most of the ca 1431 marine fish species in KZN (Junor & Coke 1992) occur only in a portion of the total available habitat or processes, most species were at risk of not being adequately protected by protecting portions of these features as surrogates. For example, the Natal wrasse (*Anchichoerops natalensis*) occurs in southern KZN on rocky reefs at 10 – 55 m depth (Heemstra & Heemstra 2004). By protecting a random 20 % of all rocky reefs in the province, which occur throughout the province from shore to ca 200 m depth, there is a chance that the rocky reefs over which the species occurs will not be protected adequately. Identification of the species that would be most likely to 'fall through the net' requires the recognition of underlying criteria that put these species at risk, such as species limited to specialized habitats that occur at a finer scale than those mapped, or species that are very range restricted, or species that are naturally rare (Kattan 1992; Gaston 1994; Lombard et al. 2003).

Previous studies have recommended that conservation efforts should focus on one or more of the following criteria: endemic species (Turpie et al. 2000; Cowling et al. 2003); threatened species (e.g. the International Union for Conservation of Nature (IUCN) list); the Threatened Or Protected Species (TOPS) list in South Africa; species with vulnerable life history strategies and/or low resilience to fishing pressure (Cheung et al. 2004); or combinations thereof (Musick 1999; Lamberth & Joubert 2005; Larsen et al. 2007).

Previous attempts to prioritize species for conservation and future research in KZN based their importance ratings on various factors including evaluating abundance trends, levels of knowledge, vulnerability, ranges and relative exploitation throughout that range (Junor 1992; Lamberth & Joubert 2005). The method employed by Junor was simple ranking within the categories, but the methodology focused on the prioritization of fisheries-important species.

Lamberth and Joubert (2005) used a Multi-Criteria Decision Analysis based on weights assigned to several of the above-mentioned factors. The strength of the method was its ability to discern priority amongst different interest groups like conservation priority of recreationally or commercially important species. The method was developed primarily to evaluate linefish species and is less applicable to non-harvested species (this is evident from their use of factors such as abundance estimates, which favour the prioritization of fisheries-important species).

Conservation efforts that only consider harvested species fall short as they disregard non-harvested species that are vulnerable to natural or stochastic environmental processes, e.g. the doublesash butterfly fish (*Chaetodon marleyi*) and the Knysna seahorse (*Hippocampus capensis*) are IUCN critically endangered due to small ranges and habitat degradation (Roberts 1996; Lockyear 2000; Atkinson & Clark 2005).

The philosophy of the SeaPLAN project was to conserve biodiversity and not just harvested species, as harvested species are not necessarily the species that fall through the 'broad habitat and processes nets'.

The advantages and shortcomings of existing methods used globally to identify species that are important for conservation are reviewed below and a new method to select appropriate species for SeaPLAN is developed (see Figure. 1). This method has the following problem statement:

By conserving proportions of broadly-mapped habitats and ecological processes, we will conserve many species, but some species are at risk of 'falling through the net'. How can we identify these species?

To solve this problem, I defined three underlying principles that can be used to identify such species:

1. Species with limited conservation options: these are species with small distribution ranges or species that occur in specialized habitats that occur at finer scales than the broad habitat and process maps.
2. Threatened species: species that have previously been identified as threatened with extinction unless appropriate conservation action is taken.
3. Inherently vulnerable species: these are species with characteristics that predispose them to being vulnerable to e.g. fishing or natural stochastic events.

## 1. Species with limited conservation options

The limited geographic region which a species occupies is often used in criteria to guide conservation decisions (e.g. IUCN Red List Criteria, Version 3.1). The limited range in which a species occurs is defined here as the range over which it naturally occurs prior to recent human interventions. For example, *Pristis* spp. natural range is regarded as extending through KZN to Port Alfred (Eastern Cape, South Africa) (Smith & Heemstra 1986) even though it is suspected that sawfishes (*Pristis* spp.) have become extinct in KZN as a result of sustained fishing pressure from anglers and sharks nets as well as a loss of estuarine function (Adams et al. 2006; B. Mann and S. Dudley pers. comm.). In conservation planning, conservation targets are based on natural ranges to avoid under-representing and selecting non-viably small areas for conservation (Margules & Pressey 2000).

Species with small natural ranges obviously have fewer spatial conservation options than species with large natural ranges and are therefore more likely to 'fall through the net' and therefore explicitly considered in this study.

A limited species distribution range is closely linked to endemism. Endemism is used to describe a species found only in a particular region. Endemism to a politically meaningful region, like a province, country or continent, is important from a management perspective, as it recognises the sole responsibility of the region to which the species is endemic to protect it.

Pioneering research in the field of systematic conservation planning in South Africa suggested that conservation efforts should be focussed on areas of high endemism and species richness (Rebelo & Siegfried 1992). Recent conservation efforts have emphasized the importance of endemic species to guide conservation efforts, as high endemism is better correlated to high biodiversity richness than areas with many threatened species (Orme et al. 2005; Possingham & Wilson 2005).

Rarity, at times closely linked to endemism (Gaston 1994), has also been a prominent criterion in the effort to prioritise species for conservation (Kattan 1992). Rabinowitz et al. (1986) used three traits to categorise the level of species rarity: (1) local population size (many, few), (2) the area of the species range (wide, localised), and (3) the habitat that the species occupy (broad, restricted). Various arbitrary (at times) cut-off values have been used for different species to define 'many or few', 'widespread or localised', and 'broad or restricted' in order to categorise them according to the system (see Gaston 1994; Manne & Pimm 2001). The actual values of these traits are of course continuous, but categorising



species aids conservation and other management decisions (Rabinowitz et al. 1986; Gaston 1994; Whittaker et al. 2005). In terms of this definition, the rarer species typically have 'fewer' individuals, and/or occupy a 'small' range and/or occupy 'restricted' habitats (Rabinowitz et al. 1986; Gaston 1994; Yu & Dobson 2000; Whittaker et al. 2005). Yu and Dobson (2000) found that the extreme cases of combinations were most prevalent in mammals (i.e. the majority of species were either: (1) abundant, widespread and occupying broad range of habitats or (2) few, localised, and occupying restricted range of habitats).

Manne and Pimm (2001) evaluated the correlation of IUCN threat status and the factors predicting rarity for, lowland-, montane-, and island- passerine birds of the New World. Local abundance, breeding range size and elevation were used as a version of the eight forms of rarity (Rabinowitz et al. 1986). Abundance and range size were the main predictors of threat status, while habitat specificity was less important.

Rarity and species threat status have been conflated in the past to set conservation priorities (Isaac & Grace. 1998; Robbirt et al. 2006) because both include the criteria of abundance and range size.

The IUCN Threatened species includes criteria that assess the threat level based on range size (area of extent and/or area of occurrence) and population size, but not habitat specificity. The IUCN criteria were revised in 2001 to, amongst other things, remove the conflation of threat and rarity (Isaac & Grace 1998; Robbirt et al. 2006). Version 3.1 of the IUCN Red Listing Criteria excludes species that are naturally rare but the population is stable. The IUCN Criterion D, however, still includes very rare (the population size is very small or the range is severely restricted species) (IUCN 2001; Victor & Keith 2004; Robbirt et al. 2006).

Although rare species and threatened species are not necessarily the same, the rarer a species, the more likely it is to be threatened (Manne & Pimm 2001). The plight for conservation of rare species that do not currently qualify for the IUCN Red List, but are likely to qualify in the near future, initiated the development of the Orange List for South African plants (Victor & Keith 2004). Victor & Keith (2004) recommend a precautionary approach whereby rare species are included on the Orange List for conservation consideration.

Species with limited conservation options (e.g. endemic and/or rare species) are not necessarily the only species that are threatened by extinction (Gaston 1994; Robbirt et al. 2006), nor do their distribution ranges necessarily overlap (Orme et al. 2005; Grenyer et al. 2006) and therefore threatened species need explicit consideration.

## 2. Threatened species

Threatened species are species that have been identified and listed as threatened by extinction, owing to the combination of recent human interventions and the species' biological constraints. The best known list of species of conservation concern is probably the IUCN's Red List, in which a classification system was developed in 1994 based on population parameter criteria (e.g. decline in species range or abundance) to identify species that were threatened by extinction (IUCN 1994).

Lists of threatened species alone, however, are not appropriate for the present study. The shortcomings of the IUCN Red List of Threatened Species and many other threatened species lists (e.g. American Fisheries Society (AFS); United States Endangered Species Act (US ESA); Committee on the Status of Endangered Wildlife (COSEWIC); Convention on International Trade of Endangered Species (CITES)) for conservation planning arises from the mismatch in their purposes of use, as well as shortcomings of the species assessed. Threatened species lists identify species threatened by extinction, whereas the purpose of conservation planning is to conserve a representative sample of biological diversity present in a region – a goal which may not necessarily be achieved by conserving threatened species alone (Musick 1999; Possingham et al. 2002; Orme et al. 2005; Possingham et al. 2005).

Threatened species lists like the IUCN Red List require quantitative data for population abundance and/or range declines to evaluate species threat status (critically endangered, endangered or vulnerable). Evaluations include observations, estimates or proxies of decline in range, and population size, (e.g. declines in Catch per Unit Effort (CPUE)). These thorough but data-intense evaluations and their reviews (see IUCN 2001) may postpone management decisions, such that it may already be too late for many species by the time the necessary information is gathered (See Cowling et al. 2010). In addition, threatened species assessments are biased towards charismatic megafauna and special research group interests, like groupers and wrasses (Norse 1993; Richardson 2002). Locally, elasmobranchs and sea breams (Sparidae) are currently receiving attention and several new species have recently been added to the IUCN's Threatened Species List (e.g. whitespotted izak, *Holohalaelurus punctatus*) (Human 2008)). The World Wildlife Fund's (WWF) methodology for assessing stocks of wild-caught species, or other methods like stock assessment-based methods, restricts conservation to consider mostly harvested species (see Lamberth & Joubert 2005; WWF 2007).

It remains important to consider these species explicitly though, as not all biodiversity can hope to be protected with the limited resources available to conservation; and threatened species may be most at risk of being lost without any formal protection (Diamond et al. 1976). Threatened species lists alone are however not sufficient to prioritise species for conservation planning because of the mismatch in objectives, the incomplete current assessments, and the long time that it will take to assess all species.

### **3. Inherently vulnerable species**

Inherently vulnerable species are species that are vulnerable owing to their biology (e.g. size, aggregative spawning behaviour). These inherent vulnerabilities predispose them to negative impacts from both anthropogenic and stochastic environmental events (e.g. large fish are targeted in fishing operations, and are typically long lived and slow to reproduce).

Given the problems with threatened species lists, an alternative method is used to determine if a species might be vulnerable to threat. Cheung et al. (2004) developed a method of pre-emptive identification of species that may be vulnerable to threat based on inherent characteristics. His method is used by FishBase (<http://www.FishBase.org>) (Froese & Pauly 2009). The method calculates an inherent vulnerability score based summation of membership (a sliding scale from low to high values) to characteristics identified as predisposing a species to being vulnerable (e.g. species that spawn in aggregations, long lived, or large species). Life history characteristics have also been used to identify plant species that are more vulnerable to extinction (Farnsworth & Ogurcak 2008). Although this method is useful because it is more rapid than listing a species on a threatened species list, it does not necessarily identify species with limited conservation options, such as rare species, or species that occur in specialized habitats.

### **Practical considerations**

Practical considerations have to be taken into account when selecting species for distribution modelling and conservation planning. Species can only be mapped (or their ranges modelled) if adequate information is available. Species do tend to occur outside of their natural ranges from time to time. A single occurrence record may therefore not be a true reflection of the species range (Malcolm & Hutchinson 1994). Vague locality records for a species may leave a species with too little information to be mapped (e.g. a single specimen recorded from locality 'Natal', with no depth or habitat association information). Further considerations include the amount of work manageable in limited time available to the project. During SeaPLAN projects, we restricted the number of species that could be

included in the conservation plan to a mapable number of species, as described in the methods.

The prioritization of species and/or habitats for conservation planning has been heavily disputed over the past 20 years without definite resolve (Possingham & Wilson 2005; Marris 2007; Bottrill et al. 2008). As discussed above, no one method is sufficient to identify species for a conservation plan. For SeaPLAN, I thus developed a method that integrates the existing methods, and supplements these with expert workshops and literature reviews. I explain this method in detail and evaluate the criteria (see Figure 1) in the remainder of this chapter. The resulting species list and the database of criteria satisfied, including expert comments, will be made available to the provincial conservation authority, Ezemvelo KZN Wildlife (EKZNW) and to the South African National Biodiversity Institute (SANBI).

## Methods

Experts were consulted to identify underlying principles that would identify species at risk of 'falling through the net' (i.e. not being represented by mapping and conserving portions of broad scale habitats and processes alone) (Figure 1). Experts included scientists with expertise in fisheries, marine ecology, biodiversity and conservation planning.

From the three underlying principles identified, seven criteria were defined, each with qualifying conditions (Figure 1). Species information was drawn from selected reference works (see results section) and compared with the qualifying conditions of each of the seven criteria. If a species met the qualifying conditions of any of the seven criteria, the species was included in List One of the conservation plan (this list rendered 280 species). We removed species with < 10 records in KZN, or species with taxonomic ambiguity from List One to produce List Two (250 species). List Two, however, still contained too many species for the resources of this study and the species on List Two therefore needed further prioritization.

In order for a species to remain on the final list (three) it had to satisfy the qualifying conditions of at least two of the seven criteria, of which one criterion had to be either endemic species or species of conservation concern. We assigned higher priority to these two criteria than the other five because they best matched the objectives of the study (SeaPLAN is a provincial conservation plan, concerned with the conservation of provincially-endemic species). Henceforth, we refer to these two criteria as the primary criteria. List Three contained 67 species and was considered manageable for the study. List Three was evaluated by the experts whose comments and suggestions are included in the discussion.

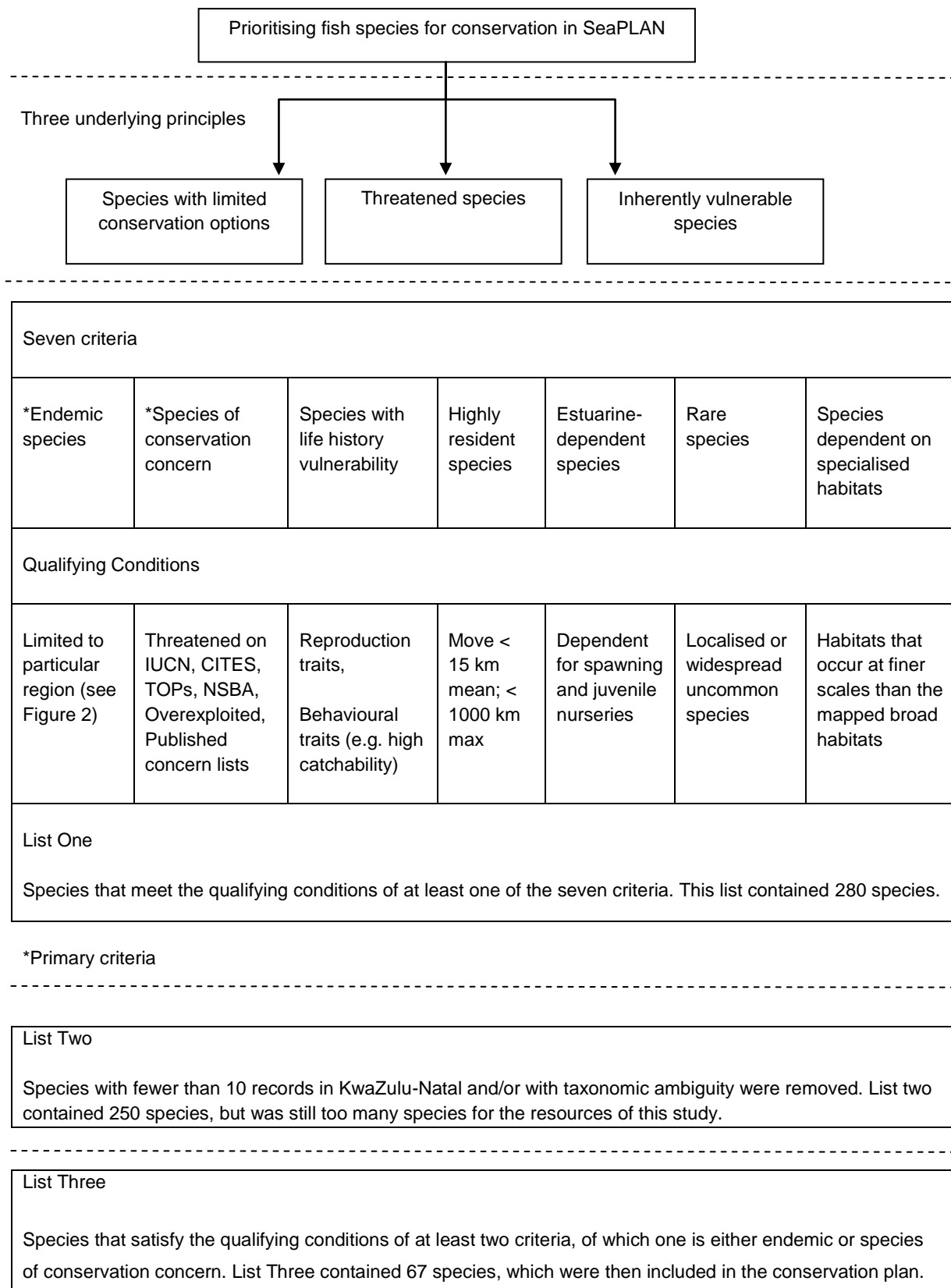


Figure 1. Diagram illustrating the process followed to prioritize fish species for the conservation plan.

For each of the seven criteria (Figure 1), the qualifying conditions were defined as follows:

## Endemic Species

Definition: Species confined to a particular political province or bioregion (see Figure 2).

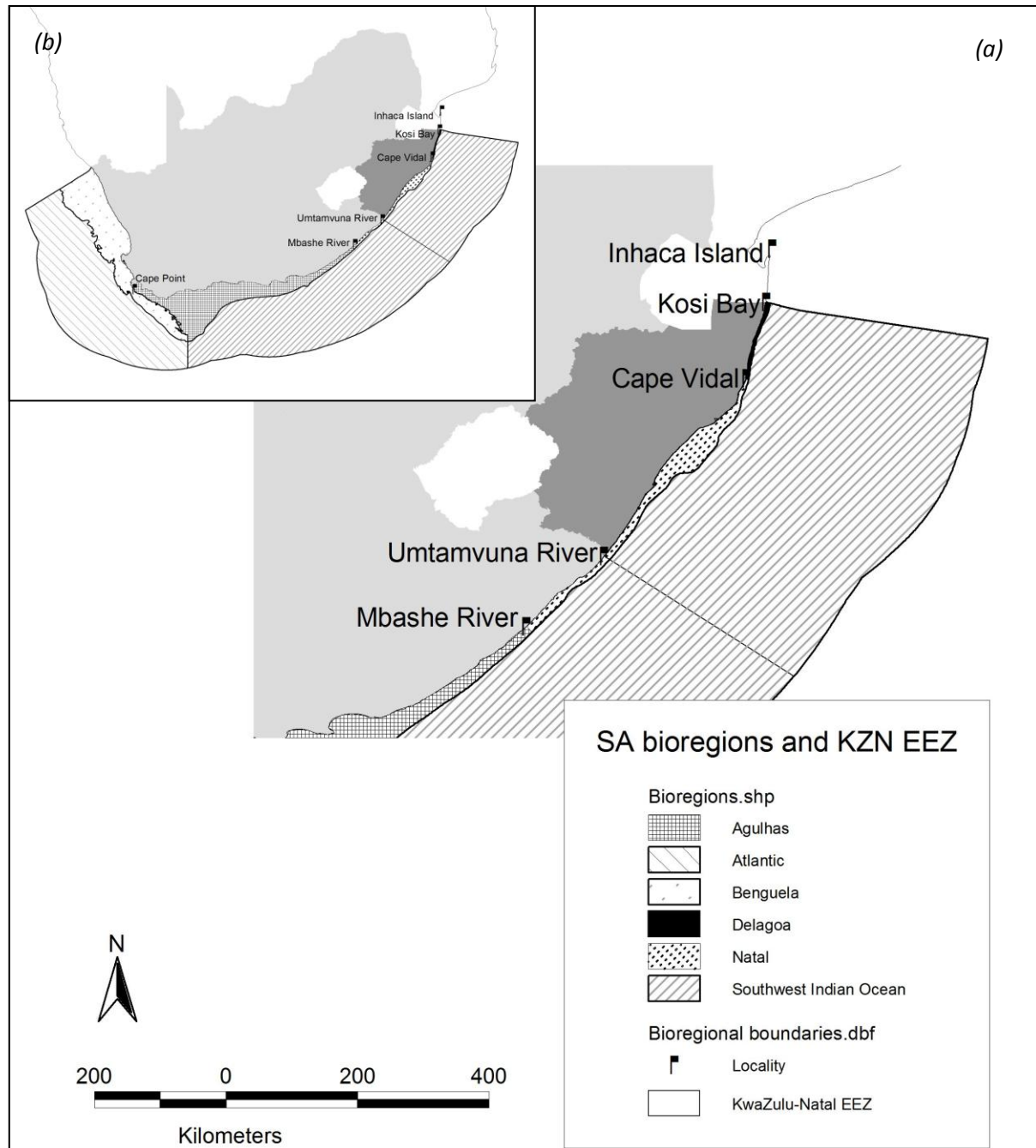


Figure 2. (a) The marine bioregions in KZN and the boundaries of the KZN EEZ. The KZN EEZ starts at Kosi Bay (north) and ends at the Umtamvuna River (south), and extends 200 Nm offshore. (b) The inset shows the bioregions for South Africa. The legend shows all bioregions in South Africa, based on Sink et al. (2010).

## Qualifying Conditions

1. Range restricted species - species that are confined to a particular range within a bioregion at a finer scale than that of province or bioregion. The KZN political province consists of three bioregions: two nearshore, the Delagoa in the north and the Natal in the south, and one offshore, the southwest Indian Ocean bioregion (Figure 2). Only species from the two nearshore bioregions were considered, as offshore pelagic species typically are much wider ranging. The species had to occur in the province. A cut off of less than 25% of a bioregion was used, because this was the cut-off used for invertebrates, reptiles and amphibians in the terrestrial KZN conservation plan (Goodman & Escott 2010). For example, the small tooth flounder (*Pseudorhombus natalensis*) is confined in the coastal area from the Tugela River to Durban.
2. KZN provincial endemic - confined to the area within the political borders of the province of KZN to the 200 nm offshore (EEZ) boundary.
3. Natal bioregion endemic - confined to the area within the Natal bioregion i.e. from the Mbashe River to Cape Vidal. This list excludes any KZN provincial endemics listed previously.
4. Delagoa bioregion endemic - confined to the area within the Delagoa bioregion i.e. from Inhaca Island in southern Mozambique to Cape Vidal. This list excludes any KZN provincial endemics listed previously. In addition, Delagoa bioregion endemics not found south of Inhaca Island were excluded.
5. Natal + Delagoa bioregion endemics - confined to the area within the Natal and Delagoa bioregions (i.e. from Inhaca Island to the Mbashe River).
6. Natal + Agulhas bioregion endemic - confined to the area within the Natal and Agulhas bioregions – Cape Vidal to Cape Point. The Agulhas bioregion was not individually considered as its northern boundary, the Mbashe River, is outside the KZN province.
7. East Coast endemic - (Natal + Delagoa + Agulhas bioregions) - confined to the area within the Delagoa, Natal and Agulhas bioregions (i.e. Inhaca Island to Cape Point).
8. The offshore boundary of the bioregions is set at the edge of the continental shelf. Species endemic to wider ranges were not considered (e.g. endemic to the Southwest Indian Ocean).

## Sources

- Smith & Heemstra (1986)
- Compagno et al. (1989)
- van der Elst & Thorpe (1989)
- van der Elst (1989)
- Junor (1992)
- Heemstra & Heemstra (2004)
- Expert knowledge: B. Mann, S. Dudley, D. King and M. Smale

## Species of conservation concern

Definition: species considered to be threatened with extinction; species with overexploited stocks; or species of high conservation concern. Species whose decline would lead to cascading ecosystem effects were also considered but finally excluded on the basis that this would duplicate overexploited species or species with published concern.

## Qualifying Conditions

1. Listed as threatened on international or national lists
  - a. IUCN threatened species list: critically endangered (CR); endangered (EN); and vulnerable (VU). Previous versions of the IUCN Red List categories were included as many species have not yet been reassessed by the new criteria (Version 3.1).
  - b. CITES listed in Appendix I or II.
  - c. Threatened or Protected (TOPs) listed species (four categories of critically endangered, endangered, vulnerable and protected species).
  - d. Listed in the National Spatial Biodiversity Assessment (NSBA) (Lombard et al. 2004, Appendix 3).
2. Overexploited species in two categories
  - a. Collapsed stocks - the stock status provides evidence that the stocks of these species are collapsed i.e. < 25% spawner biomass per recruit (SBPR) remains (Griffiths et al. Unpublished report).



- b. Overexploited - this category includes species that are not collapsed but are considered overexploited by the literature (Sources below) (i.e. < 40% SBPR).

### 3. Species of published concern

This category should include species that are caught in large quantities as bycatch of commercial fisheries. Further information is required to finalise this list, because no stock assessment has been done for the species, but it has been flagged for conservation concern by expert opinion or publication.

#### Sources

- Fennessy (1994)
- Fennessy (1994)
- Fennessy & Groeneveld (1997)
- Mann (2000)
- Lombard et al. (2004)
- Lamberth & Joubert (2005)
- Dudley & Simpfendorfer (2006)
- TOPs list (2007)
- CITES (2009) Appendix I & II

#### Species with life history vulnerability

Definition: species that are vulnerable to environmental or anthropogenic impacts because of one or more of their inherent life history characteristics, or species with limited conservation options owing to confinement to particular areas during different life-history stages.

#### Qualifying Conditions

1. Reproductive traits
  - a. High age at maturity/Slow growth and longevity
  - b. Species that aggregate to spawn (these areas are targets for fishing)
  - c. Low fecundity/Small litter size (elasmobranchs)
2. Behavioural traits
3. High catchability

Some species are prone to fishing exploitation owing to characteristics such as vulnerability to spearfishing (as a result of curiosity, e.g. Natal fingerfin (*Chirodactylus jessicalenorum*), or species that are attracted to or readily approach divers, e.g. potato bass (*Epinephelus tukula*), or species with a high likelihood to take fishing bait, e.g. catface rockcod (*Epinephelus andersoni*).

### Sources

- Smith & Heemstra (1986)
- van der Elst (1989)
- van der Elst & Adkin (1991)
- Mann (2000)
- Heemstra & Heemstra (2004)
- Lamberth & Joubert (2005)
- IUCN (2009)
- Expert advice on species likely to be targeted by spearfishers, and species likely to take bait: B. Mann, S. Dudley, M. Smale, D. King and J. Williams.

### Highly Resident Species

Definition: species known (either through tagging or behavioural studies) to have a small home range size and to remain fairly resident in one habitat during the adult stage, e.g. species that set up territories, especially during the spawning season, such as white steenbras (*Lithognathus lithognathus*).

### Qualifying Conditions

1. ORI tagging database:
  - a. Mean distance moved by tagged species < 15 km
  - b. Maximum distance moved < 1000 km
  - c. 10 individuals have been recaptured

### Sources

- ORI tagging database (Bullen et al. 2008)

### Estuarine-dependent species

Definition: species that generally spawn within estuarine systems or species that breed at sea but their juveniles use estuaries as a nursery area.

### Qualifying Conditions

Marine species that are estuarine dependent or estuarine species that generally breed in estuaries, as classified by Whitfield (1998). These species were subdivided as follows:

1. Estuarine species that breed in southern African estuaries, divided into two groups:
  - a. Resident species that have not been recorded spawning in either marine or freshwater environments
  - b. Resident species that are also known to have marine or freshwater breeding populations
2. Euryhaline marine species that usually breed at sea with juveniles showing two different levels of estuarine dependence
  - a. Juveniles dependent on estuaries as nursery areas
  - b. Juveniles occur mainly in estuaries, but are also found at sea

Marine species whose juveniles occur in estuaries but are usually more abundant at sea were not listed (2c). Marine species that occur in estuaries in small numbers but are not dependent on estuaries were also not listed (Chapter 3 Whitfield 1998). Freshwater species that may penetrate estuarine habitats were not considered and neither were catadromous species that use estuaries as a transit route. These species have been included in the freshwater and estuarine component of the provincial conservation plan (Goodman & Escott 2010).

### Sources

- Smith & Heemstra (1986)
- Compagno et al. (1989)
- Whitfield (1998)
- Harrison (1999)
- Mann (2000)
- Heemstra & Heemstra (2004)
- Lamberth & Joubert (2005)
- Expert knowledge: B. Mann, A. Connell

## Rare Species

Definition: In this study we considered rare species as a simplified version of the seven forms of rarity (Rabinowitz et al. 1986). A species can qualify for two categories of naturally rare: A localised- or widespread- species occurring in low density. This is a commonly used definition of rarity (Gaston 1994; Whittaker et al. 2005). These are species known from very few localities and are never common where found.

### Qualifying Conditions

1. Species cited as rare in key texts, or identified as rare by experts.
  - a. Localised rare - species cited as rare in key texts (see below), or species known from fewer than five locations within a bioregion (IUCN 2001).
  - b. Widespread rare - species that are considered 'thin in the water' i.e. species that are widely distributed and known from more than ten locations but occur in very low abundance.

### Sources

- Smith & Heemstra (1986)
- Compagno et al. (1989)
- Junor (1992)
- Heemstra & Heemstra (2004)
- Expert knowledge: B. Mann, S. Dudley

## Species that are dependent on specialized local habitats

Definition: species that are dependent on specialized local habitats within broad habitat types (at a finer scale than the habitat mapping) at any life stage.

### Qualifying Conditions

Species documented or reported to be confined to specific habitats that are not captured within the broad habitat types (e.g. Staghorn coral beds within coral reefs).

### Sources

- Smith & Heemstra (1986)
- Heemstra & Heemstra (2004)

## Removal of non-mapable species

Definition: species that have been recorded in KZN, but are known or thought to be vagrants from other areas, or species that have only been recorded a few times in KZN and their taxonomy is uncertain.

## Qualifying conditions

Species for which fewer than ten specimens have been collected, or there was unresolved taxonomic ambiguity, or there was insufficient distribution information to model the species' range.

## Sources

- Smith & Heemstra (1986)
- Compagno et al. (1989)
- Heemstra & Heemstra (2004)
- Expert knowledge: B. Mann, S. Dudley, and G. Cliff

List Three included species that satisfied at least two criteria of which one had to be one of the primary criteria (Figure 1).

The number of species that qualified to be listed under each of the seven criteria was calculated. As expected, many species qualified for more than one criterion. Final numbers and degrees of overlap amongst species per criterion are presented in the results.

Some species qualified for many conditions within one particular criterion. This was particularly evident in the following criteria: (i) Species of conservation concern; (ii) Endemic species; and (iii) Species with life history vulnerability. For example, the great white shark (*Carcharodon carcharias*) is placed in the IUCN Threatened Species List (VU) and in CITES Appendix I. The number of species that satisfied specific qualifying conditions was summed per qualifying condition for these criteria, and is presented in the results.

## Results and Discussion

As many as 280 of the ca 1431 marine fish species found in KZN (Junor & Coke 1992) qualified for at least one of the seven criteria (List One, Figure 3). Of these 280 species (List One), 250 species (List Two) were 'mapable' after 30 data deficient species were excluded. Of these 250 species, 67 species were selected to remain on List Three because they qualified for at least two of the seven criteria, of which at least one was a primary criterion (See Appendix 1).

As regards List Three (Appendix 1), no species qualified for all seven selection criteria, but four species qualified for four criteria, namely: the seventy-four (*Polysteganus praeorbitalis*), catface rockcod (*E. andersoni*), Natal wrasse (*A. natalensi*), and river bream (*Acanthopagrus vagus*). A further 13 species qualified for three criteria (Figure 3). A total of 67 species from 32 different families was selected for this final list. It was dominated by the following four families: Sparidae (sea breams, 22.4 %), Gobiidae (gobies, 9%), Serranidae (rockcods, 9 %), and Sciaenidae (kobs, 7%). All of the sea breams (15 species) qualified as endemic, and nine of these were also of conservation concern. The six gobies all qualified as endemic and estuarine dependent. The five kob species and five of the six rockcods qualified as being of conservation concern. Four of the five kobs, and five of the six rockcods were of conservation concern and also qualified for vulnerable life histories.

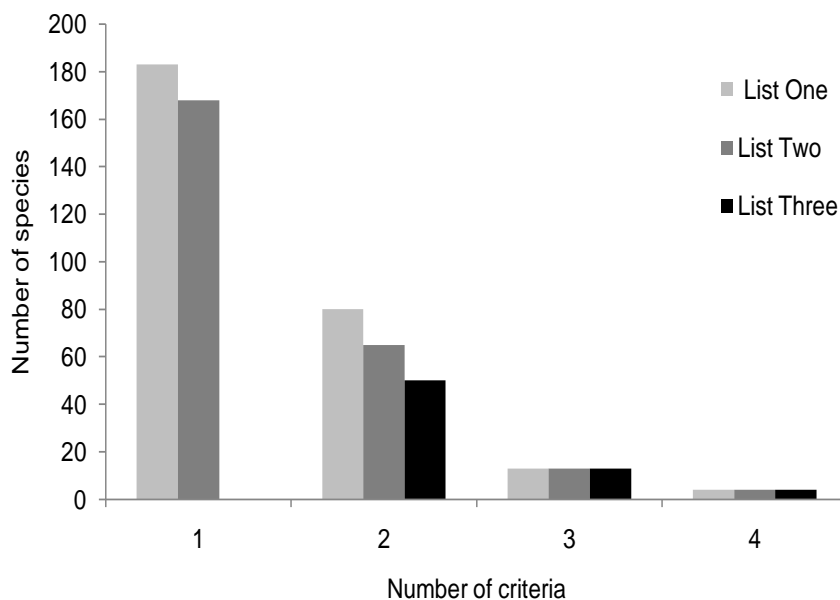


Figure 3. The number of species and the number of criteria for which they qualified. Data are shown for Lists One, Two and Three.

Of the seven criteria defined in Figure 1, only criterion 7 (Species dependent on specialized habitats) was not represented in the final List Three (Table 1). Eighteen endemic species were also of conservation concern (Table 2), including the species that qualified for three and four criteria (Figure 3). Fifty of the 67 species that qualified for two criteria, qualified for at least one primary criterion (Figure 3). In addition, the 17 species that qualified for more than two criteria also qualified for at least one of the two primary criteria.

**Table 1. Number of species that qualified for each criterion.\***

<i>Criterion</i>	<i>List One</i>	<i>List Two</i>	<i>List Three</i>
Endemic species	104	85	43
Species of conservation concern	76	72	42
Species with Life History Vulnerability	45	44	27
Highly resident species	26	26	11
Estuarine dependency species	73	73	16
Rare species	70	49	16
Species with specialized habitat dependency	4	4	0

\*Note that because a species can qualify for multiple criteria, it appears that more criteria are qualified for than there are species.

As regards List Three, of the 26 species that qualified with vulnerable life histories (Table 1), 23 were also species of conservation concern (Table 2). Eleven endemic species also had life history vulnerabilities (Table 2). Eight of the 11 highly resident species (Table 1) were also of conservation concern (Table 2). Several of the highly resident species are demersal linefish which have a population status that has been described as collapsed, or of concern (Mann 2000) (Appendices 1 & 2).

**Table 2. Pair-wise comparison of the number of species from List Three that qualifying for each criterion.\***

<i>Criterion</i>	<i>Species of conservation concern</i>	<i>Life history vulnerability</i>	<i>Highly resident</i>	<i>Estuarine dependent</i>	<i>Rare</i>	<i>Specialized habitat dependent</i>
Endemic	18	11	6	13	11	0
Species of conservation concern		23	8	4	7	0
Life history vulnerability			6	3	1	0
Highly resident				1	0	0
Estuarine dependent					1	0
Rare						0

\*Note that this table is unable to show that some species qualify for more than two criteria (this information is displayed in Figure 3). Consequently, the total numbers of species per column (or row) do not add up to those in Table 1, List Three.

## Endemic species

Endemism is the limited geographical range of a species but it also serves as the acknowledgement that it is the sole responsibility of the province, region or country to protect those species that only occur there. The smaller the distribution range of a species, the fewer spatial options are available for its conservation. Only three species, the barebreasted goby (*Silhouettea sibayi*), slender puffer (*Torquigener marleyi*), and spotted longfin (*Plesiops multisquamatus*) of the 23 range restricted endemics (List One) within KZN were included on List Three. Most range restricted species had only single point localities, and some had taxonomic uncertainty associated with the specimens collected, and were therefore omitted from List Three, e.g. the nohorn unicorn (*Naso thorpei*) known from a single specimen “off Durban”. Many Delagoa, and Natal and Delagoa endemics were omitted because these species satisfied only one criterion or lacked habitat association information. These were



typically non-fisheries species which need further investigation (e.g. gobies, clingfish, pipefish, and tonguefish). Of the 85 endemic species on List Two, 43 species remained on List Three (Table 3). A large proportion (16 of the 19) of the Natal and Agulhas bioregion endemic species qualified for List Three, because many of these species have life history vulnerability, e.g. scotsman (*P. praeorbitalis*), and/or are species of conservation concern, e.g. seventy-four (*P. undulosus*), and/or are highly resident, e.g. zebra (*Diplodus cervinus hottentotus*) (see Appendix 2).

**Table 3. Number of species that qualify for the endemism criterion.**

<i>Endemism category</i>	<i>List One</i>	<i>List Two</i>	<i>List Three</i>
Range restricted	21	7	2
KZN provincial endemic	5	4	4
Natal bioregion endemic	1	1	0
Delagoa bioregion endemic	9	5	1
Natal + Delagoa bioregions endemic	16	16	4
Natal + Agulhas bioregion endemic	19	19	16
East coast endemic	33	33	16
<b>Total</b>	<b>104</b>	<b>85</b>	<b>43</b>

### Species of conservation concern

Forty-two of the seventy-six species of conservation concern remained on List Three (Table 1). Eleven IUCN threatened species were removed during the process of prioritization from List One to List Three (Table 4). For example, the southern bluefin tuna (*Thunnus maccoyii*) is currently listed as critically endangered on the IUCN Red List on the basis of rule A (Version 3.1), as it showed population reductions in the form of an index of abundance and actual potential levels of exploitation (Punt 1996). Similarly, other threatened pelagic species that qualified for List One did not remain on List Three owing to a lack of information. For example, swordfish (*Xiphias gladius*) which is listed as data deficient on the IUCN Red List (Safina 1996), but was placed on the NSBA list (Attwood in Lombard et al. 2004), did not qualify for any other criteria in this study. Some CITES-listed cryptic species are also IUCN data deficient (DD), e.g. giraffe seahorse (*Hippocampus camelopardalis*) (Project Seahorse

2003; CITES 2010). This suggests that pelagic and cryptic species would benefit from further research. In hindsight, some species that did not remain on List Three appear to have sufficient reasons to have been included but require a more detailed literature review for which the time constraints of this study did not allow, e.g. seahorses have vital parental care and typically display high site fidelity, which would be sufficient for these species to qualify for the criteria life history vulnerability and highly resident species (see Project Seahorse 2003).

**Table 4. Number of species that qualify for the species of conservation concern criterion.**

<i>Species list</i>	<i>Category</i>	<i>List One</i>	<i>List Two</i>	<i>List Three</i>
IUCN	CR	7	7	4
IUCN	EN	4	4	3
IUCN	VU	17	16	10
CITES	Appendix I	4	4	3
CITES	Appendix II	8	8	4
TOPS	P	8	8	8
TOPS	E	1	1	1
TOPS	VU	1	1	1
NSBA	-	12	12	11
Over exploited	-	13	13	7
Published concern	-	23	22	11

The South African TOPs list of species and the marine fish listed in NSBA 2004 (Appendix 2) inform the national conservation management of species and biodiversity assessments. The inclusion of local lists of species of conservation concern and published literature of concern and over-exploitation, collectively added 23 of species to List Three that were not IUCN or CITES listed, e.g. the scotsman (*P. praeorbitalis*) and galjoen (*Dichistius capensis*). The advantage of adding national lists to the criterion is the addition of several indigenous and

nationally important species or endemic species such as the seventy-four (*P. undulosus*), and red steenbras (*Petrus rupestris*) (Appendix 2). Many of these indigenous and nationally important species do not qualify for the IUCN Red List of Threatened Species owing to a shortage of historical abundance estimates, e.g. the catface rockcod (*E. andersoni*) (Fennessy 2004). Species that are internationally and locally threatened often overlap in the various lists including TOPs, IUCN or CITES, e.g. the sawfish (*Pristis* spp.), the coelacanth (*Latimeria chalumnae*), and the great white shark (*C. carcharias*) (Table 5).

**Table 5. Pair wise comparison of List Three species that met different qualifying conditions for the species of conservation concern criterion.**

<i>Number of species that overlap (N)*</i>	<i>CITES</i> (7)	<i>TOPS</i> (10)	<i>NSBA</i> (11)	<i>Over-exploited</i> (7)	<i>Published concern</i> (11)
IUCN (17)	5	6	3	0	2
CITES (7)		5	2	0	0
TOPS (10)			5	0	1
NSBA (11)				0	1
Over-exploited (7)					1

\*(N) = Number of species that qualified. Note that a species can meet multiple qualifying conditions. Lower values indicate a low overlap between qualifying conditions.

## Species with Life history Vulnerability

Twenty-nine of the 45 species that have life history vulnerabilities remained on List Three (Table 6).

**Table 6. The number of species with different life history vulnerability traits.**

<i>Life history vulnerability trait</i>	<i>List One</i>	<i>List Two</i>	<i>List Three</i>
<b>Reproductive traits</b>			
High age at maturity/Slow growth and longevity	10	10	8
Species that aggregate to spawn (these areas are targets for fishing)	19	19	9
Low fecundity/Small litter size (elasmobranchs)	9	8	4
<b>Behavioural traits</b>			
High catchability	11	11	8
<b>Total</b>	<b>38</b>	<b>37</b>	<b>29</b>

Two main life history vulnerability traits were defined: reproductive traits, and behavioural traits. List Three contained 19 species that are vulnerable because of particular reproductive traits - these species included seven sea breams (*Sparidae*), four kobs (*Sciaenidae*) and three sharks (*Carcharhinidae*). List Three also contained eight species that are vulnerable because of particular behavioural traits, including three groupers (*Serranidae*) and two sawsharks (*Pristis*) which are considered highly catchable.

As more detailed information becomes available on the life history characteristics of marine fish species, many more will probably qualify as vulnerable as described here. This approach has been developed by Cheung et al. (2004) and it appears to be an efficient method of identifying vulnerable species in the absence of full conservation assessments. Cheung et al. (2004) use size as a variable (amongst others) to estimate intrinsic extinction vulnerability. Size may be particularly useful to predict life history vulnerability because it is easy to measure, and is known for many species (see Mann 2000). It may be a good proxy

for other intrinsic vulnerability characteristics that are harder to distinguish (e.g. slow growth, longevity, and low fecundity).

Species that undergo sex reversals are especially vulnerable to fishing pressure because a loss of the larger size class of the species (as per South African fishing regulation) not only results in a loss of most fecund mature adults (Palumbi 2004; Field et al. 2008) but has the added effect of skewing the sex ratio and has further detrimental effects on the reproductive capacity of the species (Garratt 1985; Garratt et al. 1994). This is particularly prevalent in endemic species from the Sparidae as well as several Serreanidae (rockcods) (Garratt 1985; Audibert et al. 1989; Garratt et al. 1994; Heemstra & Heemstra 2004).

### Highly resident species

On List Three, eleven species qualified for this criterion: five Sparidae, three Serranidae, two Sciaenidae and the puffadder shyshark (*Haploblepharus edwardsii*). Three highly resident species, the catface rockcod (*E. andersoni*), riverbream (*A. vagus*), scotsman (*P. praeorbitalis*), each qualified for four criteria (Figure 3).

The criterion's qualifying conditions required a species to move a short average distance of 15 km to be considered as highly resident. This low average distance resulted in species generally considered as resident, such as galjoen (*D. capensis*) (Attwood & Bennett 1994) not qualifying. The criterion could be improved by amending the qualifying conditions accordingly, e.g. less than 20 % of recaptures move more than 5 km (C. Attwood, pers. comm.).

### Estuarine Species

Several of the endemic gobies, e.g. the baldy (*Caffrogobius natalensis*), naked goby (*Croilia mossambica*) and barebreasted goby (*S. sibayi*) (See Appendix 1), are estuarine dependent. These species are small and not of fisheries importance and not often explicitly considered worthy of conservation efforts. These species are, however, under threat from estuarine pollution, sedimentation and water abstraction practices like agriculture, mining and domestic use throughout their range in KZN (Harrison 1999; Forbes et al. 2008; SADC 2008). Given their dependence on this specialized habitat and their limited distribution, these species are likely to be affected adversely by anthropogenic or natural stochastic events. Estuarine associated species like garrick (*Lichia amia*) and flathead mullet (*Mugil cephalus*) that did not qualify for inclusion in SeaPLAN were, however, included in the provincial estuarine plan, where a wider selection of species was included.

## Rare species

Sixteen of the 70 rare species remained on List Three. Most (13) of the 16 species qualified as widespread rare species (never common where found although they are widely dispersed). For example, the porcupine ray (*Urogymnus asperrimus*) occurs throughout the Western Indian Ocean and the Red Sea, but is not common anywhere. The definition of having fewer than five localities (an area that is sufficiently isolated from genetic influx) in the area of interest was adopted from the IUCN Red Listing Criteria. In the marine environment little is known about the genetic structure and isolation of various populations (von der Heyden 2009). Many of the rare species were not included beyond List One because they had fewer than ten specimens collected. Most of the removed species qualified as range restricted endemics or rare species for List One.

Authors differ in their treatment of species that lack information: The IUCN lists species that lack adequate information to warrant listing as data deficient (DD), while other efforts to prioritize species for conservation efforts (Junor 1992; Lamberth & Joubert 2005) increase the importance of species for which little is known. During SeaPLAN a precautionary approach was favoured, in which species for which information is lacking were included unless regarded as true vagrants and/or common species that suffer from misidentifications. Species with only one or two highly uncertain records could be the result of identification error, e.g. a single record for the dragon stingray (*Himantura draco*), recorded off the KZN coast, has been questioned by the experts (S. Dudley pers. comm.; Froese & Pauly 2009) and was later changed to the widespread stingray (*Himantura jenkinsii*) (Heemstra & Heemstra 2004).

Species for which fewer than ten specimens exist could either be truly rare or range restricted endemics, or rare vagrants collected outside of their range. The risk to conservation planning outcomes of excluding true rarity could lead to non-representation of a critically important species, while including false rarity could result in misguided and wasted conservation efforts (Malcolm & Hutchinson 1994). Expert advice was sought to identify species with fewer than ten specimens that should be included in SeaPLAN. This resulted in the removal of all such species, except the slender puffer fish (*Torquigener marleyi*).

## Species dependent on specialised habitats

Table 1 shows that only four species on List Three qualified for this criterion 7. This can be attributed to the lack of fine scale habitat and process information, as well as more detailed information on species' habitat associations. The lack of species dependent in this criterion

was also observed in Junor (1992) where similar qualifying conditions were met by only a few species.

### **Other criteria not considered in SeaPLAN**

Several other criteria not considered in SeaPLAN may also warrant increased conservation effort, e.g. species that are currently fished (either recreationally or commercially), or have functional, taxonomic or genetic diversity.

### **Fished species**

Lamberth & Joubert (2005) used fishery parameters such as abundance and exploitation throughout range to inform prioritization of linefish species for research, conservation and management. Such fishery parameters are less applicable to non-harvested species and were not used as criteria in this study. Studies that evaluated species of conservation concern were used to inform the criteria used in this study (e.g. Lamberth & Joubert 2005).

This study did not have fishery objectives. Management of harvested species relies on more than just spatial protection, and includes such measures as closed seasons, bag limits and size limits. This study does not intend to inform management regarding catch and control regulations, but rather to identify the species that would not automatically benefit from the protection of percentages of surrogate habitats (i.e. 'species that would fall through the net'). Many harvested species are common and widespread (i.e. do not 'fall through the net') and do not currently qualify for List Three, e.g. sardines (*Sardinops sagax*). Many non-qualifying harvested species automatically benefit from the spatial conservation efforts directed towards those species that did qualify because of overlaps in their distribution ranges. On the other hand, several harvested species did qualify for List Three, e.g. the slinger (*Chrysoblephus puniceus*) and shad (*Pomatomus saltatrix*).

### **Functional or ecological roles of species**

Including functional group diversity could be important to ensure representation of trophic level diversity and especially the ecological processes that they support (Frank et al. 2005; Knowlton & Jackson 2008). Keystone predators, e.g. the red steenbras (*P. rupestris*), and community building species (e.g. herbivores from coral reefs) maintain the biodiversity balance and energy flow of the trophic foodweb (Toral-Grande et al. 1999; Bascompte et al. 2005). Ecologically important species like sardine (*S. sagax*), chub mackerel (*Scomber japonicas*), pinky (*Pomadasys olivaceum*) and grey grunter (*Pomadasys furcatum*) form a corner stone of the food web and provide nutrition for many species. The processes that maintain ecosystem function can break down prior to the onset of biodiversity loss (Knowlton

& Jackson 2008) and therefore warrant further consideration in setting conservation priorities (Toral-Grande et al. 1999; Bascompte et al. 2005).

### **Taxonomic distinctiveness**

The aim to conserve biodiversity requires some knowledge of how diverse a species is relative to another, with the idea that priority should be given to species with fewer closely related relatives. Previous studies have recommended that a taxonomic distinctiveness or biodiversity index (BI) be calculated. This can be expressed, for example, as the inverse of the product of the number of branches at each node of genera, family and order in a phylogenetic tree (Daniels et al. 1991; Vane-Wright et al. 1991; Joseph et al. 2009). The Zoological Society of London's conservation programme project: (EDGE) Evolutionary Distinct and Globally Endangered, aims to identify such species, particularly if they are also threatened (EDGE 2009).

### **Genetic diversity**

Considering genetic diversity is the obvious next level of diversity and is important because it represents evolutionary history. It has been shown that subpopulations with different genetic structures exhibit different residency behaviour in galjoen (*D. capensis*) (Attwood & Bennett 1994). The use of genetic analyses to understand the movement between subpopulations has recently been pointed out by von der Heyden (2009), which may help understand how isolated subpopulations are in terms of the main population.

## **Recommendations and lessons learned**

1. Lists of species that are important in a spatial conservation plan should include those species that would not be receiving protection unless specifically included, i.e. the species that fall through the habitat and process net. It is therefore important to identify the criteria that limit the species' conservation options.
2. Criteria to prioritize species for conservation planning should be developed by selectively combining different existing and proposed criteria. Using only one method of prioritisation (e.g. threatened species lists) results in the exclusion of too many species.
3. Projects that aim to prioritize species stand to benefit from criteria tailored to match the project objectives because of the mismatch of existing prioritization criteria and project objectives.



4. Keep it simple: There are more criteria than time would allow for assessment. The criteria can be refined *ad infinitum*, so tailor the number of criteria to the project's time limits.
5. It may be more productive to spend time doing a detailed literature and data review of several species not currently included, than to ever-refine selection criteria.
6. It is important to build on our knowledge of fine-scale habitat and species associations such that species that may potentially not be represented by broad scale habitat and processes alone can be identified.
7. Increased research in population dynamics and conservation status of pelagic and cryptic species stand to benefit these species for which little is known and probably deserve more than the current level of protection.
8. Species with life history vulnerabilities requires further investigation (e.g. species that undergo sex changes).

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## Chapter 2

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# Spatial and temporal resolution of marine fish data in KwaZulu- Natal

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## Abstract

Information on the distribution of biodiversity features was required for species distribution modelling and conservation planning. In this chapter, available data sets were evaluated for spatial and temporal (spatio-temporal) uncertainty to identify suitable data that met the scale requirements of SeaPLAN. Data sets examined were: the National Marine Linefish System (NMLS); Oceanographic Research Institute/World Wildlife Fund-South Africa tagging data, KwaZulu-Natal Sharks Board data sources; and data from the online data repository, Ocean Biodiversity Information System which included the South African Institute for Aquatic Biodiversity (SAIAB) and Iziko Museum data. SAIAB held data for the largest number of species (50), but only a few (< 10 on average) records per species, while the NMLS held the largest number of records (658131) but for fewer species (36). A small proportion (18%) of records had a spatial resolution of one km, and was suitable for fine-scale distribution modelling. A large proportion (68%) of the data had a spatial resolution greater than 4 km, and was spatially not suitable for distribution modelling. The majorities of the data were recorded to day-level (99.98%) and met requirements for modelling distribution ranges. I

recommend that future data collection includes the use of handheld data recording devices (e.g. GPS devices) to increase the spatial and temporal resolution of data recorded, minimise human error, and improve data flow efficiency.

**Table 1. Reference list of acronyms used in text (in alphabetical order).**

<i>Acronym</i>	<i>Full Name</i>
AfrOBIS	Sub-Saharan Node for OBIS (see OBIS)
CSIR	Council for Scientific and Industrial Research
DCO	District Conservation Officer
EEZ	Exclusive Economic Zone
EKZNW	Ezemvelo KwaZulu-Natal Wildlife
GBIF	Global Biodiversity Information Facility
GPS	Global Position System
Iziko	Iziko Museum data, includes SAMC & SAMS
KZN	KwaZulu-Natal
KZNSB	KZN Sharks Board
MCM	Marine and Coastal Management
MODIS	Moderate Resolution Imaging Spectroradiometer
NMLS	National Marine Linefish System
OBIS	Ocean Biogeographic Information System
ORI	Oceanographic Research Institute
ORI/WWF-SA	Oceanographic Research Institute/World Wildlife Fund - South Africa
SAIAB	South African Institute for Aquatic Biodiversity
SAMC	South African Museum Fish
SAMS	South African Museum Sharks
SST	Sea Surface Temperature
TDWG	Taxonomic Database Working Group
UCT	University of Cape Town

## Introduction

Conservation planning requires information on the distribution of biodiversity features, like species distribution ranges (Margules & Stein 1989; Guisan & Zimmermann 2000; Margules & Pressey 2000; Ferrier 2002). The fish species prioritized for SeaPLAN (see Chapter 1), were included in the plan as biodiversity features. However, the lack of complete data on fish distributions required that their ranges be modelled. Species distribution modelling requires not only the survey and environmental data, but also explicit knowledge on the spatial and temporal resolution of the data as this determines the resulting distribution model's resolution and reliability, and affects the accuracy of the conservation plan (Morrison 1995; Guisan & Zimmermann 2000). Higher resolution data (i.e. low spatial and temporal uncertainty) produce more accurate distribution models. The trade off to be considered, however, is that as the required resolution increases, the amount of data that qualifies decreases. Historical data collections are typically coarse, especially those that span large planning areas, because these collections started before the advent of Global Positioning Systems (GPS) devices, and involve several people, including lay people, for several of the South African marine fish data sets.

Species data are typically recorded to a place (locality) and a time. The spatial uncertainty of the data depends on the specificity of the information recorded (e.g. town name, landmark, well-known offshore reef, size of intervals between coded localities, and GPS coordinates). Temporal resolution can include different levels of time specificity, for example, minutes, hours, days, months, and years. I evaluated the data that were available for modelling fish distributions for spatial and temporal uncertainty (spatio-temporal resolution) to determine which of the available data sets satisfy the scale requirements for modelling ranges for SeaPLAN. I used the resolution of the planning units used in SeaPLAN (1 km<sup>2</sup>) and the environmental data available (e.g. bathymetry 0.5 km<sup>2</sup>, Moderate Resolution Imaging Spectroradiometer (MODIS) Sea Surface Temperature (SST) 4 km<sup>2</sup>) to inform the resolution requirements for the data that I assessed in this chapter.

Ad hoc historical data collection methods have resulted in several different data formats and fields of information being recorded. In an effort to address the problems associated with non-standardised data frameworks, biodiversity information standards were established by the TDWG (formerly known as Taxonomic Database Working Group). The TDWG developed an international data storage framework, so that a consistent format is available worldwide. The framework removes data transfer obstacles among organisations that have adopted the standard format, as well as among the online data repositories such as Sub-Saharan Node

for the Ocean Biogeographic Information Facility (OBIS), (AfrOBIS), and the Global Biodiversity Information Facility (GBIF). International biodiversity data standards require locality information to include geographical coordinates, along with their spatial uncertainty. All data also require the date, amongst other mandatory minimum fields of information (TDWG 2009). The data available for marine fish species in KwaZulu-Natal (KZN) are collected by several independent programmes (Figure 1), and few have adopted the biodiversity information standards. Below I discuss the data and the methods used to collect them. Only data for species identified for SeaPLAN, and within the study area (the KZN Exclusive Economic Zone (EEZ)), were considered.

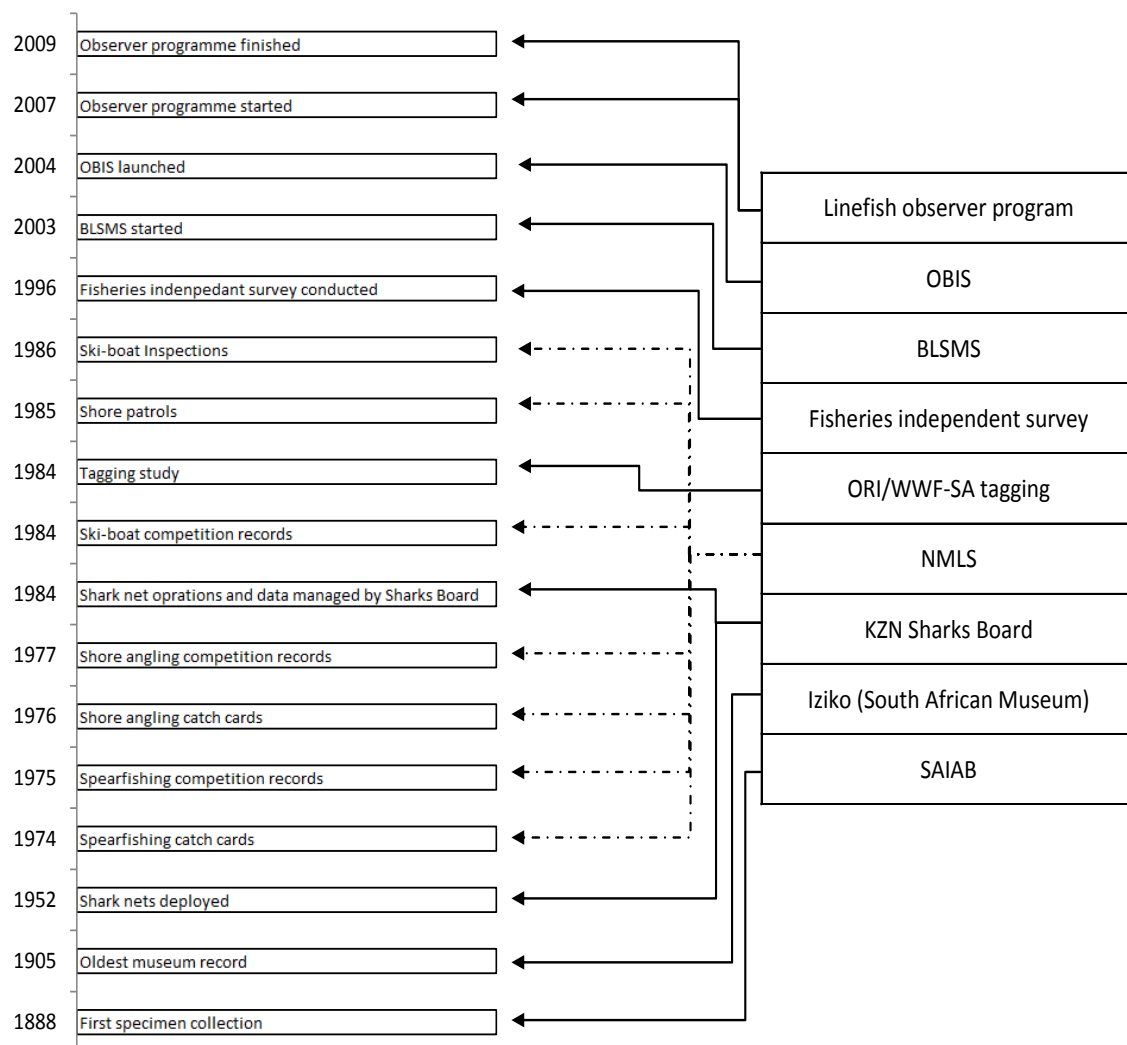


Figure 1. Non-scaled timeline of the historical developments of marine fish data collection programmes in South Africa.

The South African Institute for Aquatic Biodiversity (SAIAB) was started by Drs J.L.B. Smith and M.M. Smith and holds some of the earliest (1888) fish specimens in South Africa. The

data collection includes specimens from all over the world, but is focussed on southern Africa, and includes a wide range of teleost and elasmobranch species. SAIAB holds the largest fish specimen collection in the world and includes specimens, DNA, observations and images (W. Coetzer pers. comm.). Data and specimens have been collected opportunistically by anglers, scientists, and students and submitted on an *ad hoc* basis to the institute. Species identification is very good, as often the data are accompanied by photos or the specimens are sent to SAIAB for identification and storage (W. Coetzer pers. comm.). The data collected on an *ad hoc* basis are known to be less reliable in terms of locality when compared with the data collected by SAIAB researchers (W. Coetzer pers. comm.). Data can be obtained via the web portal (<http://saiab.ac.za/infoportal/>) or on request from SAIAB. SAIAB recently (2006) adopted the international biodiversity data standards (TDWG 2009) for biological field information. The standards to which data have been recorded and the recent development of an online Geographic Information System (GIS) for SAIAB's freshwater component provide a valuable tool for conservation assessments (Darwall et al. 2009). The marine data have only had limited use for modelling marine fish distributions in South Africa (see James et al. 2009), but have been used for fine-scale movement studies (Mann et al. 2010).

Iziko Museum (South African Museums, Cape Town) has been collecting and accessioning marine fish specimens into their collections since 1901. These specimens have been collected through internal or external exploits, e.g. Marine and Coastal Management (MCM<sup>1</sup>), University of Cape Town (UCT), KwaZulu-Natal Sharks Board (KZNSB), etc. (W. Florence pers. comm.). The data were accessioned with associated data (locality, collector, date of collection, etc.) in most cases. Records from different sources used different locality recordings systems (e.g. GPS positions, detailed locality descriptions, beach locality code names, town names, and province names) and were of different spatial resolutions. Iziko Museum holds data for a wide range of species, including sharks (elasmobranchs) known as South African Museum Shark (SAMS) data, and bony fish (teleosts) known as the South African Museum Fish (SAMC) data, because its research focus is based on both taxonomy and systematics.

The first coordinated effort to collect marine fish catch and effort data on a continuous basis in South Africa was initiated by a group of linefish researchers in 1974 (Maggs 2008). The system became known as the National Marine Linefish System (NMLS). The NMLS records data from multiple sectors for linefish species and is primarily used for reporting on catch

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<sup>1</sup> MCM has subsequently been split into two departments, Department of Agriculture, Fisheries and Forestry (DAFF) and Department of Environmental Affairs (DEA), but is referred to as MCM here, as changes to departmental names had not yet been updated for data set sources at the time of writing.

statistics, typically catch and effort (Mann-Lang 1996). All NMLS data are coded for locality, species, and fishery sector, prior to being entered into the database housed at MCM (Maggs 2008). Landmarks, well-known offshore reefs, place names, locality codes or GPS positions are recorded by voluntary participants or during inspections that are then submitted to the Oceanographic Research Institute (ORI) or MCM (commercial data) (Figure 2) where the data are coded to conform to the NMLS database requirements. The localities are assigned a code based on a one kilometre (km) interval along the southern African shoreline, starting at the Mozambique-Tanzania border (code 0), around South Africa, to the Kunene River (code 8082) on the Namibia-Angola border, and are known as the beach locality code system (Mann-Lang 1996; Maggs 2008). Only some of the one km codes of this system are used, and therefore not all km codes have data assigned to them (J. Maggs pers. comm.). The codes may be accompanied by a distance offshore to represent a well-known offshore reef or fishing spot. The beach locality code system has been adopted by various data collection efforts like the ORI/WWF-SA (Oceanographic Research Institute/World Wildlife Fund-South Africa) tagging study and the KZNSB. The different data collection programmes and the gear types of the various fisheries influence the accuracy of the beach locality code system and are discussed below.

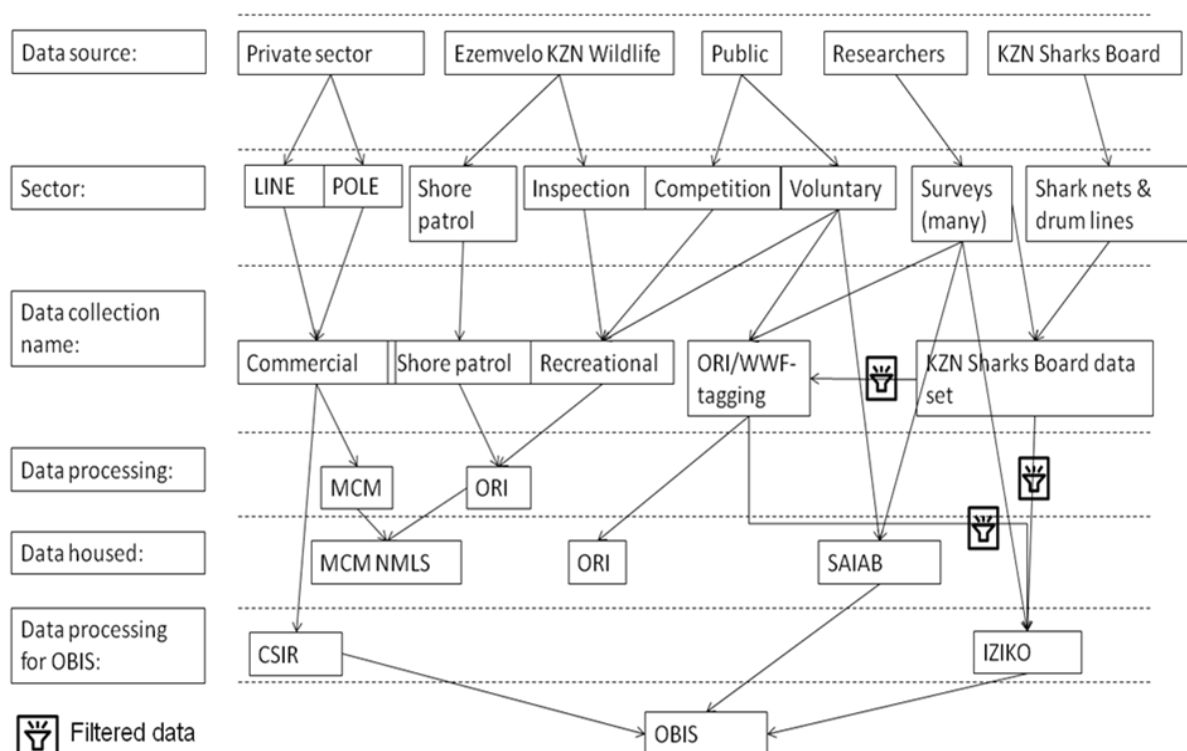


Figure 2. Flow diagram showing the complex pathways the data analysed in this study follow, from collection to storage.

Coordinated shore patrols are carried out by Ezemvelo KwaZulu-Natal Wildlife (EKZNW) staff in KZN, and form the bulk of the NMLS data. Shore patrols attempt to record biodiversity data from recreational shore and estuarine angling while policing poaching in the KZN. During a shore patrol, EKZNW staffs walk a patrol-beat along a section of beach within the area designated to the 16 district conservation officers (DCOs) in the province (J. Maggs pers. comm.; C. Van Tichelen pers. comm.). The patrol's beat distance varies considerably from one DCO to the next. For example, EKZNW staff may be stationary at Durban piers during the shad (*Pomatomus saltatrix*) run, while long stretches of coastline may be covered at other times of the year. All biodiversity data (fish) are recorded to a unique patrol beat number, which includes a start and finish beach locality code, date and time of day.

Data that are submitted voluntarily or collected by EKZNW during spot inspections of recreational shore, estuarine and ski-boat angling are recorded differently from shore patrol data. When the NMLS was first started, only catch cards submitted on a voluntary basis by recreational spear-fishermen, shore- and estuarine-anglers were recorded (Mann-Lang 1996). In 1984 the system was expanded to include shore, estuarine and ski-boat inspection data, collected by EKZNW (Mann-Lang 1996) (Figure 1). Each inspection or voluntary submission may be recorded as a name (e.g. Richards Bay) or a beach locality code (e.g. 3799) prior to being checked and coded by ORI. The recreational shore angling data are recorded to a beach locality code and do not include a distance offshore, while estuarine data are recorded to the estuary. Similarly, recreational ski-boat data are submitted voluntarily or recorded during an inspection by EKZNW to a locality code or name before being coded by ORI. The distance offshore is recorded only if the locality is a well-known offshore fishing locality. The distances offshore of 'well-known fishing grounds' were established by National Sea Fisheries (later became MCM) during the early expansion period (1984 – 1985) of the NMLS when fisheries managers accompanied and liaised with commercial fishermen from Durban (B.Q. Mann & J. Maggs pers. comm.). The distance offshore of well-known fishing grounds is assigned to data reported from such a locality by ORI. Data without distance offshore may be accurate in terms of their beach locality code (latitude), but their distance offshore (longitude) is unknown. Data that are recorded to ski-boat launch sites are less accurate because neither the actual beach locality code (latitude) nor the distance offshore (longitude) were recorded.

Commercial fisheries in KZN are smaller than the rest of the South Africa, owing to less productive ecosystems sustaining fewer commercially exploitable fisheries (Branch & Branch 1983). Only NMLS commercial ski-boat data were analysed for this study. Commercial catches are recorded daily and declared monthly at port to EKZNW staff, who then send the

data to MCM for coding, control and capture (B.Q. Mann & J Maggs pers. comm.). Similar to recreational ski-boat data, the commercial line-fish data are coded for species, fishery and locality and a distance offshore is assigned to the records based on the well-known fishing grounds for different species. The commercial data are recorded to quarter-degree grid cells. The well-known fishing localities within these quarter degree grid cells are known for different fish species. The commercial catch data are then assigned to the well known fishing localities based on the quarter degree grid cell and fish species caught (Y. Snyder & C. Wilkie pers. comm.).

In 1984 the ORI/WWF-SA tagging project was initiated to provide tag and recapture data which are used for research on the migration routes, growth rates, stock identity and population dynamics of important linefish and elasmobranch species (Mann & Bullen 2009). Tagging takes place around the South African coastline, including volunteers and highly active nodes where tagging programmes are proactively undertaken by scientists, like Cape Vidal in KZN and De Hoop in the western Cape (Mann et al. 2010, B.Q. Mann & J. Maggs pers. comm.). Participants are required to have their credibility vouched for by fellow anglers prior to acceptance into the programme (Mann & Bullen 2009). The data are focussed on important linefish and elasmobranch species, and exclude species and individuals that are smaller than 30 cm (or less than 0.5 kg) (Mann & Bullen 2009). Capture and recapture locality information is recorded and coded to the beach locality code system, as for the NMLS recreational inspection data. No distinction is made between ski-boat and shore angling data in the ORI tagging database. The data are available as periodic data reports (see Figure 2), popular articles, scientific papers and regular newsletters for participating anglers (Mann & Bullen 2009).

Shark nets were deployed in 1952 following a spell of shark attacks in KZN, known as black December, but have been contributing reliable data only since 1984, when the deployment and data collection responsibilities were transferred from independent contractors to the KZNSB (Dudley & Simpfendorfer 2006). This database is used for biological studies, shark net impact assessments and balancing beach user safety with shark mortality (see <http://www.shark.co.za/overview.htm>). Shark nets have semi-permanent positions, and are anchored at 300 – 400 m offshore at specific beaches from Mzamba in the Eastern Cape to Richards Bay in central KZN. Released sharks are also tagged and contribute data to the ORI tagging database. The data are recorded to beach locality codes as for NMLS recreational shore angling data and include the date.

KZNSB and ORI record GPS coordinates of whaleshark (*Rhincodon typus*) and other whale sightings during aircraft flights along the KZN shore. The collection of data is dependent on



availability of resources to conduct the flights. The data include GPS coordinates and the date, amongst other biodiversity and flight data.

The Census of Marine Life (CoML) project has been making some of these data sets available online (e.g. SAIAB, Iziko, NMLS commercial). CoML's primary concern is the distribution of global marine biodiversity (Grundlingh et al. 2007; CoML 2010). The OBIS System (OBIS, [www.iobis.org](http://www.iobis.org)) is the data handling and inventory hub of the CoML's projects. OBIS offers the user the advantage of instant access to, currently, 22.2 million records of 112000 species from 744 databases and has a graphic support system to allow visualization of the distribution of marine biodiversity (Grundlingh et al. 2007; AfrOBIS 2010). The Sub-Saharan node of OBIS, AfrOBIS, contains over 3.4 million marine species records and is hosted by the Council for Scientific and Industrial Research (CSIR) who coordinated the transition of data from online data contributors, like MCM, SAIAB and Iziko museum, to be standardized to the required format - see <http://rs.tdwg.org/dwc/index.htm> (TDWG 2009). The standardization of data fields, including georeferenced coordinates, and its instantaneous availability, removes hurdles to data transfer between organisations and potentially offers substantial time saving and data sharing opportunities. These benefits provided the incentive for fish distribution modelling in SeaPLAN. It is tempting to use the vast resources (including coordinates) of readily available data as is, but there is a risk of large scale spatial errors and misinterpretation of the data (see Robertson 2008; Robertson et al. 2010). Typical problems are the spatio-temporal accuracy of the data point in relation to the underlying habitat, and the scale to which the data are accurate (Robertson et al. 2010).

The amalgamation of various data sets into a larger database required the standardization of information fields (Grundlingh et al. 2007). OBIS accepts only georeferenced data (i.e. with latitude and longitude). The South African georeferenced data on OBIS are checked for positional errors using an independent bathymetric map and visual inspection by the CSIR (Grundlingh et al. 2007). The data available on the OBIS webpage are quality controlled at regular intervals but not peer-reviewed prior to online publication. OBIS recommends users undertake the necessary inspection of the data and gain an understanding of the data and metadata prior to use (see <http://afrobis.csir.co.za/AfrOBISContributors/providers/data/policy/disclaimer>).

Many of the South African records were not recorded using a GPS, but rather just a locality name, which was subsequently georeferenced for inclusion in OBIS. SAIAB, Iziko and NMLS commercial data were allocated quantitative map coordinates by the data contributors or by the CSIR using maps and gazetteers of unknown scale (W. Coetzer, pers. comm.);

Grundlingh 2009; Wilkie 2009). The coordinates assigned to South African marine fish data sets in KZN were alleged to be accurate to 100 m resolution (AfrOBIS 2010). Therefore, many of these do not reflect the true spatial accuracy of the data. The risk of greater hidden spatio-temporal uncertainty of the data owing to its complex underlying nature, could lead to misinterpretation and misuse of the data for applications for which it is unsuitable. I required the data for species distribution modelling (Chapter Three), and therefore it was necessary to evaluate its true spatio-temporal uncertainty. I assessed the uncertainty of the coordinates and dates, of the South African marine fish data provided to OBIS by MCM, SAIAB, and Iziko museum.

The shortcomings and strengths of using the different kinds of data are discussed in this chapter to establish which of the data sets meet the minimum standards required for the species distribution modelling. Valid records that satisfy the spatial resolution required for modelling species distributions were identified and prepared for distribution modelling following chapter. Recommendations of how to improve data collection efforts and data quality (e.g. spatial, temporal resolution, etc) are discussed. Several databases (e.g. boat launch site monitoring data) were not considered in this study owing to the limitations of the study or obvious shortcomings of the data for distribution modelling, but some additional data sets were included in the discussion and in the qualitative evaluation in Appendix 4.

## Methods

### The data evaluated

The data were evaluated for spatial and temporal resolution and the numbers of species per data set were reported, for the following data sets:

- SAIAB
- Iziko Museum
- NMLS
  - Recreational shore angling
  - Recreational ski-boat angling
  - Shore patrol
  - Commercial linefish
- ORI/WWF-SA tagging data
- KZNSB data
- KZNSB and ORI flight data

I analysed a subset (SeaPLAN species in KZN) of the full data sets for spatial and temporal uncertainty (resolution). No estuarine data were considered in this study. Only the 67 species that qualified for List Three in Chapter One are presented in the results. The analysis was conducted separately for each of the data sets as the different data collection and recording methods required individual analysis. The common treatments that are repeated throughout data sets are described below, and then referred to within the methods per data set. The nested data are discussed and treated per data set.

A record was regarded as a species caught at a particular locality during a single data capturing event - I did not multiply the number of fish per species caught during a single fishing event for two reasons. Firstly, recording the number of fish of the same species caught during one single event might skew the resolution of the data towards that of shoaling species (e.g. shad (*Pomatomus saltatrix*)) which are caught in large numbers at stationary points therefore artificially improving the spatial resolution of the data. Secondly, the modelling approach that was used in the following chapter did not rely on abundance data.

### **Spatial uncertainty**

I considered two meaningful scales to evaluate the point data against that would be appropriate for distribution modelling in the following chapter based on the resolution of environmental data: 1) one km<sup>2</sup> and 2) four km<sup>2</sup>. One km<sup>2</sup> was planning units used in SEAPLAN, whereas the MODIS Aqua remotely sensed environmental data are at ca 4 km<sup>2</sup> (Feldman 2009). Ski-boat data were evaluated to five km, because this was the finest scale for ski-boat data. The environmental variables are discussed in greater detail in the following chapter.

### **Temporal uncertainty**

The numbers of records were counted for each of five temporal categories: no date, only year, month (and year), day (and month and year), and time (and day, month and year). It was known that the Iziko data obtained from KZNSB reflected the date the data were acquired and not the date collected, and were regarded as not having a date.

### **SAIAB and Iziko data via OBIS**

I downloaded data for marine fish species via the Sub-Saharan node of OBIS, (<http://afrobis.csir.co.za:8000>, 26 October 2009) for the following data sets: South African Institute for Aquatic Biodiversity (SAIAB), Iziko SAMC and SAMS; MCM - Linefish data set; - Fish Collection, and FishBase (downloaded on the 30 September 2008). The OBIS MCM-

Line data set is the same as the NMLS commercial linefish data set. I preferred to analyse the data set obtained from MCM because it included NMLS commercial linefish data up to end 2009. I removed the FishBase data set for KZN SeaPLAN species because it had been removed in the interim from AfrOBIS owing to high (95%) duplication of SAIAB data sets (R. Froese and N. Bailly pers. comm.). The online data were analysed per data contributor, but limited to SeaPLAN species in KZN.

### **Spatial and temporal analysis**

Two kinds of locality information were assessed for spatial resolution and collectively presented per data collector: (a) locality descriptive and, (b) coded locality data (the names of beach locality codes).

- a) The locality descriptive data included records from the SAIAB, Iziko SAMC and some Iziko SAMS data. I assigned spatial uncertainty estimates to different kinds of locality descriptions, with uncertainty increasing as locality-specific information decreased (Table 2). The estimated spatial uncertainty was based on the system used for scoring the accuracy of coordinates that are allocated to locality description data by EKZMW (Goodman & Escott 2010). The total numbers of records that pertain to a spatial uncertainty class were summed per data set to give an idea of the spatial accuracy associated with the data set.

**Table 2. Accuracy classes in meters associated with locality descriptive data**

<i>Spatial uncertainty class</i>	<i>Description or e.g.</i>	<i>Distance of uncertainty (= &lt; km)</i>
GPS - DD (6 decimals), after 1994	GPS - averaging or differential mode	0.05
Coordinates DMS, DD 4 decimals, or any GPS data before 1994	GPS - Single point mode or 1: 10 000 map	0.1
Specific, well identifiable small feature name or very particular locality in town	Aliwal Shoal: Cathedral reef	0.5
Distance in m from landmark or path in town	Park Rynie, 23 m off railroad station	0.75
Distance in km from landmark or path in town	2.5 km S of Boteler Point	1
Specific, well identifiable large feature name or small indistinguishable feature in larger area	Aliwal shoal; 7 mile reef etc;	1.5
Larger area in town	In front of named golf course	5
Small town/place name	Park Rynie	7.5
Large town/place name	Richards Bay	15
Large town – ‘area’	Durban - area	30
Vague direction from Large town, or area along coast	South of Durban	200
Province, NA, empty field, unspecified, etc	KwaZulu-Natal	600

b) Duplicated records (1999-03-15 KZNSB, and Cliff and Dudley (1992)) were removed from the Iziko SAMS data. I used the same method to calculate the spatial uncertainty of the coded data as for KZNSB data (see below). The spatial uncertainty of SAMS coded and descriptive localities are presented together in a single figure.

### **National Marine Linefish System (NMLS)**

I analysed spatial and temporal uncertainty for NMLS data collected in the period from 1985 – 2009. The NMLS data were grouped into the contributing sectors as follows: Recreational shore and ski-boat angling, Shore patrol data, and Commercial Line fish data.

### Recreational shore angling data

I treated the 'return types' (inspection, competition, voluntary submission) that make up the data set the same, as I could not confidently provide a quantitative spatial difference between them. The recreational data set was grouped according to 'gear type' used, as the spatial uncertainty differs between shore angling and ski-boat data.

I calculated the spatial uncertainty by summing half the distance from the preceding 'used' locality (only the subset of beach locality codes that are in use) to half the distance to the following 'used' locality (Figure 3). The distance between two used beach locality codes was calculated from the code value which reflects kilometer intervals along the beach. The more unused codes there are between two used locality codes, the greater the spatial uncertainty becomes. I assigned the calculated spatial uncertainty of a given beach code locality to all the shore angling data recorded to that code.

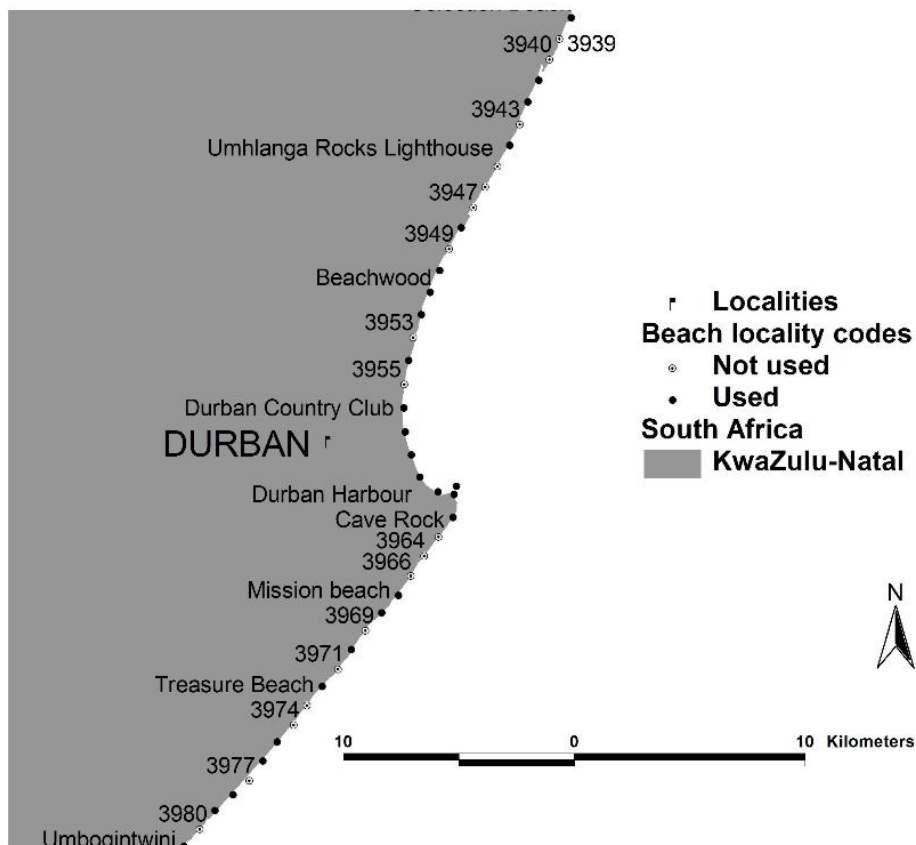


Figure 3. Section of KZN coastline (near Durban) showing used (black dots) and unused (grey dots) beach locality codes that were used to calculate the spatial uncertainty of data sets (not all localities are labelled).

### Recreational ski-boat data

I used the estimate provided by MCM (C. Wilkie pers. comm.) and ORI (B.Q. Mann & J. Maggs pers. comm.) for spatial uncertainty of ski-boat data for records that had both a beach

locality code and a distance offshore. Data with no distance offshore or only recorded to ski-boat launch sites were regarded as not useable.

### **Shore patrol data**

The distance of each shore patrol beat was calculated from the start and end locality code recorded for each patrol. I added half the distance from the locality code preceding the start locality, and half the distance to the locality code following the end locality to the patrol beat distance to account for the inaccuracy associated with locality codes at the edge of each patrol beat. Shore patrol data are recorded to the start and end times of each patrol. I counted the number of records per temporal resolution category, as follows: time interval, day, month and year.

### **Commercial linefish data**

Only commercial line fishing data were considered. Although commercial fishing in KZN includes other fisheries they were considered not substantial (e.g. beach seine netting) or did not include the selected species (e.g. tuna pole fishing). All ski-boat records were awarded the same spatial uncertainty estimate as that of recreational ski-boat data based on consensus from MCM and ORI scientists at the time.

### **ORI/WWF-SA tagging data**

ORI tagging data were analysed for SeaPLAN species. There were two kinds of data: Shore based and ski-boat. I distinguished between the two types of records by considering only nearshore species, defined as the subset of species that occurs only between 0 and 30 m. I considered only adult ranges for bony fish (teleost) because the tagging study records data only for specimens that are at least 5 kgs or 30 cm (Maggs & Bullen 2010). Species that occur in depths between 0 – 30 m only, and occur in greater proportions of NMLS recreational ski-boat data than shore angling data were removed because they were considered offshore species in this study. Species for which the data were insufficient were grouped based on the literature classification of the species (Heemstra and Heemstra 2004), e.g. potato bass (*Epinephelus tukula*). All species of tagged sharks that are commonly caught in shark nets, e.g. ragged-tooth sharks (*Carcharias taurus*) were included, because these species are mostly tagged after being caught in shark nets, for which I could estimate the spatial uncertainty (see below).

### **KZNSB data**

I obtained data for SeaPLAN species from the KZNSB (Dudley 2010). The data were assessed for spatial uncertainty associated to beach code locality, as for the NMLS recreational shore angling data.

### **Flight data**

GPS coordinates were obtained for the whaleshark (*Rhincodon typus*) from ORI and KZNSB. All are GPS coordinates and were assumed to be accurate to 100 m based on an accuracy scoring system used by EKZNW (Goodman & Escott 2010).

### **Qualitative analysis**

A descriptive account of the various data collection programmes was obtained through literature review, informal interviews and correspondence with respective data controllers, managers of data collection programmes, and regular data users. Examples of the various data collections were obtained from the respective institutions and organisations as well as data collection forms from the various data contributors. The discussion is focussed on the spatio-temporal aspects of the respective data sets and is presented in a metadata table (Appendix 4), following the format of Table 3.



**Table 3. The format and definitions of the metadata attributes assessed per data set, and presented in Appendix 4.**

<i>Attribute</i>	<i>Attribute description</i>
Data type	Recreational, commercial or bycatch
Species focus	Limited to particular suites of species
Research aims	Main objectives and uses of data
Data contributors	Sources of data
Extent of data	Geographical coverage of data
Data housed	Electronic facility storage of data
Spatial information	Code, descriptive, coordinates
Estimated spatial uncertainty	Spatial resolution based on expert opinion
Other Errors	Other errors associated to the data
Quality assessment	Conducted by, and detail
Reference data	Source to be referenced
Processing steps required	Specifically for spatial implementation
Online facility available from	Name of online facility, websites
Online data last updated	Continuous or year.

## Results

### SAIAB

The SAIAB data set held records for 50 of the 67 SeaPLAN species, but only a few records per species. For example, the data set held the most records (50) for the yellowbelly rock-cod (*Epinephelus marginatus*), but 76 % of the species had fewer than 10 records.

The minority (25 %) of the data records for SeaPLAN species had spatial uncertainty less than one km, while a further 10% had a spatial uncertainty of four km or less. High proportions (32 & 20 %) of the data were recorded to small town or large town level, respectively, and the majority (65%) of the data records were of a spatial resolution of greater than four km (Figure 4). The majority of the SAIAB data (69%) had a year, month and day associated with the locality and specimen, while 18% of the records had no date information (Figure 5).

## Iziko Museum

The Iziko data set available from OBIS held 278 records for 33 species, with few records per species, e.g. slinger (*C. puniceus*, 2 records) and includes rare species, e.g. green sawfish (*Pristis zijsron*, 1 record), but includes more records for some large elasmobranch species e.g. ragged-tooth shark (*Carcharias taurus*, 50 records) owing to internal research interests of the museum.

Collectively, 31% of the Iziko data records had a resolution of one km, while a further 38% had a spatial resolution of four km or less (Figure 4). The majority (83%) of the Iziko data were SAMS data, while the SAMC yielded 46 records for SeaPLAN species. Half of the SAMC records were recorded to large town level (e.g. Durban, Figure 4), with only 22% having a resolution of one km or less. The majority (63%) of the localities recorded for SAMS data matched the KZNSB data. The coded SAMS data (234 records) included 100 locality records that were obtained from KZNSB and 42 records from Cliff and Dudley (1992) (S. Dudley, G. Cliff & S. Wintner, pers. comm.). Collectively, a high proportion (72%) of the Iziko data had no date, of which the SAMS data included 83% and the SAMC data included only 17% with no date (Figure 5).

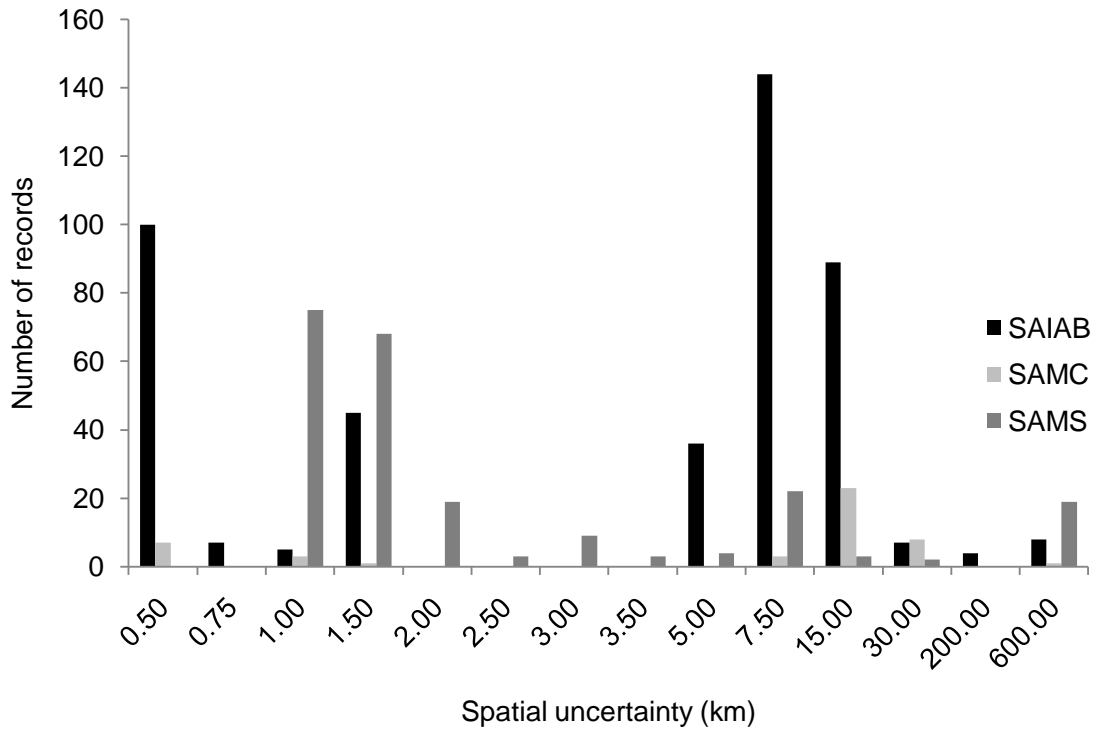


Figure 4. Spatial uncertainty of SAIAB and Iziko (SAMS & SAMC) data from the OBIS online database.

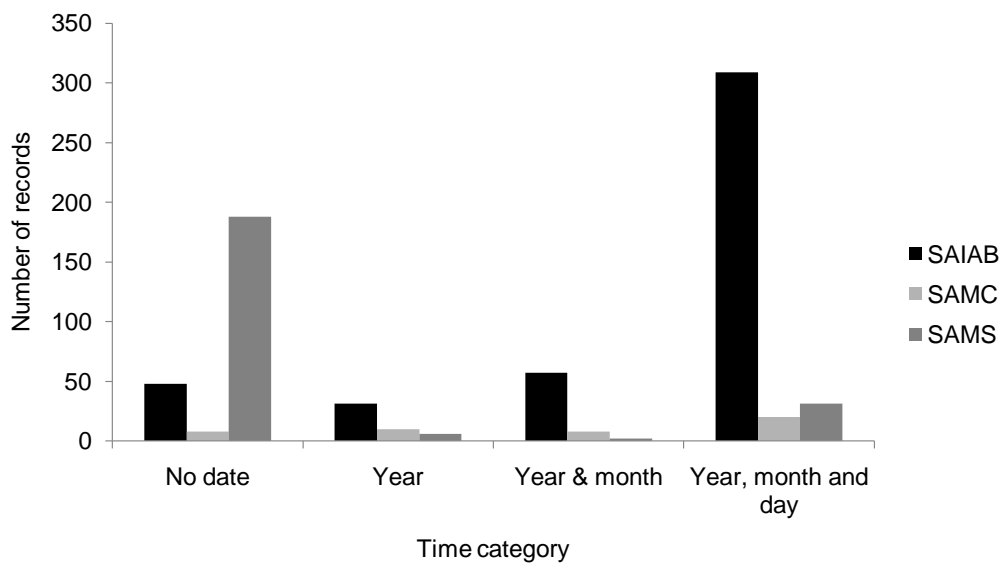


Figure 5. Temporal uncertainty of SAIAB and Iziko (SAMS & SAMC) data from the OBIS online database.

## National Marine Linefish System

### Recreational shore angling

The database included data for 36 of the 67 SeaPLAN species. The recreational shore data included 120961 records for these 36 species, of which 61% were accurate to one km and 95% accurate to  $\leq$  four km (Figure 6). Five percent of the data had a spatial resolution of more than four km and were regarded as non-useable for fine-scale modelling. All records included at least, year, month and day information, and of these 97% also included a time (Table 4).

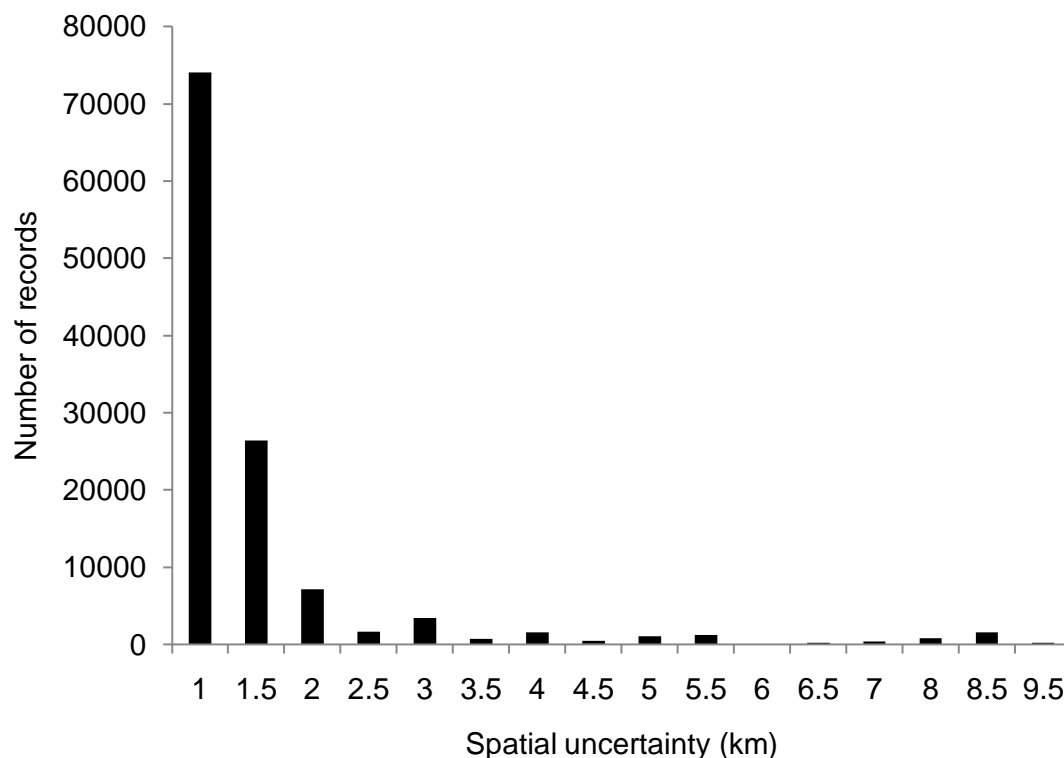


Figure 6. Spatial uncertainty of the NMLS recreational shore-angling data.

### Recreational ski-boat

The recreational ski-boat data held 75923 records for 29 SeaPLAN species, of which 32% were accurate to ( $\leq$ ) five km resolution (Figure 7). Sixty-seven % were accurate only to  $>$  five km and were regarded as not useable for spatial modelling purposes (Figure 7). Of the 75923 records only 28% did not have time data associated, while all data were recorded to include year, month and day information (Appendix 4).

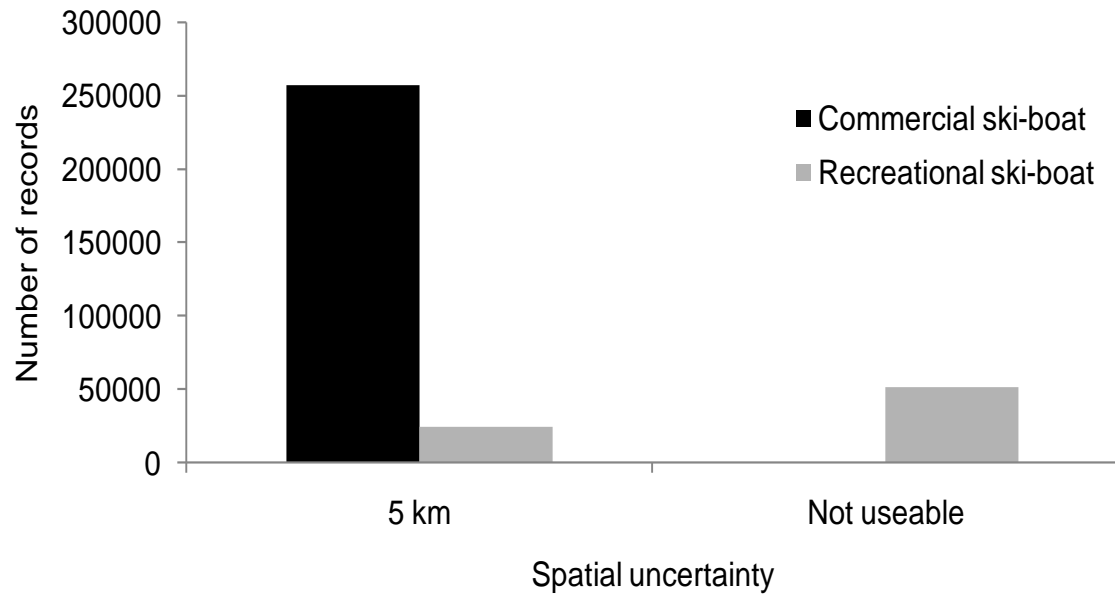


Figure 7. Spatial uncertainty of commercial and recreational ski-boat data.

### Shore patrol

The NMLS shore patrol data held 202228 records for 33 of the SeaPLAN species in KZN. The composition was dominated by shad (*Pomatomus saltatrix*) (71% of records), and blacktail (*Diplodus sargus capensis*) (19%) and dusky kob (*Argyrosomus japonicus*) (3%), collectively contributing to 93% of the data.

Twenty percent of the records were spatially accurate to one km, and an additional 30% were accurate to four km (Figure 8). A large number (8407) of records had a spatial resolution of between 25-100 km. Twenty-nine records had a spatial resolution great than 100 km. A very high proportion (99%) of the data was recorded to a time interval, and all records were recorded to include day, month and year (Table 4).

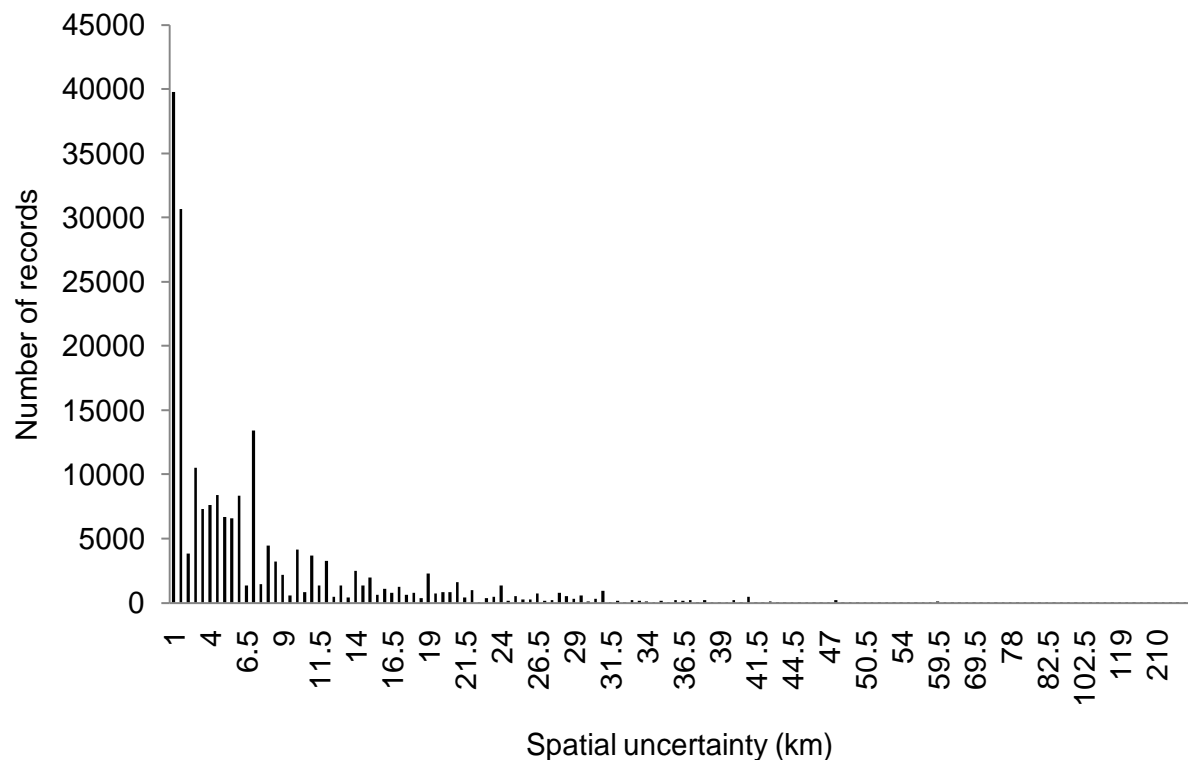


Figure 8. Spatial uncertainty of NMLS shore patrol data.

### Commercial linefish

Almost all (99.98%) of the 259019 commercial linefish records for SeaPLAN species in KZN had offshore distance associated with them, and were estimated to be spatially accurate to five km (Figure 7). The remaining 0.02% had no offshore distance and was ignored.

Commercial ski-boat data are all recorded to day level (including year and month) (Table 4).

### ORI/WWF-SA tagging data

The ORI tagging database held a total of 13901 records (including recaptures) for 41 of the 67 SeaPLAN species (Appendix 4), of which 68% of the records were usable (i.e. shore based records). Twenty-two percent of the ORI tagging data had a spatial resolution of one km, and a further 33% were accurate to four km. Only 12% had a spatial uncertainty greater than four km (Figure 9), excluding non-usable records. Therefore 44% of the SeaPLAN species data were regarded as not useful for finer-scale (1 km<sup>2</sup>) distribution modelling. No time data were recorded, but all records include year, month and day (Table 4).

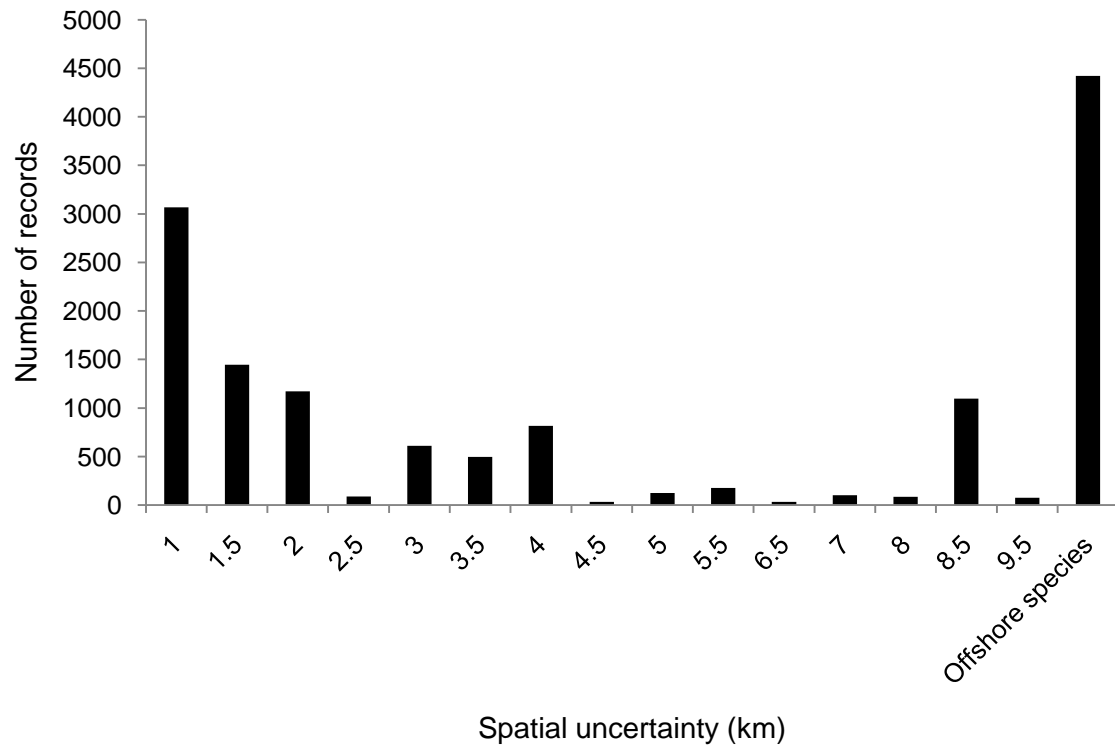


Figure 9. Spatial uncertainty of ORI/WWF-SA tagging data.

### KZN Sharks Board

The KZNSB held 7685 records for ten SeaPLAN species of which all data were accurate to  $\leq$ four km and 52% had a spatial resolution of one km (Figure 10). All data included year, month and day information (Table 4).

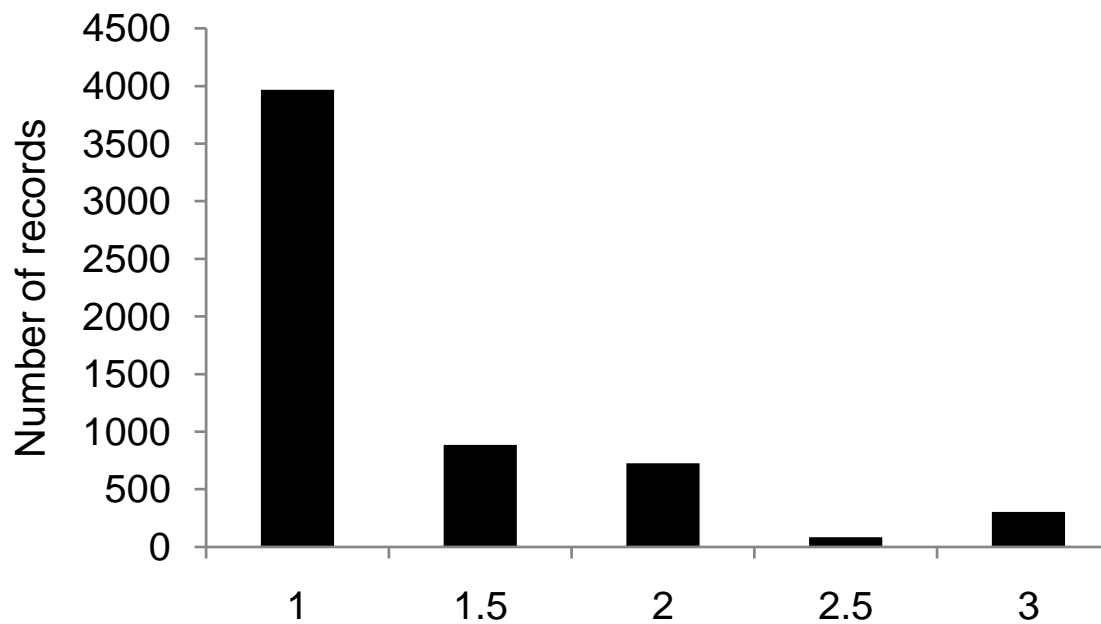


Figure 10. Spatial uncertainty of KZN Sharks Board data.

## Flights

There were 470 GPS positions reported for whale shark sightings made from aircraft, during flights conducted by ORI and KZNSB from February 2005 to February 2008. All data were estimated to be accurate to 100 m and were recorded to include year, month and day information (Table 4).

**Table 4. Temporal uncertainty associated with the data sets showing the number of records per time category.\***

<i>Data source</i>	<i>Time</i>	<i>Day</i>	<i>Month</i>	<i>Year</i>	<i>No date</i>
NMLS recreational shore angling	116877	4084	0	0	0
NMLS recreational ski-boat	21587	2885	0	0	0
NMLS shore patrol	200829	1399	0	0	0
NMLS commercial	0	258960	0	0	0
ORI tagging	0	13072	0	0	0
KZNSB	0	7685	0	0	0
Flight (ORI and KZNSB)	0	470	0	0	0

\*The record is added only to the most detailed level of time available for the record, and not repeated throughout all levels.



## Overall

Overall, 121558 records (18%) for SeaPLAN species had a spatial resolution of less than one km, while 109354 records had a spatial resolution of two to four km. A large proportion (345148 records, 52%) was spatially inaccurate (four to eight-and-a-half km) and 104845 records (15%) were not useable because the spatial uncertainty was greater than 8.5 km or could not be determined. The overall temporal accuracy of the data was high, with 40% of the data recorded to time level and 60% to day level. Less than one percent of the data records did not have a date.

## Discussion

I evaluated the spatial and temporal resolution of the data available for marine fish species in KZN that were identified for the conservation plan, SeaPLAN. A large number of records, 121558 (18%) in total, had a spatial resolution of less than one km, and would be suitable for modelling at this scale. An additional 109354 records satisfied the four km spatial resolution cut-off. A large proportion (52%) of the data was spatially inaccurate (four to eight-and-a-half km) and 15% was not useable because either the spatial uncertainty was larger than nine km or it could not be determined. The large number of records with great spatial inaccuracy is largely a result of outdated data collection methods used by the various programmes. All data with a resolution of one km or less came from shore-based sources, while all ski-boat, and hence offshore data, had a spatial resolution greater than one km. This compromises the ability to model the ranges of offshore species, because the data are not of a suitable resolution. The high level of temporal accuracy associated to the data (99.95 % to at least day level) allows for various temporal applications (such as seasonality) to be taken into consideration when modelling species ranges or identifying areas for conservation. Below I discuss the various data collection methods and make recommendations as to how the programmes can be improved.

## SAIAB

SAIAB holds data for a wide range of species, including data for species with little information available elsewhere, e.g. the green sawfish (*P. zjisron*). The few records held for more commonly caught species, e.g. the slinger (*C. puniceus*) are useful to supplement other smaller data sets (e.g. from Iziko) in order to verify distribution models based on the larger data sets (e.g. NMLS).

SAIAB data were a mixture of historical and new data, collected from 1901 to 2006, with varying spatial resolution. The SAIAB data set included a relatively high proportion (25%) of data with detailed locality descriptions (within one km). However, the remaining data (75%) analysed had a spatial uncertainty greater than one km, and were not suitable for fine-scale distribution modelling (see Figure 4). The data had a fairly high temporal accuracy, with 67% of the data including year, month and day (and time for some), which is remarkable when considering that it dates back to ca 1900. The data qualifying for distribution modelling were clearly more constrained by a lack of spatial resolution than that of temporal resolution. The lack of spatial resolution arose from historical data collection methods, where localities did not include coordinates, and were often no more specific than a town name. SAIAB has updated historical data collection methods to conform with international data standards, and all data since 2000 include GPS positions which are also recorded for the 082 TAG FISH Project (W. Coetzer, pers. comm.; Mann et al. 2010). The sample of data analysed (SeaPLAN species in KZN) did not match the species for which SAIAB collected GPS locality data post 2000, and therefore the sample of data held no GPS coordinates. Unfortunately, research and conservation priorities are not always aligned but stand to benefit from coordinating their efforts.

## Iziko

The two components (SAMS and SAMC) of the Iziko museum data set reflected high variability in spatial and temporal resolution. The locality information of the SAMC (bony fish) component was primarily based on locality names or descriptive accounts of localities, which included a few (46) records for SeaPLAN species in KZN. Fewer than 14% of these satisfied the one km resolution requirement for fine-scale distribution modelling. Data that are collected by the organisation itself are far easier to improve than data collected outside of the organisation (e.g. public). Museums have no control over the external data provided and have limited capability to refine this. It is, however, important to provide a reasonable estimate of the spatial uncertainty when such data are assigned coordinates to match the locality description. The use of already available technology (e.g. GPS devices on cell phones) should be encouraged to stimulate the general public to record accurate locality information. The fish data and the accuracy of the localities were said to be addressed in 2010 and may lead to an improvement of the data quality (W. Florence pers. comm.).

Iziko museum records typically provide the date that the collection was made, but the 100 records obtained from KZNSB gave the date that the data were obtained. The actual dates

for these collections can be obtained from KZNSB and if corrected will significantly reduce the temporal uncertainty from 72 % to 43 % for the Iziko data set.

### **The spatial uncertainty of locality information based on the beach locality code system**

The beach locality code system is used by the NMLS (all sectors), ORI tagging data, and KZNSB data but also several other programmes not included in this study (e.g. Boat launch site monitoring system, Fisheries independent data, and the Observer programme).

Localities recorded to the beach locality code system had higher spatial resolution than those only recorded to locality names. For example 'Durban' has a spatial resolution of one km (if regarded as 'Durban harbour' or nearly any other locality in Durban) in the beach locality code system, while 'Durban' has a spatial uncertainty of 15 km (if considered that the locality may imply anywhere along the shore of Durban) based on the locality descriptive method used in this study (e.g. for SAIAB data) (Figure 11). Data that are only recorded to "Durban" and then coded to a beach locality code can mask the locality's actual spatial uncertainty. The coding of the data is quality controlled by the data controllers at ORI based on years of experience, before being entered into the database (B.Q. Mann & J. Maggs pers. comm.).

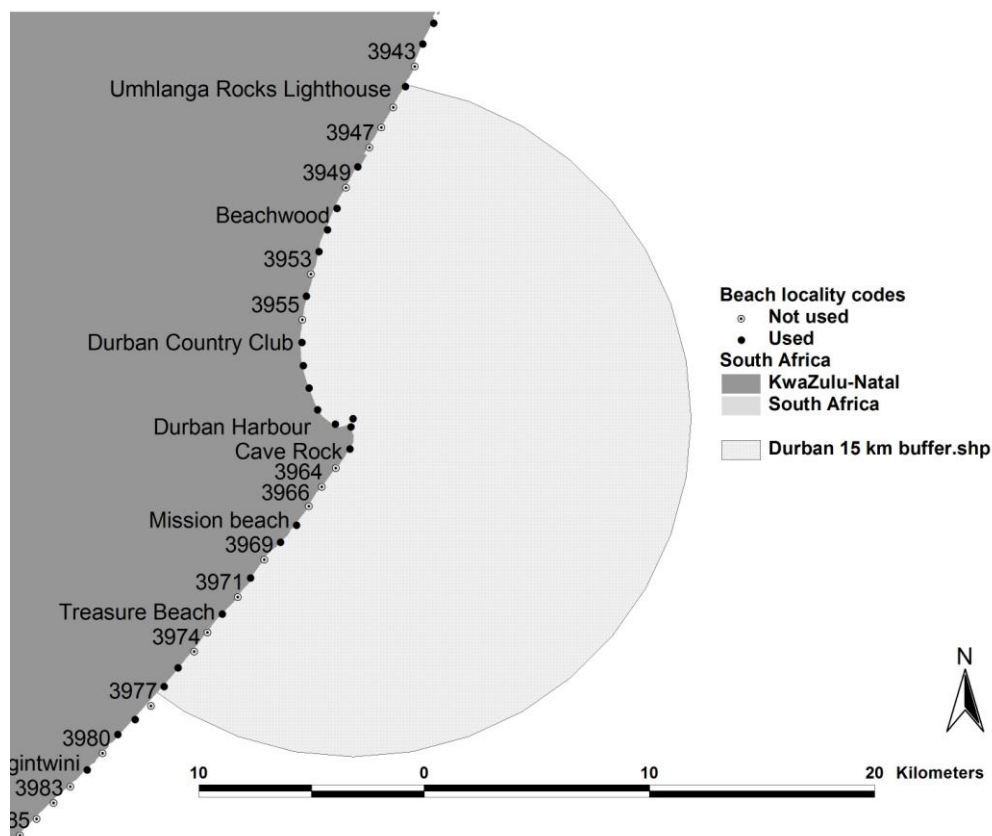


Figure 11. The difference in the spatial uncertainty of the two data collection systems. The data recorded to locality names (e.g. Durban) have a greater spatial uncertainty than data recorded to locality codes in the same areas.

Implicit in the kilometre interval based beach locality code system is the one km spatial uncertainty. Only some of the codes are in use (named or well-known beach localities), and therefore increase the spatial uncertainty from one locality to the next based on the distance between used locality codes. The different data collection programmes implement the system in different ways, affecting the spatial uncertainty of each of the data accordingly. I argue that the system is based on an outdated methodology and the technology which unnecessarily loses spatial resolution. Spatial information is being lost because there is no facility to record the more detailed spatial information (e.g. GPS coordinates). The database is housed at MCM, and will greatly benefit by including fields to record GPS positions when provided. EKZNW, responsible for collecting the majority of NMLS data, is currently developing a database system to accommodate such fields of information to avoid further loss of spatial resolution (C. Van Tichelen pers. comm.). I point out some of the shortcomings of methodology used within the various programmes, and recommend how it can be improved for the programmes using the beach locality code system.

## **National Marine Linefish System (NMLS)**

### **Recreational shore angling**

The NMLS recreational shore angling data held 74064 records for 36 of the 67 SeaPLAN species that satisfied the spatial scale requirements for modelling ranges at one km. The spatial uncertainty, which can be as much as 9.5 km, is a result of the large distance between used (Figure 3). Although the spatial resolution of the point data is represented by a one dimensional value, it has two dimensions, latitude and longitude, that may differ in their spatial uncertainty. In KZN, the change in latitude affects the long shore distance, and the change in longitude affects the distance offshore. The spatial uncertainty is greater in terms of the distance alongshore (latitude), than distance offshore (longitude), because it is restricted to ca 100 m by the distance that a fisherman can cast (B. Mann & J. Maggs pers. comm.).

The species for which there were no data were mostly non-linefish, e.g. goby species (Gobiidae spp) for which data are limited in general. The temporal resolution of the data were at a finer scale than required for purposes of modelling at a seasonal scale. Spatial uncertainty was large and limited the use of the data for the purpose of species distribution modelling in Chapter Three.

The data are known to contain accidental and deliberate misreporting on species, quantities and incorrect or vague locality information (Brouwer et al. 1997; Mann et al. 2010). These errors were not taken into account other than omitting data that were not recorded to species level. Error checking performed by ORI does remove a substantial number of the problems (B.Q. Mann & J. Maggs pers. comm.). For example, data that are recorded to obscure localities are followed up, and corrected where possible by ORI. Records that report fish species well outside of their known ranges are also followed up (B.Q. Mann & J. Maggs pers. comm.).

### **Recreational ski-boat**

The recreational ski-boat data included 35 of the 67 SeaPLAN species. These data had low spatial resolution (five km) at best. Most (66%) of the data were not usable because the spatial resolution could not be determined for data recorded to beach locality codes whilst distance offshore is unknown. The five km resolution estimated for the data were based on a consensus of expert opinions, but needs to be validated by empirical data. It was based on the same estimated spatial uncertainty associated with well-known commercial ski-boat fishing grounds, although commercial ski-boat data are reported differently. Recreational ski-boat fishermen report catches at launch sites, to EKZNW inspection staff. The catch data may include locality, of which some are well-known offshore fishing localities for which

locality codes and distances offshore have been established (B.Q. Mann & J. Maggs pers. comm.). Data that are reported to beach code localities without distances offshore have similar spatial uncertainty as shore angling for the latitude (along shore), but a very high degree of uncertainty in terms of longitude (distance offshore) (Figure 12). Data recorded to ski-boat launch sites have high offshore uncertainty and longshore (latitudinal) uncertainty, as neither the distance offshore nor the actual beach locality code is known (Figure 12). The spatial uncertainty for Durban (Figure 12) does not represent all ski-boat launch sites because these vary from one launch site to the next (B.Q. Mann & J. Maggs pers. comm.). These records were simply considered as not useable in this study. The distance that ski-boat fishermen are prepared to travel offshore varies with fishery and the species targeted, and varies for different parts of the coast, as well as from one beach locality code to the next. For example, ski-boat fishermen are known to travel up to 40 km to get to offshore reefs around Richards Bay, while ski-boats generally travel five km offshore on the south coast of KZN when fishing for demersal species. Pelagic species are less associated with spatially fixed habitats, and their spatial uncertainty may even be greater (B.Q. Mann & J. Maggs pers. comm.).

The large proportion (96%) of data that were recorded to time level allow for fine scale temporal applications of the data.

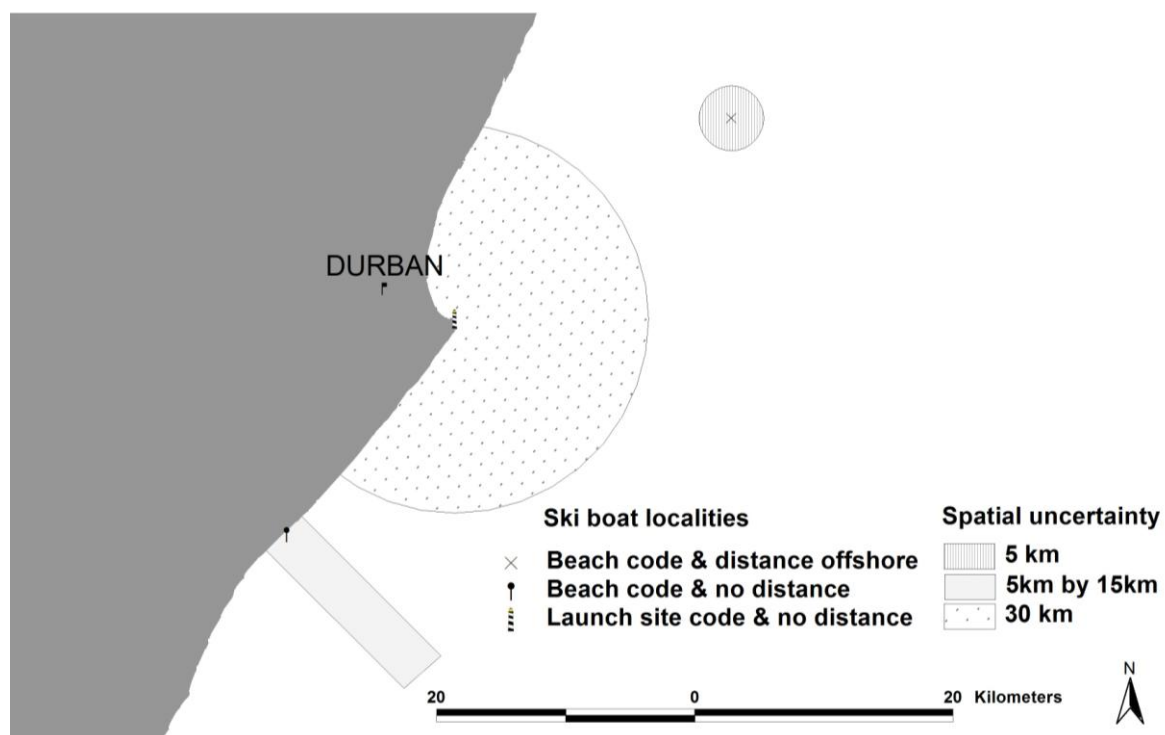


Figure 12. Three types of ski-boat beach locality code data showing the different spatial uncertainty associated with records with beach code and distance offshore; beach code and

*no distance offshore; and launch site code with no distance offshore. Note that spatial uncertainty for the latter two varies from site to site and was not included in earlier calculations.*

### **Shore patrol**

The shore patrol data included the largest number of records (202228), for 33 of the 67 SeaPLAN species. The data were dominated by records for a single species, shad (*P. saltatrix*), and contributed far fewer records for other species. Shore patrol beats used to cover larger distances prior to the beach vehicle ban in 2002, which may account for the 8407 records that are between 25 and 100 km in spatial resolution. Several records had a large spatial uncertainty, > 100 km, which may be a result of recording error. Time information may be less useful, or even misleading, because some hours can pass from the time a fish is caught until the time the data is recorded. A new system is currently being considered by EKZNW, in which shore patrol staff will use handheld biodiversity recorders (e.g. Trimble Juno ST with CyberTracker software) to record data, including GPS localities and time for each record, rather or in addition to data per patrol beat. This will greatly increase the efficiency with which data are recorded, as well as its spatio-temporal accuracy, and may improve species identification (if species photo IDs are added by the data recorder).

### **Commercial NMLS**

Almost all (99.98%) of the commercial ski-boat data included a distance offshore, and were assigned a spatial uncertainty of five km based on the estimated spatial uncertainty of well-known fishing localities. The geographical localities of the well-known fishing grounds have not been updated since they were first established when (MCM) fisheries managers attributed beach locality codes and offshore distances to them. Even if the well-known fishing localities have not changed spatially, they can certainly be delineated more accurately with the aid of GPS devices today.

Spatially explicit offshore fishing data would greatly benefit offshore conservation action. It is naive to believe that fishermen would reveal their fishing spots, especially in the commercial sector where people's livelihoods depend on guarding this information. It is however important for conservation authorities to obtain this information to conserve the resource (Sink et al. 2010). SANBI used coordinates from commercial crustacean trawlers in KZN to assess threats (amongst others) in an exercise to identify offshore marine protected areas (OMPA) (Sink et al. 2010). Fisheries legislation may have to be changed to require ski-boat fishermen to submit GPS track coordinates of fishing localities to gain better knowledge of species distributions and ensure responsible management.

## ORI/WWF-SA tagging

The lack of distinction between ski-boat and shore angling made it impossible to assess the spatial uncertainty of 32% of the data available for SeaPLAN species. As a result, only 22% of the records were suitable for modelling distributions at one km resolution. I assumed that shore anglers catch only nearshore species, and offshore species are caught only by ski-boat anglers, which may not always be true. For example, some species are caught by shore anglers and ski-boat fishermen, for example, yellowbelly rock-cod (*Epinephelus marginatus*). Although estuarine localities were not considered in this study, Kosi lakes included marine species like potato bass (*E. tukula*), and was therefore included in the analysis. The majority of the data (903 records) were, however, for riverbream (*Acanthopagrus vagus*), which is primarily an estuarine species, and is occasionally found in the nearshore environment (Heemstra & Heemstra 2004). The large number of records from Kosi lakes decreased the average spatial resolution associated to the data because it has a high spatial uncertainty (8.5 km). The temporal accuracy of the data were suitable for the seasonal distribution modelling of this study.

Current ORI tagging can be improved by differentiating ski-boat data from shore angling data, because the spatial resolution of these fishing methods differs. SAIAB recently started a fish tagging programme, 082 Fish Tag, that aims to record high resolution locality information, preferably using GPS devices. This will facilitate fine-scale movement studies, and other spatial applications (Mann et al. 2010). The spatial resolution of future data collection would be improved if the locality data included GPS positions, and an effort was made to collect locality information from participants to a higher spatial resolution. The rapid increase in technology (e.g. camera & GPS service on cell phones) is potentially a useful resource to record spatially explicit information by volunteer participants. The data can then be sent instantaneously to the data centre.

## KZN Sharks Board

Although the locality information was recorded using the beach locality code system, the highest spatial uncertainty was 3.5 km. The areas where shark nets are placed are typically popular bathing beaches, with high levels of access and thus small distances to 'used' locality codes, resulting in a few localities with poor spatial resolution being included. The same method was used to assess the spatial resolution as for the NMLS recreational shore angling data, because the net and drum positions are recorded only to the 'used' localities. Although 52% of the KZNSB data had a one km spatial resolution, it can be increased by georeferencing the net and drum line positions using a GPS. KZNSB has recently



georeferenced some of the shark nets and drum lines, but this process was incomplete, and the coordinates were yet to be verified at the time of this study (S. Dudley pers. comm.). The spatial resolution of the historical KZNSB data can be improved by georeferencing the net and drum positions because their positions are relatively stable, even over long periods of time, and their positions are well known (S. Dudley, G. Cliff & S. Wintner pers. comm.). Although shark nets and drum lines are only installed in the central and southern parts of KZN, the data are a valuable resource because they hold more information than other data sets for several shark species that are not typically caught by shore anglers or ski-boats (e.g. great white shark, *Carcharodon carcharias*).

Shark net data do not necessarily reveal whether the sharks caught were feeding or merely passing through. Specimens caught on drum-lines, would necessarily have to be feeding. No effort was made distinguish between the two data sets, but such a distinction could potentially increase the spatial resolution of the data, because drum line spatial accuracy should be higher than the position of shark nets (drum lines are much smaller). Shark net data are also spatially biased towards popular beaches, with limited options for improving this bias, other than supplementing the data with ORI tagging data.

### **Data not used**

Several data sources have not yet been evaluated, for example, fisheries-independent survey data, boat launch site monitoring system data (BLSMS), the launch site observer program, and other commercial fisheries, like trawling and long-lining.

Fisheries independent data surveys are conducted from time to time (ca 10 year intervals) to validate NMLS data and provide insight on deliberate and non-deliberate non-reporting (Brouwer et al. 1997; B.Q. Mann & J. Maggs pers. comm.). These data are highly accurate in terms of species identification, but suffer from the same spatial and temporal uncertainty as NMLS shore patrol data, because the same data recording methods are used as for the recreational NMLS.

The BLSMS records catches landed by all vessels at launch sites (Khumalo et al. 2009; Mann et al. 2009), but the data are not useful for fine-scale modelling because they have the same spatial uncertainty as the recreational ski-boat data recorded to launch sites. The spatial uncertainty of data recorded to launch sites cannot be estimated accurately at present, and is likely to be greater than the resolution at which regional conservation planning happens, especially in the nearshore environment where planning can take place at a fine-scale (e.g. SeaPLAN).

The Launch site observer programme reports on non-reporting from launch sites, but data are available only from Oct 2007 - Sep 2009, when the project was abandoned in KZN. The data reports on commercial and recreational ski-boat catches focussed on the Richards Bay and Scottsburg-lower south coast area, because only two observers were conducting the research (Mann et al. 2009). The data have the same spatial uncertainty as the NMLS recreational ski-boat data.

Other commercial fisheries that are operated in KZN include intensive commercial deep and shallow water crustacean trawling, in which bycatch fish species of commercial value are retained, and limited long-lining and beach seine-net fisheries (Fennessy & Groeneveld 1997). Commercial long-line fisheries include local and foreign vessels and focus on large pelagic species, like tuna and swordfish, as well as shark species. Several commercial fishery operations like trawling and long-lining suffer from great spatial uncertainty owing to the nature of the in-motion fishery operation. The data may, however, be useful for offshore spatial planning that can happen at broader scales (e.g. 10 km SeaPLAN).

### **Duplication of data among data sets**

Duplication can arise from a record being submitted to multiple data collectors. For example, specimens that were sent to SAIAB, and data also submitted to the NMLS. The removal of duplicate records prior to modelling species ranges has been recommended for both presence-absence and abundance techniques, and can be automatically removed by some software (e.g. Maxent) (Phillips et al. 2006). It is therefore important to recognise duplicates. Initially an attempt to identify duplicated records was made, but was abandoned owing to limited information on identification fields (e.g. dates and collectors), which limited the detection of unique, duplicated, and multiple duplicate records within the large volumes of data. It would be beneficial to assign unique identity numbers to records or specimens collected in a centralized database to avoid duplication.

### **OBIS**

OBIS supplies the user with locality records for several South African fish data collections, but falls short by providing misleading spatial resolution estimates and lacks sufficient background information on the data collection programme to provide the user with the necessary information on limitations of the data. The coordinates that were available from OBIS for SeaPLAN species could not be used for distribution modelling at one km resolution. For example, in an initial test 11 % of SAIAB records fell ca 3 km inland and one locality fell 38 km from water, while two Iziko data points fall at four and five km inland respectively. The

SAIAB data set contained one record that fell far offshore (> 20 km offshore) for a shore based species, *Rhinobatos annulatus* (lesser guitarfish). The spatial uncertainty estimate (100 m) that was assigned to the SAIAB and Iziko data were artefacts of uniformly allocating the uncertainty associated with coordinate data (100 m) and took no cognisance of spatial uncertainty prior to assigning coordinates. The value of providing coordinates and estimates of spatial resolution is lost for some data because their coordinates are incorrect, and the spatial uncertainty estimates are thus incorrect and misleading. I have shown the importance of assessing the data available from the online data repositories, as has been recommended by other studies (Robertson 2008; Robertson et al. 2010), and found that significant proportions of the data are not usable owing to a lack of spatial or temporal resolution. I recommend that users assess the spatial resolution of the data made available from online data repositories, like OBIS, before using the data for spatial applications.

In spite of the poor spatial resolution of data currently available via OBIS for marine fish species in KZN, the development of online data repositories offers an unparalleled potential for spatial applications (e.g. biogeographical studies, species distribution modelling, conservation planning) because it has several advantages over privately-housed data sets: improved communication among data holders; increased use of the data; ease of data access; standardisation of data fields collected; larger data sets can be filtered to include a sufficient number of records that meet the resolution requirements for spatial applications; and independent data sets for training and testing data are readily available. The OBIS online data repository includes geospatial software (e.g. Marine geospatial ecology tools), distribution modelling software (GARP, K-mapper), and effective geographical display options for Google Earth (e.g. KML converter) (see [http://iobis.org/news\\_items/](http://iobis.org/news_items/)). The developments of these features are enhanced by an international community participating in readily available data and problem sharing.

Every effort should be made to improve the data collection practices that result in poor spatial data. I also urge the data collectors to collect spatially explicit information and provide better estimates of spatial uncertainty. Readily available technology (e.g. handheld data recording devices, GPS devices and the software to increase the data transfer between device and database), increase quality and confidence in data by reducing human induced error and increase efficiency (e.g. species identification, spatial and temporal accuracy and ease of access and accuracy of data transfer).

In the light of rapid biodiversity loss (WRI 2005), alarming rates of collapsed- and declining fish stocks (The State of the World Fisheries and Aquaculture 2008), and irreversible climate change (IPCC, Fourth Assessment, 2007), it has never been more appropriate for data

collection to comply with international biological data standards. Standardised methods will increase the usefulness of data, and will increase conservation efficiency (TDWG 2009). I urge data collectors to adopt international data standards to avoid further unnecessary loss of spatial and other data quality properties. Adopting international data standards (TDWG 2009) would serve to add confidence to the data, and remove several hurdles to data retrieval, increase its use and interpretation, and would hardly compromise current applications.

Conservation planning, however, cannot wait for data collection programmes to comply with the requirements for accurate distribution modelling and alternative approaches to modelling have to be sought. These are explored in Chapter Three.

### **Recommendations to improve data collection programmes in KZN**

1. Data fields recorded should be expanded to comply with international standards that have been adopted by 56 of the largest online data repositories (see <http://www.tdwg.org/biodiv-networks/>), of which OBIS is one.
2. Coordinates should be recorded using GPS devices to increase spatial accuracy.
3. Skippers of ski-boats should be required to upload their fishing track coordinates when back at the launch.
4. Diver-based and offshore data collection programmes should be initiated to collect data for offshore and non-linefish species and to address the lack of data north of Cape Vidal in Marine Protected Areas. Many divers and dive operators are present on a regular basis, and some divers possess good species identification skills that can contribute useful data.
5. Research priorities need to address the lack of offshore data.
6. EKZNW reported that 50 training programmes were attended by shore patrol staff from 2001 to 2005 (<http://www.kznwildlife.com/index.php?/Coastal-Compliance.html>). These programmes should be expanded to include fish species identification courses and GPS and handheld data recorder operation skills.
7. Public awareness raising campaigns concerning species identifications, regulations and the importance of protecting biodiversity should be promoted. Awareness campaigns need to explain the value of public participation in biodiversity conservation efforts, for example, data collection programmes like the NMLS.
8. Better estimates of spatial uncertainty provided by OBIS would serve to identify data of suitable resolutions for modelling purposes, and would avoid users from going back to the source and calculating spatial uncertainty individually. The consistency that this

information would provide would help ensure comparable results and applications of the data.

9. In the past, data collection procedures recorded locality information to locality name or to the beach locality code system. South African data collection programmes need to include estimates of spatio-temporal uncertainty for historical data. It is hoped that the methods used in this study to determine spatio-temporal accuracy for marine fish data in KZN will serve as a model upon which similar analysis of historical data for the rest of South Africa can be based.
10. Resources are currently spent on data transfer from paper to digital format, data checking, coding, and storage in outdated database systems. Modern data collection technologies offer paperless, instant, high spatio-temporal resolution options to improve biodiversity monitoring. For example, handheld data recorders with GPS capability and free software like CyberTracker (<http://www.cybertracker.org>).

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## Chapter 3

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# Modelling seasonal species life cycle envelopes for marine fish in KwaZulu-Natal

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## Abstract

Marine fish species distribution models (SDMs) were developed during the SeaPLAN project to provide predicted distributions of species life cycle envelopes (SLICES). Cartographic habitat association and range models (CHARMs), and Maxent software, were used. CHARMs provide simple presence-absence models of a species range, based on a synopsis of the literature. Maxent, however, uses environmental and point locality data to model a probability of occurrence. Although probability of occurrence distribution maps are more informative to conservation planning and management than presence-absence maps, skewed sampling effort and uncertainty of nearshore remotely-sensed data required the careful evaluation of Maxent models. Consequently, I used a decision framework to identify the most appropriate model for any given species data set. The framework was based on statistical strength (area under curve > 0.75), and (visual) fit of Maxent models to CHARMs. Relative variable contribution to Maxent SDMs was evaluated to gain an understanding of the most explanatory variables. Bathymetry was the dominant (65.57% ± 20.11) driver, while all other variables contributed far less (3.46% ± 5.45 on average). SDMs were constructed

for three temporal partitions of the data: annual, summer and winter. Summer and winter biodiversity richness patterns conformed to knowledge about seasonal variations in distribution. Biodiversity richness was higher on the south and central KZN coast during winter, while it was higher on the central and northern KZN coast during summer.

In this Chapter, a concerted effort was made to develop and identify suitable SDMs for all species included in the study. Fish species are important features to consider in a marine conservation plan. The SDMs that were developed are the biodiversity features that were used in the conservation planning in the next chapter (Chapter Four). The differences in seasonal distribution of biodiversity richness were also used to assess the differences in seasonal conservation plans in Chapter Four.

## Introduction

Conservation planning requires knowledge on the distribution of biodiversity features to inform spatial allocation of conservation effort (Margules & Pressey 2000). Distribution models of biodiversity features are often used in the absence of fully known distributions (Guisan & Zimmermann 2000). In this chapter I construct species distribution models (SDMs) for the fish species identified for spatial conservation in KwaZulu-Natal's provincial marine conservation plan, SeaPLAN (SeaPLAN species hereafter). The SDMs were based on a conceptual model that aims to incorporate the spatial distributions throughout a species' life cycle, known as species life cycle envelopes (SLICES). The different areas occupied throughout a species' life cycle are often spatially incongruent. For example, the nursery grounds of shad (*Pomatomus saltatrix*) are in the southern Cape, whereas their spawning areas occur from the Eastern Cape to central KwaZulu-Natal (KZN). Different life cycle stages may have different levels of vulnerability (e.g. spawning areas may be highly vulnerable), and may thus require different levels of conservation action. Two techniques were used to model species distributions: cartographic habitat association and range models (CHARMs), and Maxent (maximum entropy) models.

CHARMs are simplistic presence-absence SDMs that are based on associating knowledge of a given species distribution to environmental variables. Species distribution information was based on a synopsis of the literature (van der Elst & Vermeulen 1986; Compagno et al. 1989; van der Elst 1989; van der Elst & Thorpe 1989; Mann 2000; Heemstra & Heemstra 2004), whereas the environmental variables used (bathymetry, slope, rock reefs, coral reefs, and canyons) were those prepared for SeaPLAN (Lagabriele et al. 2010). The environmental space occupied by a given species in each of the environmental variables

was converted to presence-absence (1/0) GIS layer. The respective presence-absence layers (for each environmental variable) were then multiplied to yield a single presence-absence geographic distribution range for each species in KZN. For example, the final presence absence CHARM for a species distributed from Durban to Margate in 100 m water depth over rocky reefs, would have been constructed by multiplying the GIS layers for the following three variables: (1) 'Durban to Margate' present (value = 1) and everywhere else absent (value = 0), (2) Bathymetry, 0 – 100 m present, everywhere else absent, and (3) rock reefs present, and everywhere else absent. After multiplying these layers, only rock reefs between Margate and Durban in 0 – 100 m of water will be marked as present, while everywhere else in the planning region will be absent. CHARMs often covered large portions of the planning area (the Exclusive Economic Zone (EEZ)), because many species' ranges span the length of the coastline and may even extend beyond its provincial boundaries. In order to refine the CHARMs, preferred ranges for each species were identified, based on literature or expert advice (see Literature Cited). Preferred ranges are those areas within a full distribution range where the probability of occurrence is higher. However, the need for more informative SDMs on preferred ranges warranted further investigation with the use of alternative modelling approaches.

Preferred ranges have previously been used to identify conservation priorities. As a local example, Turpie et al. (2000) identified 'core' areas for marine fish conservation on a national level. Core areas were defined as the species range in South Africa (SA) excluding 25 % of the length of coastline in the total distribution, or cut-off at the national boundary. This approach fails when a species has a bimodal distribution, for example, stonebreem (*Neoscorpis lithophilus*) prefer areas north and south of the Natal Bight (the wide continental shelf that extends from Durban to Cape Vidal) (Expert Workshop 2008). In addition, this approach does not include depth zone preferences per species.

Aquamaps, developed by Kaschner et al. (2006) are widely used by online data repositories, like OBIS, GBIF, and Fishbase (Froese & Pauly 2009; GBIF 2010; www.iobis.org 2010) to visualise global scale marine fish distributions. Aquamaps estimate relative environmental suitability (RES) based on the environmental tolerances of a given species with respect to depth, salinity, temperature, primary productivity, and the species' association with sea ice or coastal areas (Kaschner et al. 2006). Aquamaps predict RES by assuming that a species has an environmental tolerance that fits a trapezoidal shape, i.e. RES decreases with environmental distance from suitable range. The method used by Aquamaps is useful to predict species distributions at a global scale, but suffers shortcomings at finer scales for the following reasons: 1) knowledge of species' tolerances to most oceanographic variables

(e.g. sea surface temperature) is incomplete and involves coarse estimates at best (Agenbag et al. 2003), and 2) the resolution of some of the environmental parameters used (e.g. salinity) is too coarse to be used at the finer-scale required for SeaPLAN.

Several reasons prompted the investigation of alternative distribution modelling approaches: 1) the availability of long-term point locality data (Chapter Two); 2) the availability of remotely-sensed oceanographic data (see <http://oceancolor.gsfc.nasa.gov/>); and 3) the rapid development of distribution modelling software (e.g. Maxent). Models can offer far more informative results than point data alone, or broad ranges reported in field guides (Phillips et al. 2006; Rondinini et al. 2006).

The point data available included data sets that were analysed in Chapter Two: the National Marine Linefish System (NMLS); Oceanographic Research Institute (ORI) tagging data; KZN Sharks Board data; South African Institute for Aquatic Biodiversity (SAIAB) data; and Iziko Museum data. Environmental variables included bathymetry, slope and substratum (soft sediment, rock reef, coral reef) data used for SeaPLAN (Lagabrielle et al. 2010), as well as remotely-sensed data. Remotely-sensed data included oceanographic climate data for sea surface temperature (SST), chromophoric dissolved particulate organic matter (Cdom), chlorophyll a (Chl), and turbidity (Kd).

The importance of choosing the appropriate distribution modelling technique is discussed by (Guisan & Zimmermann 2000) and has been shown to lead to different outcomes using the same data (Hijmans & Graham 2006; Ward 2007; Hernandez et al. 2008). Elith et al. (2002; 2006; 2008), Valavanis et al. (2008) and Austin (2009) provide evaluations and comparisons of currently available distribution modelling techniques. Locally, a study by Agenbag et al. (2003) used general linear models (GLMs) and general additive models (GAMs) to predict the distribution of three pelagic species, sardine (*Sardinops sagax*), anchovy (*Engraulis capensis*) and round herring (*Etrumeus whiteheadi*). Although presence-only data have been used in GLMs and GAMs (Ferrier et al. 2002), these models are best used with presence-absence or abundance data (Fielding & Bell 1997; Elith et al. 2006; Hirzel & Le Lay 2008; Austin 2009). Such data are rarely available for marine fish species (Kaschner et al. 2006; Valavanis et al. 2008). Other techniques that can use presence-only data include DOMAIN (Carpenter et al. 1993), Mahalanobis typicality (Hernandez et al. 2008), BIOCLIM (Nix 1986; Meynecke 2004; Beaumont et al. 2005), BIOMAPPER (Hirzel et al. 2002), GARP (Stockwell & Noble 1992; Stockwell & Peters 1999), Maxent (Phillips et al. 2004) and Boosted regression trees (BRTs) (Elith et al. 2008). Maxent and BRTs consistently outperform other modelling methods (Elith et al. 2006; Phillips & Dudík 2008). Maxent was chosen for the

present study because of its simplicity, compared with the relatively complex nature of BRTs, and the availability of local training courses in Maxent at the time of the study.

Maxent is a machine learning distribution modelling method (Phillips et al. 2004) that uses efficient algorithms to build relationships with the data that converges at maximum entropy (i.e. the most spread out distribution under the constraints of the relationships built with the environmental variables) (Phillips et al. 2006; Phillips & Dudík 2008). Maxent models the ecological niche occupied by the species, and the logistic output provides a probability of occurrence (Phillips et al. 2009). Maxent requires presence-only data, it can use both categorical and continuous variables and makes no assumptions about the distribution (Phillips et al. 2006). Another benefit of Maxent is that the algorithms used are transparent, and its development is an ongoing topic of research (Phillips et al. 2006; Phillips & Dudík 2008; Phillips et al. 2009). Maxent provides powerful statistical confidence measures, e.g. area under curve (AUC), to evaluate the strength of the models, and model performance in relation to the variables used. The statistical measures are useful in a pragmatic model building process, because variables that decrease model strength can be removed to improve the model.

Maxent is also used to model how introduced or alien species might spread through suitable habitat (Ward 2007), or distributions might change relative to climate change (Hijmans & Graham 2006), as well as how seasonal distributions change (Suárez-Seoane et al. 2008). Seasonality in fish assemblages is a well known phenomenon worldwide. The annual sardine run in KZN is a well-known example of marine fish migrations. The run occurs when large shoals of sardines migrate from the cooler Cape waters into warmer KZN waters to spawn during the winter months (van der Elst 1988; Armstrong et al. 1991; Heemstra & Heemstra 2004). Predatory winter migrants like the geelbek (*Atractoscion aequidens*) and red steenbras (*Petrus rupestris*) also migrate into KZN waters in pursuit of feeding and spawning grounds (Heemstra & Heemstra 2004; Connell 2007). A second suite of migratory fish arrive in KZN from the northern tropical waters in summer, e.g. king mackerel (*Scomberomorus commerson*), which extend their feeding range throughout the province in late February to March (van der Elst 1988; B.Q. Mann pers. comm.). Several of the species included in the present study are migratory.

Conservation priorities might vary as a result of the difference in spatial areas occupied by seasonal assemblages. In this chapter, three temporal analyses were undertaken and SDMs were developed accordingly: annual, summer and winter. Annual analyses took no temporal changes into account. Summer analyses included all year residents and summer visitors, and winter analyses included all year residents and winter visitors. Seasonal cut-offs were

associated with marine fish migratory turnovers periods (van der Elst 1988). The distributions of species richness in the three temporal analyses were compared to identify any spatial differences. Based on the literature and expert advice, a higher winter richness was expected in southern KZN, whereas a higher summer richness was expected in northern KZN.

Kaschner (2006) argues that point locality data for marine fish and mammals are scarce, and often suffer from sampling distribution bias, especially at a global scale. Poor spatial resolution and the skewed sampling effort in the data sets available to this study (Brouwer et al. 1997; B. Mann pers. comm.), as well as a lack of data for non-harvested species, make the use of sophisticated modelling methods questionable (Moisen & Frescino 2002). The validity of nearshore values of remotely-sensed oceanographic data is also questionable (Thomas et al. 2002).

Given the shortcomings of the data available for this study, a decision framework was used to identify the most spatially detailed, and appropriate SDM for a given species. To do this, Maxent SDMs were examined for statistical strength, and their outputs were compared with the CHARM SDMs (which included descriptions of known ranges from the literature and experts).

To date, statistical measures of model performance have been developed (Fielding & Bell 1997). For example, confusion matrices are often used to measure the model's predictive strength by assessing the number of false positive and false negative predictions relative to the number of true positive and negative predictions (Fielding & Bell 1997). The problem with this method is that the threshold value is not always known for all variables. The relative operating characteristic (ROC) is a threshold-independent statistical measure of the model's strength, that provides a measure similar to the confusion matrix under all possible threshold values, with a single figure, known as the area under the ROC curve (AUC) (Fielding & Bell 1997; Phillips et al. 2006). This statistical measure enables a comparison of the outcome of different distribution modelling techniques that model locality points to a set of environmental conditions (Elith et al. 2006). An AUC value of 0.75 was used in this study as a benchmark to determine if a Maxent SDM was useful (after Elith et al. 2006). The CHARM, however, represents the described range limits of a species within KZN waters. Consequently, a visual comparison of the Maxent SDM with the CHARM was made to estimate the goodness of the fit. Maxent SDMs that satisfied both statistical and 'CHARM fit' conditions were preferred over CHARMs alone. Once a Maxent model was applied to a species, the variables that contributed the most towards the model were calculated in order to gain an understanding of relative variable importance.

Finally, the winter and summer species-richness distribution patterns were compared. The chapter concludes with recommendations regarding data collection and model use.

## Methods

### Data preparation

#### Scale

Scale has three components in this study: 1) the geographical extent of species' distribution ranges; 2) the spatial resolution of species distribution data; and 3) the temporal component of species' distributions. The use of the term 'fine-scale' refers to small area/lots of detail/high resolution, whereas 'broad- or coarse-scale' refers to large area/little detail/low resolution.

Both the CHARM and Maxent models were restricted to the extent of the KZN EEZ, (see Chapter One), and all data were rasterised to a spatial resolution of 1 km<sup>2</sup> (Appendix 5). The individual 1 km<sup>2</sup> pixels are hereafter referred to as cells.

Data were modelled to three temporal divisions: annual, summer and winter. Annual data included all the months of the year, summer was defined as the months from January to June, whereas winter was July to December. These divisions were based on expert knowledge of times of the year when seasonal fish assemblages change (van der Elst 1988; B.Q. Mann pers. comm.).

#### Environmental data

Environmental data used for the SDMs included permanent (or semi-permanent) physical features, and remotely-sensed ocean climate data. Geographic Information System (GIS) layers of permanent physical features were obtained from SeaPLAN and included bathymetry, slope, rock reefs, coral reefs, and canyons (Lagabrielle et al. 2010). Climate data included four data sets: sea surface temperature (SST, night 11 µm), chlorophyll a (Chl), chromophoric dissolved organic matter (Cdom) and turbidity (Kd). Monthly climatologies (the average of a given month for all years) were downloaded from NASA's Ocean Color Website (Feldman 2009). Three of the variables (Chl, Cdom, and Kd) were MODIS 4.6 km data for 2002-182 to 2009-212 (Year-Julian day), while SST were Advanced Very High Resolution Radiometer (AVHRR) data from Pathfinder Version 5, 4.6 km, night SST: 1985 – 2001. Pathfinder data were preferred to MODIS data owing to the improved accuracy and longer temporal coverage, but were available only for SST at the time of writing.

Minimum and maximum composite GIS layers were prepared from the averaged monthly climatologies to yield a minimum and maximum climate scenario for each of the respective variables, for the three temporal partitions, e.g. summer SST maximum and summer SST minimum. The minimum and maximum of each variable was preferred to average or median values, because minima and maxima better define species range limitations than averages or medians (as recommended for Maxent, Phillips et al. (2006)). Minima and maxima were determined from averaged monthly climatologies (as opposed to minimum or maximum monthly climatologies) as they better represent the prevailing environmental fluctuations, than absolute minimum or maximum cell values. The climatic oceanographic variables are continuous variables and were interpolated from 4.6 x 4.6 km to 1 km<sup>2</sup> using a distance weighted average value between points.

### **Point locality data**

Locality data sources included the NMLS, ORI, KZN Sharks Board, SAIAB, and Iziko Museum data. Data requiring coordinates were georeferenced (see Appendix 5). The data were assessed in Chapter Two for spatial and temporal resolution. Shore-based data included only data with a spatial resolution of 1 km<sup>2</sup> or finer, while offshore data were accepted at a resolution of 5 km<sup>2</sup>. No offshore data had a spatial resolution finer than 5 km<sup>2</sup>, but the data were vital to model offshore distributions. It was preferred to model at a finer resolution, and if needed, the resolution could be adjusted at a later point, without compromising the models. Point locality data were grouped into summer or winter seasons, depending on the date of the record. All records were added to annual data. All duplicate records were removed from the three temporal groups to yield three sets including only unique locality records of SeaPLAN species. A total of 15009 unique locality records for the 67 SeaPLAN species were available for species SDMs, but see Table 1 for seasonal differences between the numbers of records. A total of 40 of the 54 species for which point locality data have been recorded had more than 5 records, and were used in Maxent for SDMs.

Data for the green sawfish (*Pristis zijsron*) and freshwater sawfish (*Pristis microdon*) were pooled into a single *Pristis spp* model because both species had too few records for SDMs, but collectively satisfied the minimum requirements. A single genus model was justified as these species' ranges overlap and share similar habitat in South Africa (Heemstra & Heemstra 2004). Furthermore, species identification has often been confused between these species (S. Dudley pers. comm.). Thirteen species were removed because they had fewer than five records (Table 1).



**Table 1. Data set statistics showing the breakdown of data available for species distribution modelling in Maxent.\***

	Annual	Summer	Winter
Total number of species with sufficient records for SDM	40	38	37
Total number of records	9881	7522	7324
Minimum records per species	9	9	7
Maximum records per species	851	706	658
Mean records per species	247.03	197.95	192.94
Median records per species	177	153	126

\*The annual does not add up to the seasonal sum because records recorded at a particular locality can only be used once.

### Species distribution modelling

Species life cycles were divided into three phases and the objective was to model the areas that the species occupied during these phases (Figure 1).

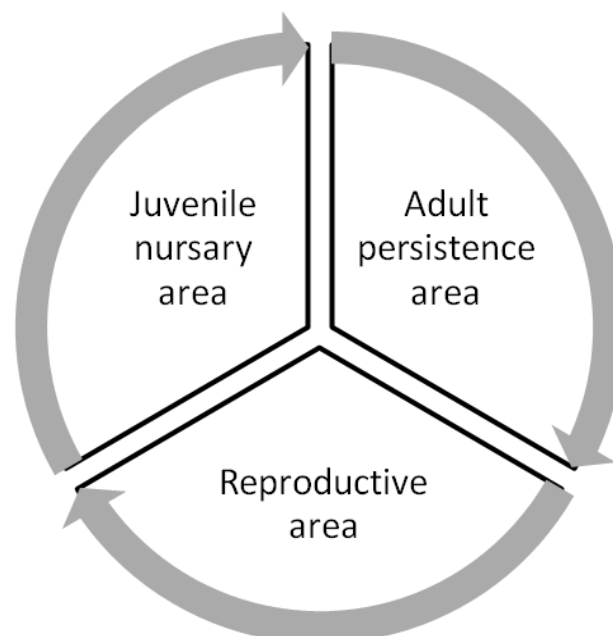


Figure 1. A conceptual model of species life cycle envelopes (SLICES).

The SLICE conceptual model is proposed here with the intention of future development as more data becomes available to better model reproductive and juvenile nursery areas, as well as the pathways that connect them. This first attempt was not equally useful for the three categories for several reasons: the point locality data could be used to model the adult persistence area only; lack of data limited the ability to model reproductive areas to the few species for which these areas are well known, and no juvenile nurseries were included. Other than the lack of data, juvenile nursery areas often fell outside of the planning area (either north or south, or in estuaries) or required environmental variables that were not available at the time of writing (e.g. tide pools). In this chapter therefore, only adult ranges and reproductive areas were modelled (with Maxent and CHARMS).

### **Maxent**

Maxent Version 3.3.3a (downloaded from <http://www.cs.princeton.edu/~schapire/Maxent/>) (Dudík et al. 2010) was used to construct SDMs for adult persistence areas (Figure 1). Temporal groups of point locality and environmental data sets were used together to construct annual and seasonal Maxent SDMs. The Maxent annual SDM was used to replace the species seasonal SDMs if there were insufficient seasonal point locality data.

The environmental data were prepared for Maxent as follows: a single substratum layer was prepared by amalgamating rock reef, coral reefs and the surrounding areas into a single categorical GIS layer. All Maxent SDMs started with all variables, including bathymetry, slope, substratum and ocean climate variables.

Maxent was 'run' three iterative times for each of the temporal data partitions (annual, summer, winter) for each species with at least 20 presence records. After each run, the resulting models were evaluated, and variables and parameters were adjusted accordingly to increase the statistical strength and better match known ranges.

Species that had more than five but fewer than 20 records were 'resampled' ten times by the cross validation option in Maxent to obtain statistically meaningful results. No distribution modelling was attempted for species with fewer than 5 locality records per data partition. Species data from different sources (Chapter Two) were pooled to obtain a more even spread of sampling effort, because the various data sources are different in their spatial biases. Background bias files were used to exclude certain areas from the background data in Maxent to compensate for sampling distribution bias in the data sets (further explained in Appendix 5). A random selection of 70 % of the point data was used to train the model, while the remaining 30 % was used to test the outcome.

Maxent has several parameters that influence the resulting model. All default parameter values were used, as recommended by (Phillips & Dudík 2008), except for the regularization parameter. This was adjusted as a last attempt to relax the entropy and increase an offshore prediction for species for which the models were under-predicting (as a result of offshore under sampling, see Appendix 5 for technical report).

The first Maxent run included all environmental variables and after each run these variables and other parameters were systematically adjusted to improve the model strength based on known distributions. The AUC using all variables was compared to the AUC value calculated by rerunning the same model but excluding the variables iteratively. Variables that decreased the model strength (AUC) were removed, if the following conditions were met (see Appendix 5 for details):

Variable contribution < 0.1, and

AUC without variable > Training AUC

Variables that decreased the model strength in more than three of the 10 outputs for species that were resampled were removed during the second run. Maxent is robust against model over-fitting (Phillips et al. 2006), and therefore a conservative approach to variable removal was preferred to minimize loss of predictive specificity (removing variables decreases model specificity).

Maxent SDMs were evaluated after the second run, and variables were removed using the same conditions as above. The map output was overlaid onto the CHARM map and locality points and the fit was visually evaluated, while considering literature descriptions of species range.

This study reports on the statistical strength and variable contribution of the Maxent SDMs, after the third run. The total contributions of variables were averaged over all SDMs and discussed in relation to the respective spatial scales at which the different variables affect species' distributions.

### **Cartographic habitat and range models (CHARMs)**

Cartographic habitat association range modelling (CHARM) was used to model the species life cycle envelopes (SLICES). I gathered information on the distribution ranges and habitat preferences for the SeaPLAN species from selected references (van der Elst & Vermeulen 1986; Compagno et al. 1989; van der Elst 1989; van der Elst & Thorpe 1989; Mann 2000; Heemstra & Heemstra 2004) and refined the information during consultation with experts

(Expert Workshop 2008). CHARMs were constructed by matching species range and habitat association information to the environmental GIS layers used in SeaPLAN, including bathymetry, rock reefs, coral reefs and canyons (Lagabrielle et al. 2010). Where information on preferred ranges was available, it was used rather than the full range. The preferred range was defined as the area described in the literature, or defined by experts.

All CHARMs were assigned to the annual data set and to a particular season if occupancy period was known. Permanent residents and species for which seasonality was unknown were assigned to both seasons.

### **Decision framework**

A decision framework was used to identify the SDMs with the most spatial information (Table 2). A species that did not have sufficient data for modelling in Maxent was automatically assigned to a CHARM. If a species had a preferred range CHARM it was automatically favoured over a full range CHARM, as both were constructed using the same process, but the former achieved the goal of refining the range better.

All Maxent models were evaluated against statistical performance measures: For a Maxent SDM to be considered for selection it had to achieve an AUC value of at least 0.75 (after Elith 2006). Maxent models that satisfied this condition were visually compared for fit against the CHARM, locality records, information from the literature and expert workshops. Two factors were considered during a visual comparison of fit: 1) range limits, and 2) general direction of increasing probability along-shore and offshore. CHARMs were favoured over Maxent models that did not satisfy statistical conditions, or fit (Table 2).

**Table 2. Decision support framework to determine modelling method for species range.**

		SDMs							
Sufficient point data?		Yes				No			
Maxent: AUC > 0.75		Yes		No		n/a	n/a		
Preferred range CHARM?		Yes		No		Yes	No	Yes	No
Visual comparison on Fit?	Good fit between Maxent & pref. range CHARM	Bad fit between Maxent & pref. range CHARM	Good fit between Maxent & full range CHARM	Bad fit between Maxent & full range CHARM	n/a	n/a	n/a	n/a	
MODEL	Maxent	Preferred range CHARM	Maxent	Full range CHARM	Preferred range CHARM	Full range CHARM	Preferred range CHARM	Full range CHARM	
Reason	Maxent fills in info missing from pref. range CHARM	Point data are more questionable than the pref. range CHARM	Maxent more informative than full range CHARM	Point data are more questionable than the pref. range CHARM	Point data are more questionable than the preferred range CHARM	Point data are more questionable than the full range CHARM	Pref. range is more informative than full CHARM	Only full CHARM available	

Preferred shortened (Pref).

### Richness distribution patterns

A species may have more than one SDM representing its different stages during a SLICE. For ease of explanation, the SLICES are referred to as separate species (nominal or pseudo-species), even if they refer to the same species. The SDMs therefore refer to all modelled SLICES (nominal SDMs). The three temporal sets of SDMs were summed and the richness distribution patterns described. The summed SDMs do not reflect the number of species but the total value of summed SDMs per cell (not all SDMs were presence-absence). The summed summer (summer richness) map was subtracted from the summed winter (winter richness) map to highlight the differences in seasonal richness distribution. The cell values of the differential seasonal map were calculated and plotted on a log scale to quantify the difference between the richness patterns.

## Results

### Maxent SDMs

After the point locality data were partitioned into temporal groups, a total of 115 Maxent SDMs were developed for adult persistence areas for 40 of the 66 species (a few species did not have sufficient records for seasonal SDMs, Table 1). In 97 of the 115 cases the statistical condition of  $AUC \Rightarrow 0.75$  was satisfied.

The SDMs for six nominal species under-predicted range limits for all three of their temporal partitions. Although increasing the regularization parameter of these models improved the fit to the known range, it decreased statistical performance significantly, ( $P(T \leq t)$  one-tail  $AUC < 0.05$ ;  $P(T \leq t)$  one-tail  $RTG < 0.0001$ ;  $df = 17$ ). CHARMS were used for five of the six species where the AUC value fell below 0.75. All five were shark species, for which data were collected predominantly from shark nets, and limited data exists (e.g. Bass 1968) for their offshore distributions (e.g. bull shark *Carcharhinus leucas*).

Thirty species satisfied the statistical and fit requirements (average  $AUC 0.918 (\pm 0.041)$ ). One species, the bigeye stumpnose (*Rhabdosargus thorpei*), did not have sufficient data for temporal partitions, and its annual Maxent SDM was used for both seasons.

### Environmental variables contribution to Maxent SDMs

Bathymetry contributed most ( $65.57\% \pm 20.11$ ) to the Maxent SDMs (i.e. it explained the species distributions the best). All other variables contributed far less ( $3.46\% \pm 5.45$  on average, see Figure 2). The minimum values of the oceanographic variables consistently contributed more to the SDMs than the maximum values. Substrata contributed the least to the models (Figure 2).

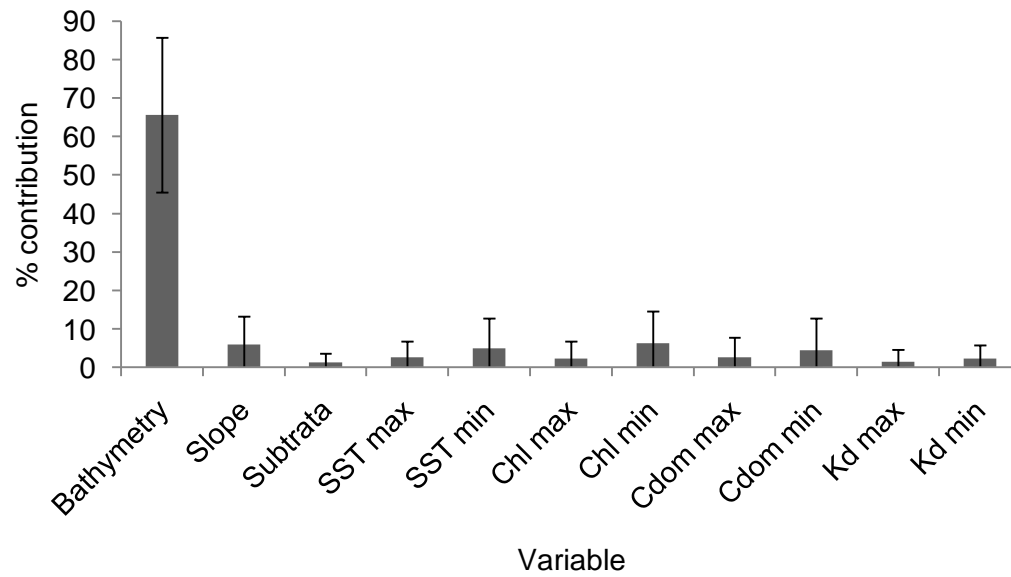


Figure 2. Average ( $\pm$  stdev) contribution of variables to Maxent SDMs ( $n = 90$ , unique final number of Maxent models). Bathymetry was the dominant contributing variable. See Appendix 5 for calculation on variable contribution.

### CHARMs

Seven adult persistence areas were modelled using preferred ranges, and 28 using full ranges. All CHARMs of adult persistence areas were used for annual and both seasons as residency for the species were not known. Reproductive areas were modelled for 12 species using preferred range CHARMs. Eleven of these were attributed to winter months, and six to summer months. No juvenile nursery areas were modelled.

### Overall SDMs

After the three temporal data sets were compiled, the annual data set contained 77 SDMs, 71 for summer and 76 for winter (Table 3).

**Table 3. Number of SDMs per model per season.**

SLICES	Model type	Annual	Summer	Winter
Adult persistence areas	Maxent	30	30	30
Adult persistence areas	Full range CHARM	28	28	28
Adult persistence areas	Preferred range CHARM	7	7	7
Reproductive areas	Preferred range CHARM	12	6	11
Juvenile nursery areas	NA	0	0	0
Total number of SDMs		77	71	76

### Richness distribution patterns

Inshore cells had higher values than offshore cells in the summed set of SDMs in all three temporal data sets (Figure 3 and Appendix 6). The southern and central regions of KZN had more high value cells than northern KZN (Figure 3). The highest values (> 40) were at the following localities: South of Scottburgh, Margate, between Margate and Port Edward, and Durban Bluff. Aliwal Shoal (Umkomaas - Scottburgh), Durban bay and its associated offshore reefs covered large areas with high cell values. Other areas with high values included the Tugela banks (offshore from Tugela River mouth) and the offshore reefs, south of Richards Bay and nearshore north of Richards Bay. The highest values in northern KZN were at St. Lucia, north of Cape Vidal, Sodwana Bay, and Kosi Bay (Figure 3).



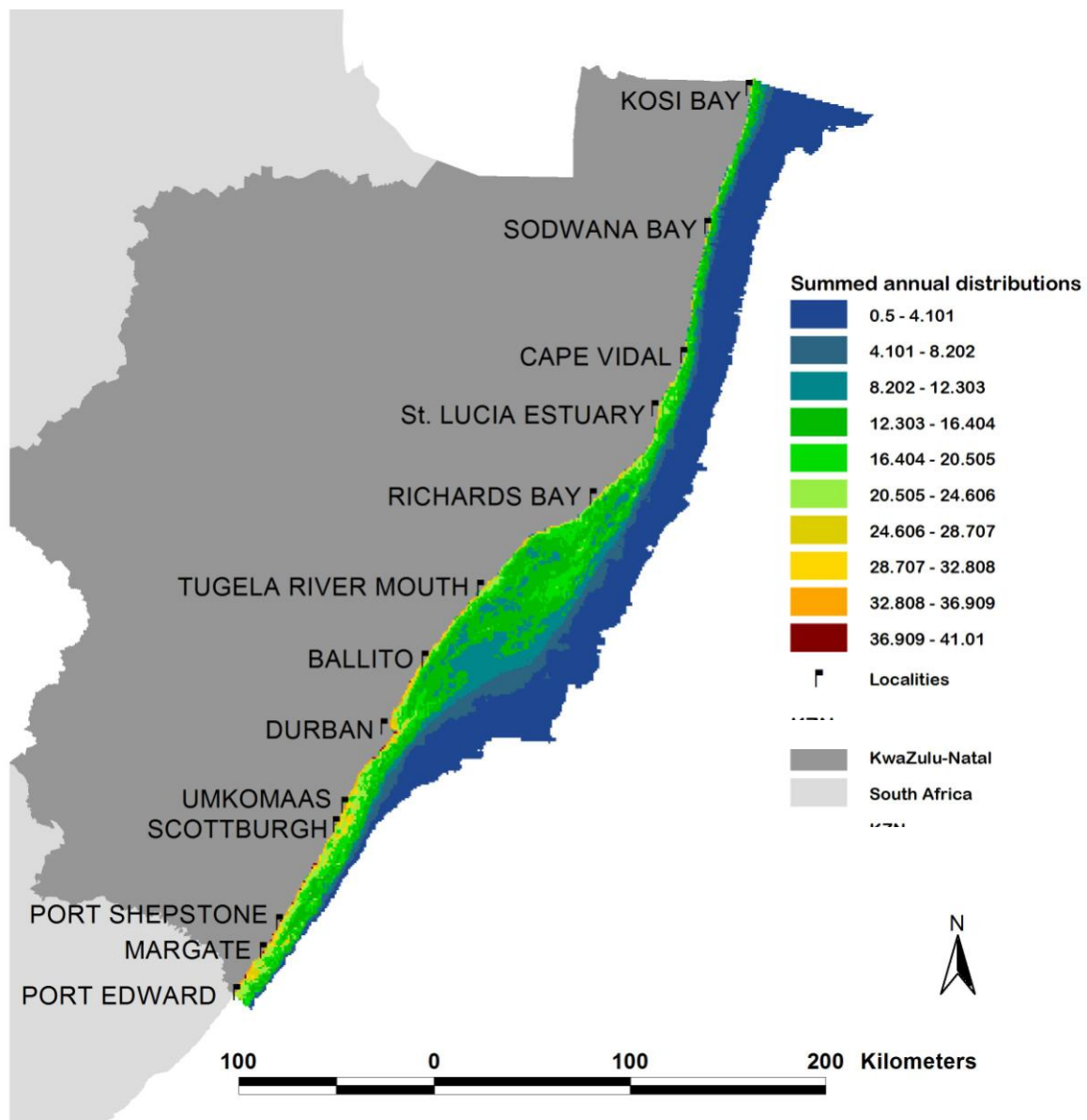


Figure 3. Summed annual distribution. Cell values were consistently higher inshore, and the south coast had more high value cells than the north coast of KZN.

The difference in species richness between summed summer and summed winter SDMs was calculated, and is shown in Figure 4 (see Appendix 6 for individual maps). The summer set consistently had more high value cells north of Cape Vidal than the winter set (indicated by orange in Figure 4). A surprising clump of high summer values intruded between high winter values just north of Umkomaas. The winter set consistently had more high value cells in southern KZN (indicated by the blue values in Figure 4), especially over offshore reefs (30 – 70 m) around Port Shepstone, than the summer set. Winter also had more high values in the nearshore area (particularly around Ballito) and again further offshore reefs in the Natal bight (Durban to Richards Bay) (Figure 4).

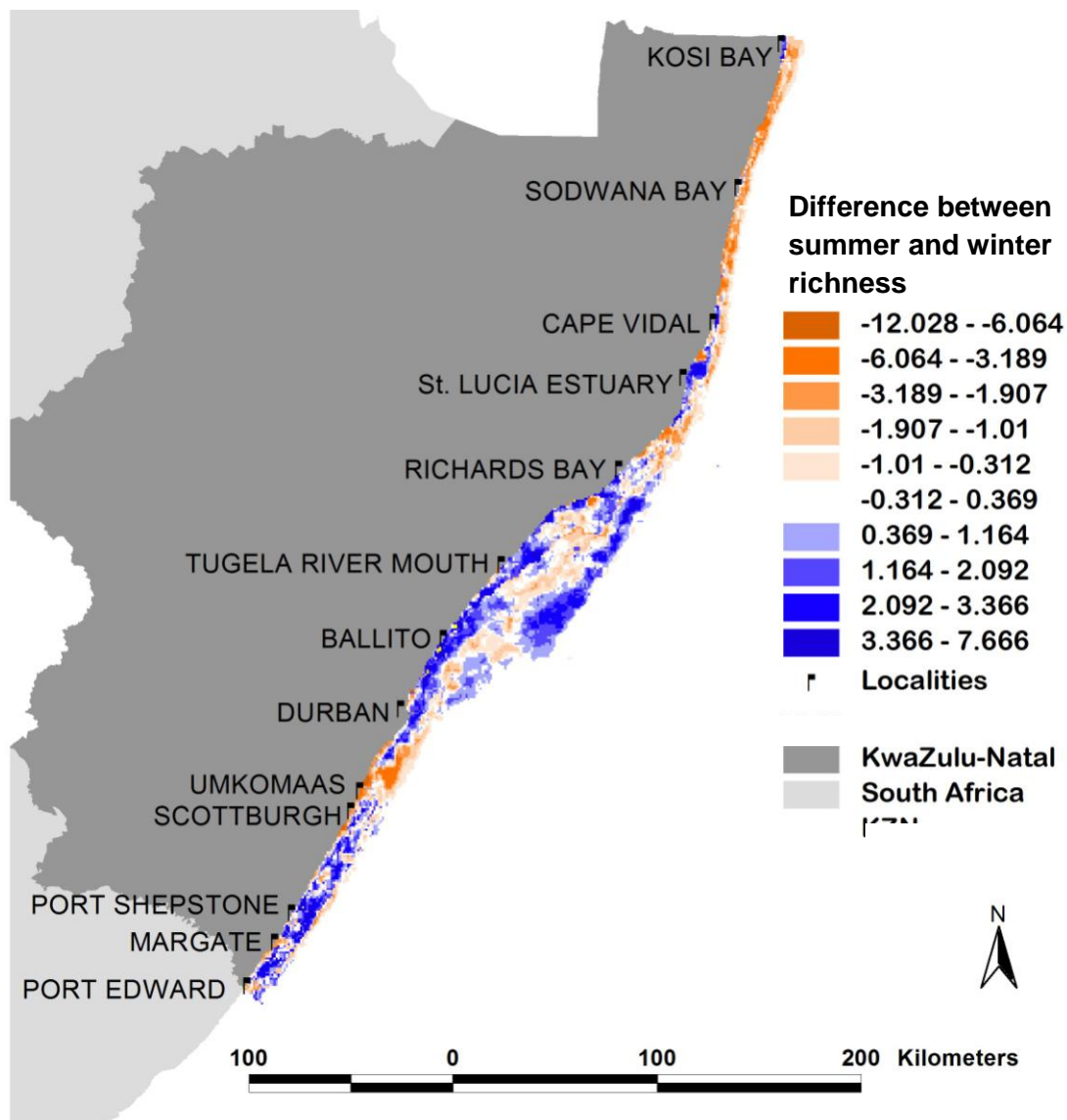
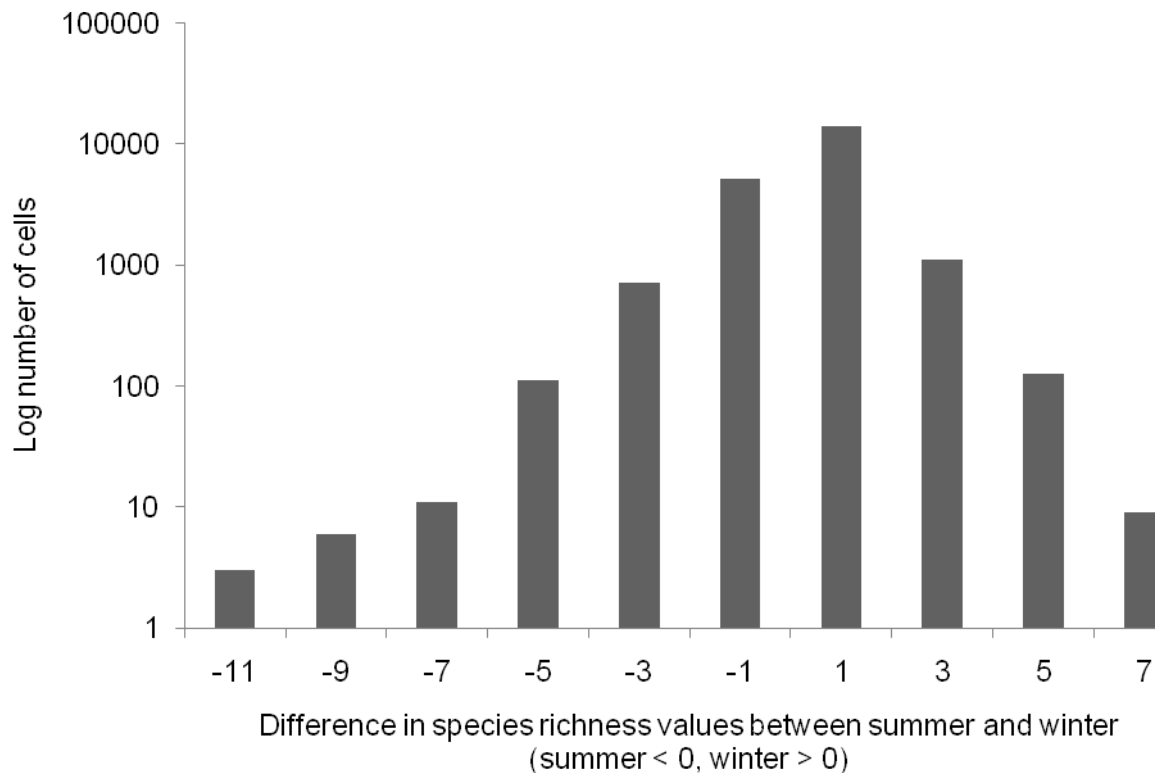


Figure 4. Differences between modelled species richness distributions in summer and winter: orange colours are areas where species richness is higher in summer, and blue colours are areas where species richness is higher in winter.

Although the seasonal model results conformed with knowledge of seasonal fish assemblages, the differences between winter and summer species richness values per cell were small (most differed by only one, Figure 5). This indicates that the SDMs cannot fully account for current knowledge of seasonal differences in fish assemblages.



*Figure 5. Number of cells with different species richness values in summed summer and winter. Negative values are cells with higher summer values, while positive values indicate higher winter values.*

## Discussion

Marine fish species are particularly data deficient, and enterprising approaches are required to provide meaningful information on their spatial distributions (Kaschner et al. 2006). In this chapter I used a combination of simplistic and more complex modelling techniques to map species distribution ranges.

Although the SLICE method of modelling life cycle distributions lacked data on reproductive and juvenile areas, it sets a platform for a holistic approach to spatial planning. SLICES incorporate all components of a species' life cycles, and the pathways that connect these. This provides an important aspect of systematic conservation planning, by including biodiversity features and the ecological processes required to maintain them (Possingham et al. 2005). During SeaPLAN we attempted to model the pathways that connect the various SLICES, but insufficient evidence was available on transport mechanisms. There is currently a need for data on reproductive areas and juvenile nurseries, and a better understanding of the transport mechanisms that drive the movement between SLICES. These processes

often occur at scales broader than the provincial level. Consequently, it may be worthwhile to further develop SLICES and the pathways that connect these at national or oceanic scales.

In the decision framework, Maxent models were preferred over CHARMS because they are based on robust statistical modelling methods that utilise locality data and are therefore less subjective than preferred range SDMs. Maxent SDMs were also preferred because a probability surface provides more meaningful information for spatial decision making than the presence-absence surfaces produced by CHARMS. The rate of errors of omission (predicting absence where a species is present) and commission (predicting presence, where the species is absent) is dependent on the model structure (Rondinini et al 2006). CHARMS are more likely to include errors of commission, because they assign equal probability of occurrence over the entire geographic range (Rondinini et al 2006). Maxent makes explicit inferences based on point locality data, and the underlying environmental space to predict a probability of occurrence (Phillips et al 2006). The AUC statistic calculated for each Maxent model enables interpretation of distribution model performance, and the graphic output allows for visual comparison (fit) between the two modelling techniques. Below is an example of a Maxent model that satisfied both the statistical and fit requirements.

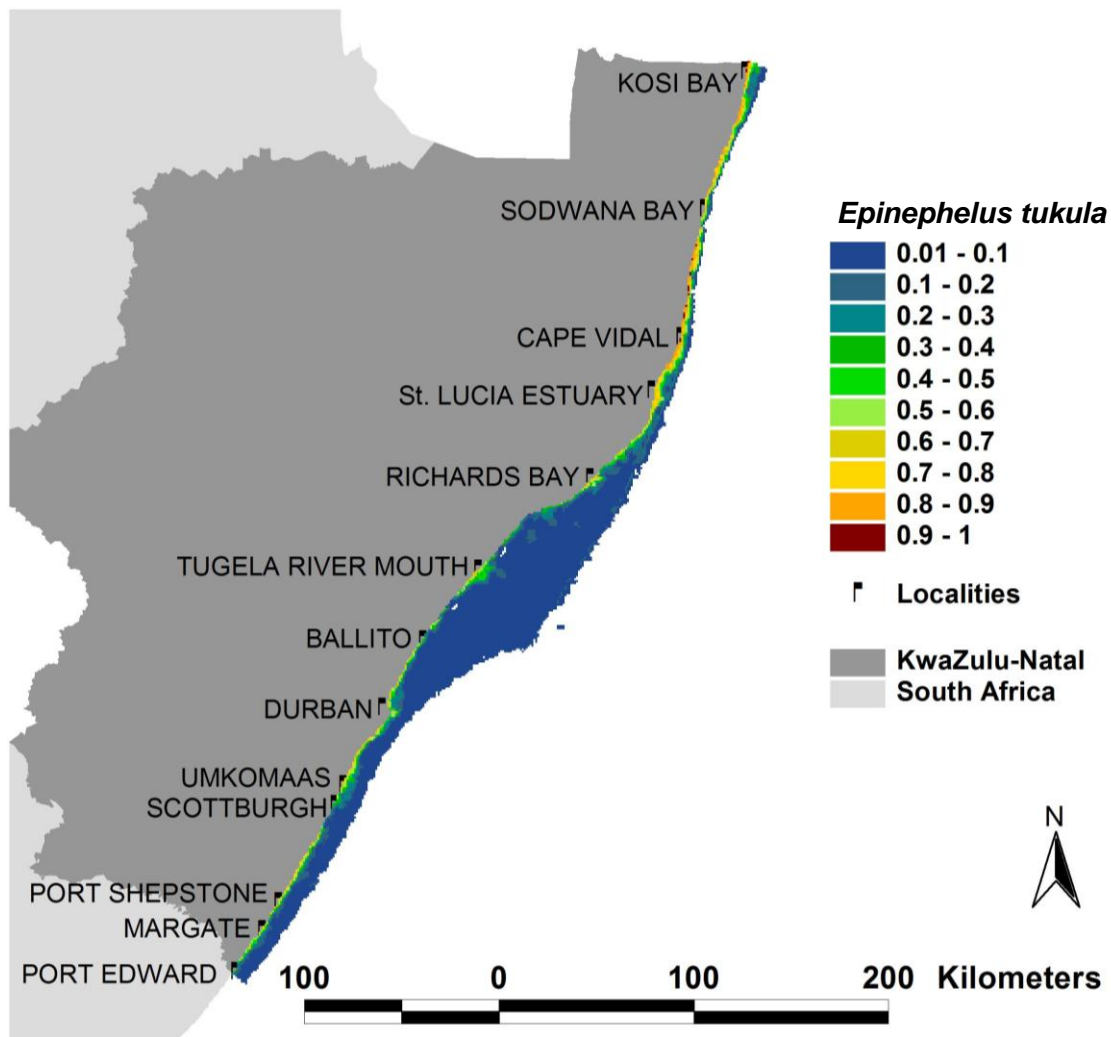


Figure 6. Maxent probability of occurrence for potato bass (*Epinephelus tukula*) in KZN (AUC 0.9). This distribution matches the range description in the literature for KZN: Kosi Bay to Port Edward, in 10 – 230 m of water, with a higher probability of occurrence in northern KZN and shallower than 100m.

The Maxent SDMs which did not satisfy the statistical or fit requirements were replaced by preferred range SDMs or by full range SDMs, based on the decision criteria. Below is an example of a poor Maxent SDM (Figure 7) that was replaced by a full range CHARM SDM (Figure 8).

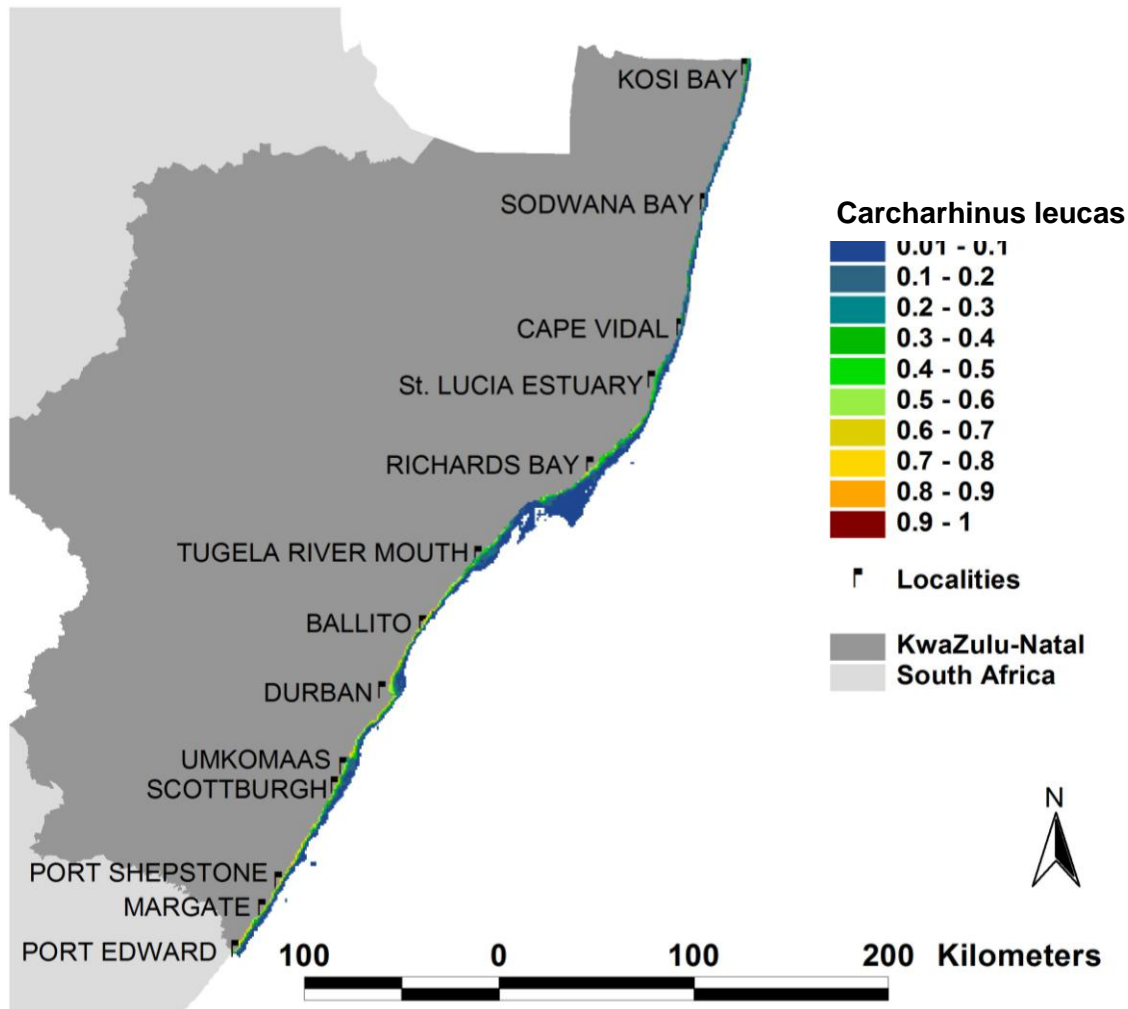


Figure 7. Maxent probability of occurrence for bull shark (*Carcharhinus leucas*) in KZN ( $AUC = 0.77$ ). This distribution is a poor fit to known range, because bull sharks are more prevalent in northern KZN than southern KZN (Dudley & Simpfendorfer 2006).

The SDM in Figure 7 reflects the sampled distribution rather than the probability of occurrence of the species, because the locality data were primarily from shark nets, which extend only as far as north as Richards Bay (Dudley & Simpfendorfer 2006). The area north of Richards Bay in turn coincides with a drastic change in oceanographic variables (e.g. SST) (see Sink et al. 2010), and as a result of the associations made between variable values and point localities in Maxent, the model under predicts in the north of KZN. The CHARM (Figure 8) was used instead.

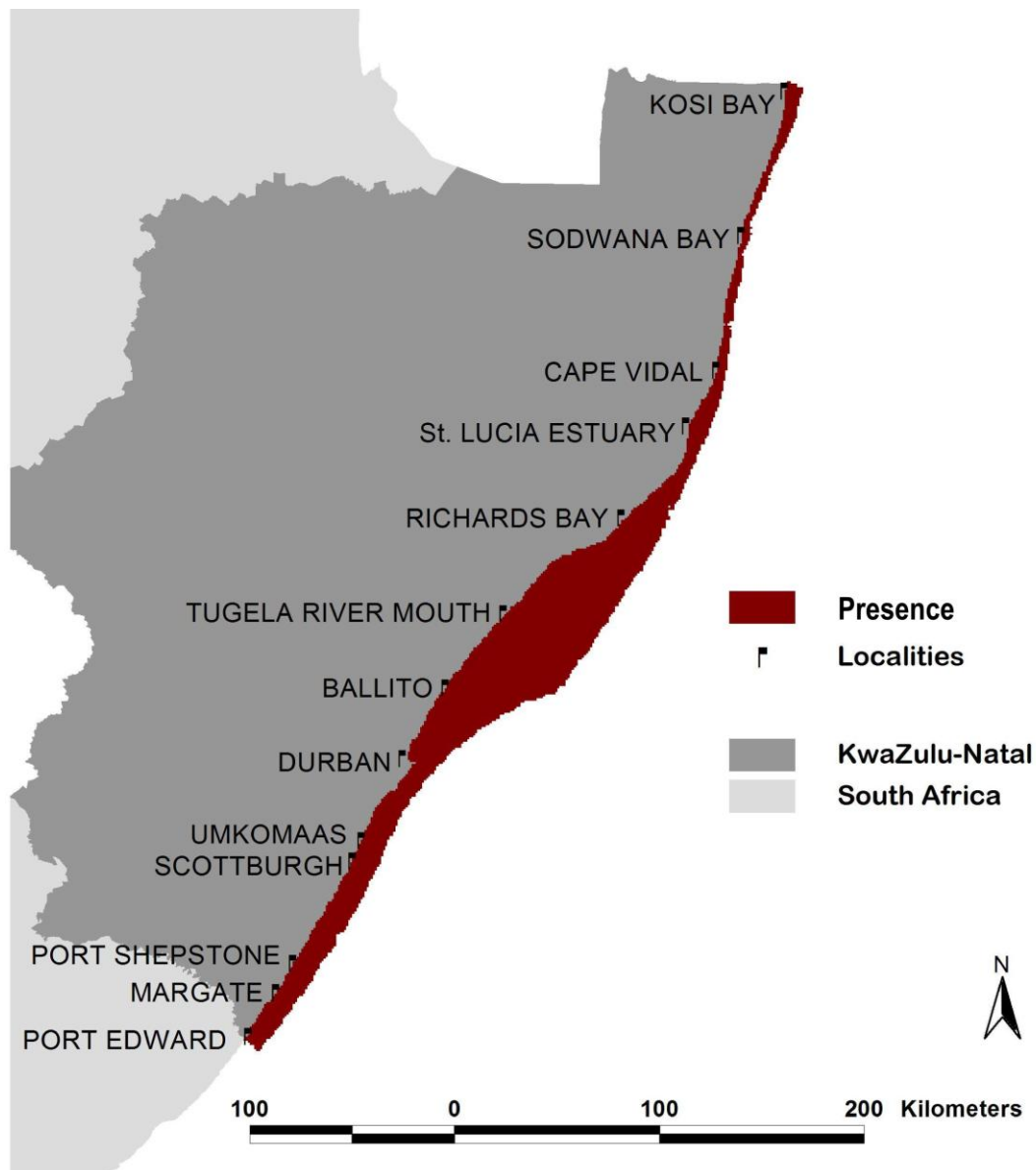


Figure 8. CHARM full range presence-absence distribution for the bull shark (*Carcharhinus leucas*) in KZN.

Although presence-absence over such a large area may not be very informative for identifying critical areas for conservation (Rondinini et al. 2006), it was preferred over the more specific but incorrect Maxent model in this case. Several species of shark suffered from under sampling in areas overlapping with a change in environmental conditions, e.g. great hammerhead shark (*Sphyrna mokarran*), which is also known as a more tropical species, could not be modelled using locality data. Non-sampled areas were excluded from training the Maxent models, in order to compensate for skewed sampling effort along the shore (for relevant species). This technique could not correct for non-sampled areas coinciding with a change in environmental variable values. Clearly a more representative data set is required for under-sampled species, especially shark species, and in particular

from the following areas: North of the Tugela river mouth and areas north of Cape Vidal, in particular the MPAs. This is in accord with the recommendation made at the end of Chapter Two that data should be collected from northern KZN, in particular from MPAs. A more complete data set will allow for more accurate and informative SDMs, and hence better information for spatial conservation management decisions.

Kaschner (2006) recommended that simplistic SDMs (such as used by Aquamaps) are more useful in instances when the data are inherently poor. This argument was substantiated by a comparison of modelling methods on forest patches in North America by Moisen & Frescino (2002) who found little added value in the use of sophisticated modelling techniques in instances of poor input data.

Although the simplistic CHARM modelling approach was not as informative as the probability of occurrence maps produced by Maxent, it could be used for all species and was readily used to model reproductive areas without locality data. For example, geelbek (*Atractoscion aequidens*) is a winter migrant that spawns in 40 – 60m of water as far north as Cape Vidal (Figure 9).



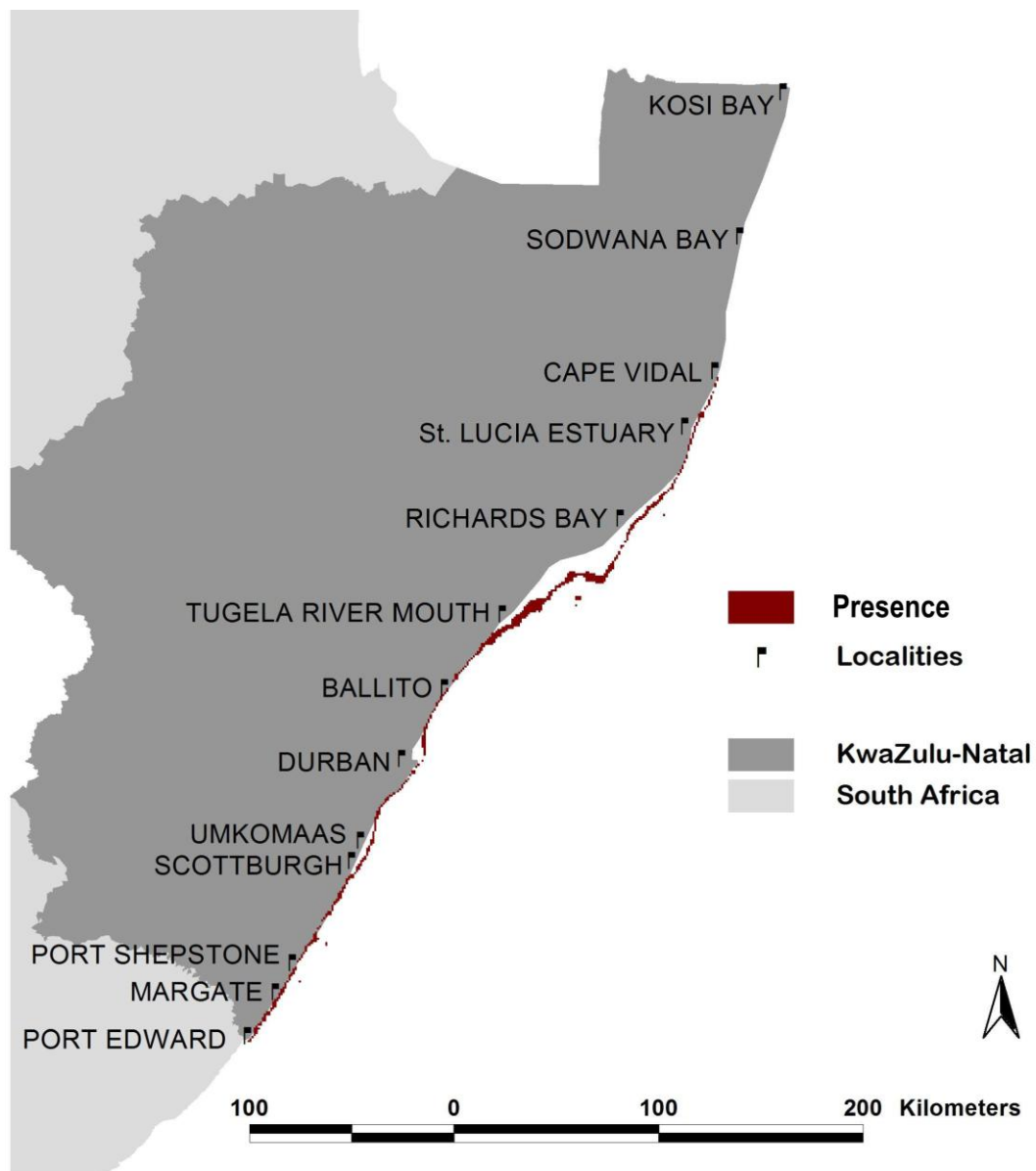


Figure 9. CHARM presence-absence distribution of the reproductive area (spawning area) of geelbek (*Atractoscion aequidens*) in KZN.

Preferred ranges were a useful refinement of full range SDMs because they remove less desired areas as options from a spatial conservation plan. Figure 10 shows the preferred range of the squaretail kob (*Argyrosomus thorpei*) (15 – 50m in the Natal bight, Durban to Richards Bay). The preferred range is far smaller than the full range (0 – 80m throughout KZN) and will provide more meaningful information to spatial management.

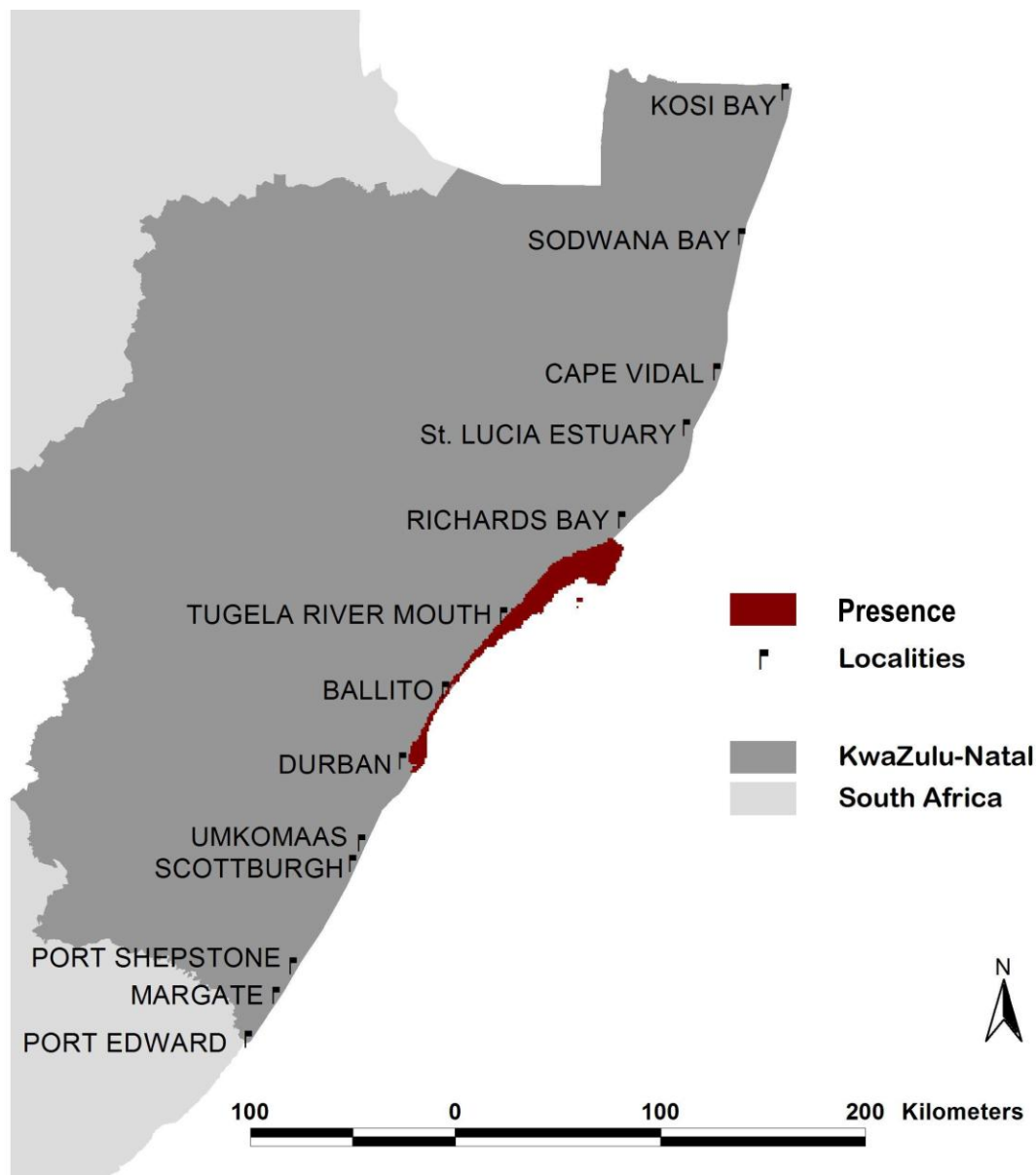


Figure 10. CHARM preferred range presence-absence distribution of the squaretail kob (*Argyrosomus thorpei*) in KZN.

### Environmental variable contribution to Maxent SDMs

Factors that lead to prediction errors in SDMs can be defined as algorithmic or biotic (Field et al. 2008). Algorithmic errors are the result of shortcomings of the algorithms or the data used, whereas biotic errors arise from the use of inappropriate variables (i.e. variables that do not adequately describe species ecology, Fielding & Bell 1997; Rondinini et al, 2006). A difficulty faced when constructing SDMs is the availability of environmental data, which is particularly problematic in South Africa (Moloney & Shillington 2007). Although it was generally assumed that all environmental variables available affect species distributions to some degree, the relationships are not well understood.

The high contribution of bathymetry to Maxent SDMs (Figure 2) is partly due to the shore-based locality data, and bathymetry data, being auto correlated to distance from shore. This auto correlation does, however, affect all of the variables to some extent. Bathymetry is a topographic-scale variable that affects species distributions on a local scale, and is a very appropriate parameter to use at provincial scale modelling (Phillips et al. 2006). The bathymetric range varies dramatically over the planning region (0 - 3600 m), and with most species being confined to areas shallower than 50 m, it is hardly surprising that it is the most important variable.

The low contribution of oceanographic variables (SST, Chl, Cdom and Kd) to Maxent SDMs can partly be explained by the differential scale at which the variables act, and the relatively fine scale (1 km) at which they were used in SDMs. The oceanographic variable data operate at a meso to global scale (Phillips & Dudík 2008), and limit species' distributions over provincial scales and broader (e.g. Wiley et al. 2003), but still have an effect at local scales if there is sufficient variability within the area of interest (Hassan 2004). The lack of more marine environmental data has been highlighted by Skov et al. (2008) and Moloney & Shillington (2007). Although several of the reefs have been included in the reef map prepared for SeaPLAN, the map is still rudimentary, and provides no insight into habitat complexity, reef composition, or reef size, and also lacks some infra-tidal reefs. This complexity is important for nearshore species (Crowder & Cooper 1982; Gratwicke & Speight 2005).

Fine scale variables used in other studies that were not available to this study include structural complexity of the substrata and specific habitats like seagrass beds. Lunar phase and tidal effects have been used to study the effects of marine fish distributions (Abou-Seedo et al. 1990; Agenbag et al. 2003), but operate at a finer temporal-spatial scale than considered here. Other commonly used variables that were not used in this study include salinity (Kaschner et al. 2006; Lenoir et al. 2010), and bottom temperature (Murawski 1993). Both were excluded because they were available only at the coarse scale of 1 degree. Primary productivity has been identified as a key environmental requirement to sustain global fisheries (Nixon 1982; Pauly & Christensen 1995). I used remotely-sensed chlorophyll as a surrogate for primary productivity. Although chlorophyll is a variable in the calculation of primary productivity, the values differ because primary productivity integrates the effects of sea surface temperature, euphotic zone depth, and day length in an already established equation (see <http://www.science.oregonstate.edu/ocean.productivity/vgpm.model.php>). In hind sight, primary productivity would have been a better environmental variable to use than chlorophyll (Chl). The loss to the SDMs as a result of using chlorophyll instead may have

been negated because some of the factors that are used to calculate Net Primary Production (NPP) were included as independent environmental variables in this study, e.g. SST. Day length, another factor used to calculate NPP, varies very little over KZN.

### **Spatial confidence of offshore point locality data in Maxent SDMs**

The poorer spatial resolution of the offshore point locality data (5 km<sup>2</sup>) did not affect Maxent models greatly because the environmental variables, especially in oceanographic parameters, are less variable in the offshore environment than the nearshore. Also, the Maxent models that used poor resolution offshore point data were generally well supplemented with the high resolution inshore point data, and this helped improve model outputs.

### **Species richness distribution patterns**

The higher values along the south and central KZN coast compared with the north coast reflect the richness distributions of the fish species that were selected for this study, and not the overall richness or distribution of marine biodiversity in KZN. The selection included more species that occur in the central and southern parts because several of these are either endemic or threatened or both, e.g. red steenbras (*Petrus rupestris*). Several of the endemic and threatened species migrate into KZN waters to breed and feed during the winter months. The strong southward flowing Agulhas current, which comes close inshore where the coastline projects outward, e.g. Port St John's, Durban, and Richards Bay, may form barriers to northward dispersal of winter migrants (Expert Workshop 2008). Sharp changes in SST north of Richards Bay may also limit the northward dispersal of winter migrants that normally occupy the cooler Cape waters (Expert Workshop 2008). The lower water temperatures south of Richards Bay limit several tropical species to the north of the province. Few of the tropical species are endemics. They are often wide-ranging Western Indian Ocean species that occupy large distribution ranges. Therefore KZN includes more endemic species in its southern and central areas, than in the north of KZN. The threatened status of several of these species may partly be attributable to their limited distribution ranges, as the overfished stocks of these species cannot be replenished from elsewhere.

### **Seasonality of distribution patterns**

The seasonal richness distribution patterns agree with current knowledge of seasonal assemblages of the species used in this study (several are migratory species). Two suites of migrants are known to occupy KZN during the two defined seasons. Winter migrants enter KZN waters from the south and extend their feeding and breeding ranges as far north as Richards Bay, but occur predominantly along the south coast of KZN. A second suite of species migrate into northern KZN from more tropical areas to extend their feeding range

during the summer months when water temperatures are higher. These migrants often occur as far south as Durban, but are more common north of Cape Vidal (van der Elst 1988; Compagno et al. 1989; Mann 2000; Heemstra & Heemstra 2004).

The high summer values just north of Umkomaas may be related to a skewed sampling effort. The offshore reef at Umkomaas, Aliwal Shoal, is a popular diving site that may attract more visitors during summer's months, and perhaps explain this anomaly.

Winter migratory species return to feed in the cooler more productive Cape waters during summer. Summer migrants return to the more tropical waters as KZN water temperatures cool down during winter and start limiting tropical fish ranges. It was evident from the data that not all the individuals of these seasonal migrants leave KZN waters at the end of the season. Sufficiently large numbers of individuals stay behind (evident from the large number of records of caught individuals during the low season (MCM 2009)) to model their distribution ranges during the low abundance season. Although Maxent could discern a difference in seasonal distribution pattern, it was not suited to quantify the differences beyond spatial pattern, i.e. the values of the differential seasonal map cannot account for the difference in seasonal abundance of a species. Alternative modelling methods, like GLMs or GAMs, that can utilize abundance data, may be better suited for quantifying the difference between seasonal probabilities of occurrence.

Finally, the temporal division used for the two seasons may not be suited to all species. The influx of migratory species into KZN varies temporally from one species to the next. It would be a worthwhile exercise to model each migratory species using monthly partitions of the data to gain further insight into the onset of their migrations and gain a better understanding of the areas that they occupy during the different months of the year.

Given the data available, a concerted effort was made to develop and identify suitable SDMs for all species included in this study. Fish species are important features to consider in a marine conservation plan. The SDMs that were developed are the biodiversity features that were used in the conservation planning in the next chapter (Chapter Four). The differences in seasonal distribution of biodiversity richness were also used to assess the differences in seasonal conservation plans in Chapter Four.

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Note that the comment and suggestions from the Expert Workshop (2008) and the independent review thereof is available in the database that was made available to Ezemvelo KZN Wildlife and SANBI.

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**Chapter 4**

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# **Spatio-temporal marine conservation planning for fish species in KwaZulu-Natal**

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**Abstract**

Coastal marine conservation planning has largely ignored spatio-temporal variation of distribution and abundance of biodiversity. Nine scenarios that vary in spatio-temporal distribution of features, and in zonation status of existing Marine Protected Areas (MPAs) in KwaZulu-Natal (KZN), were investigated. Data sets included annual distributions of selected marine fish species, as well as summer and winter distributions. MPA zonation scenarios included zones A (sanctuary zones where no fishing is allowed), zones B (restricted zones, restricted fishing allowed) and zones C (controlled zones, open to several forms of fishing), which allow different activities. Conservation targets were set for the features (species' distribution ranges modelled in Chapter 3) based on a standard baseline target and a biological retention target. Resulting targets were then adjusted for seasonal data sets by multiplying them with the 4<sup>th</sup> root transformed value of winter-summer abundance ratios. A conservation status assessment of the current MPA network was undertaken (recognising only zones A as contributing to targets). For further analyses, zones B and C were also

which biological retention targets were added to increase the percentage required to adequately meet the conservation requirements for species that are endemic, rare or have vulnerable life histories (Lagabrielle et al. 2010). In this study, the sum of the baseline and retention targets was adjusted to compensate for differences in seasonal abundance, estimated from catch per unit effort data (CPUE) during winter and summer.

The outcome of the conservation status assessments showed that the current MPA network does not offer adequate protection to the marine fish species considered under any of the nine scenarios. Target achievement was consistently higher for more of the features in scenarios where zones B and C were allowed to contribute to targets. Target achievement for summer assemblages was higher than winter assemblages in all scenarios. The conservation plan allowed us to identify areas that had a high selection frequency to meet unmet conservation targets for the nine scenarios. Areas near existing MPAs in northern KZN, and the Tugela River mouth (in central KZN), and Aliwal Shoal and Margate (southern KZN) had high selection frequencies. The winter selection frequency had more high values in southern KZN, while summer included more high values in northern KZN. The difference in spatial solutions required to meet seasonal targets indicates that conservation plans that are stationary in time and space may not be meeting targets as efficiently as could be achieved by seasonal closures.

FishPLAN selected areas in the southern and lower central parts of KZN more frequently than SeaPLAN to meet currently unmet targets. SeaPLAN's output identified more clustered areas, and had fewer spatial options to meet targets, than FishPLAN. Similarities in areas between the two plans included areas around the Tugela River mouth, Aliwal Shoal and Margate. These similarities may have been driven by the similar features (species and costs) used in the two conservation plans (FishPLAN data were a subset of SeaPLAN data), or by the fact that the marine fish species used in this study are a good surrogate for more complete biodiversity data sets, in places. Differences between the two plans were driven by the additional features used by SeaPLAN, and use of Admiralty zones as areas with their conservation status predetermined for the analysis (existing MPAs) in SeaPLAN. It is recommended that conservation plans include as representative a sample of biodiversity as possible.

## **Introduction**

International and national treaties require increased marine spatial conservation (WSSD 2000), (Jackelman et al. 2007), that achieves representation of biodiversity, as well as

promoting its persistence. The current network of marine protected areas (MPAs) inadequately represents marine biodiversity in South Africa (Lombard et al. 2004; Sink et al. 2010). Ecosystem-based management, and the use of spatial marine reserves, have been shown to be highly successful measures to protect marine fish stocks (Roberts et al. 2001; Gell et al. 2003). The existing network of MPAs in KwaZulu-Natal (KZN) offers only limited protection to an unrepresentative sample of biodiversity (Lagabrielle et al. 2010). The MPAs include the Isimangaliso Wetland Park, an extensive nearshore MPA in the north of KZN, starting north of Cape Vidal in the south and extending to Kosi Bay in the north. The only MPAs in the rest of the province are Aliwal Shoal and Trafalgar, in southern KZN (Figure 1). There are currently no MPAs in central KZN (Richards Bay to Durban), a length (> 200 km) of unique coastal habitat and oceanographic processes that spans nearly a third of the province.

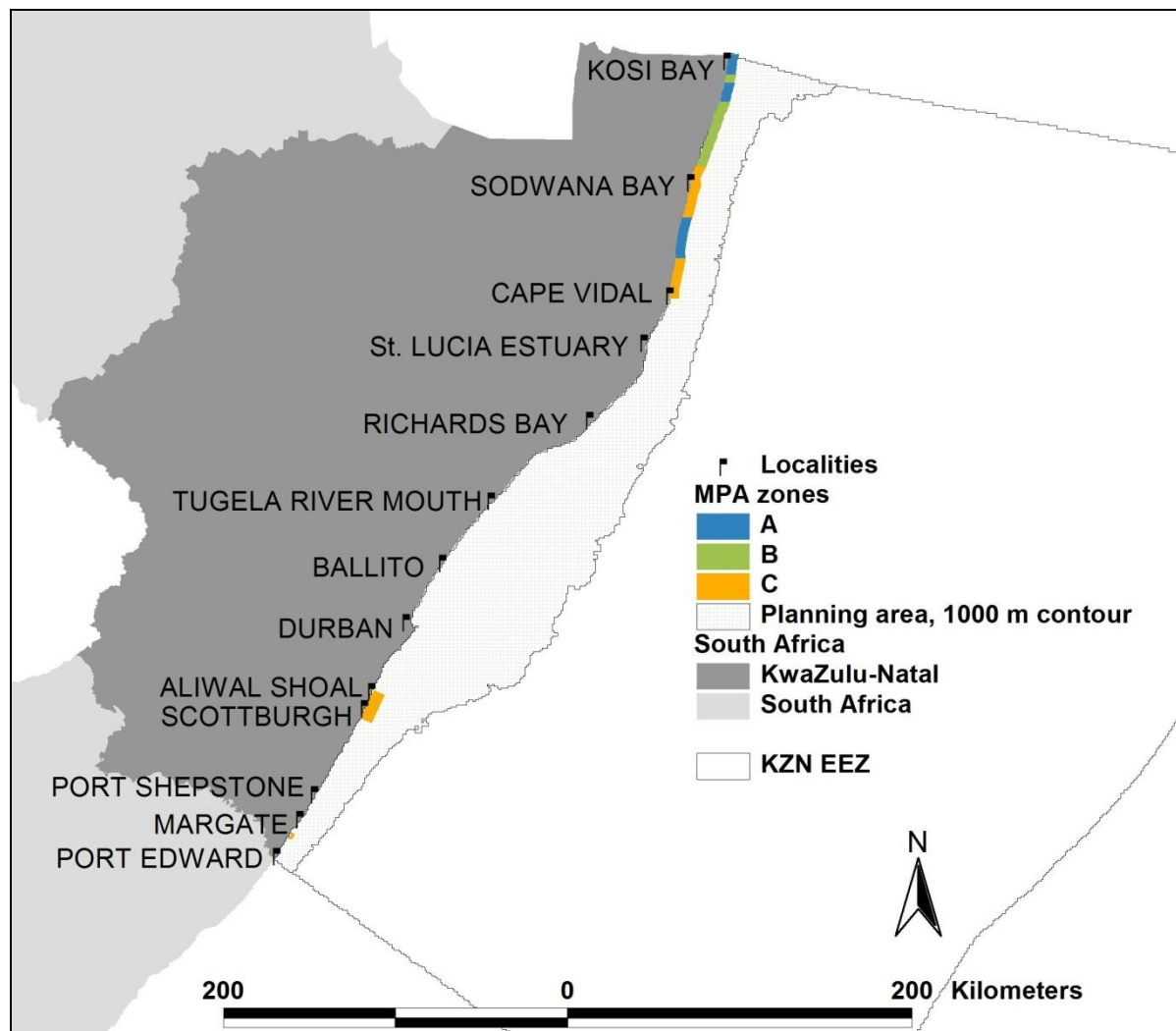


Figure 1. Map of the KwaZulu-Natal Exclusive Economic Zone, showing the planning area and current configuration of marine protected areas. (Refer to General introduction Table 1 for description of MPA categories.)

The *ad hoc* allocation of MPAs in the past has been shown to represent biodiversity inadequately in South Africa, including KZN (Robinson & de Graaff 1994 Gell et al. 2003, Lombard et al. 2004). Systematic conservation planning (conservation planning hereafter) has become the international and national standard for allocating spatial protection (e.g. Cowling et al. 2003; Lombard et al. 2004; Fernandes et al. 2005; Sink et al. 2010) because it specifically aims to meet representative biodiversity targets efficiently, while avoiding areas of high socio-economic cost (Margules & Pressey 2000).

SeaPLAN is the KZN provincial conservation plan that aims to conserve three aspects of marine biodiversity, including habitats, species and ecological processes. Habitats included rock and coral reefs, beach composition, oyster beds and coral reefs, and clusters of

statistically similar habitats based on ocean climatology, bathymetry, and slope. The species included marine fish, mammals, and turtle nesting sites. Ecological processes included chlorophyll fronts, estuarine influence on the marine environment, and migratory pathways of marine mammals. Human uses of the marine environment were mapped and considered as opportunity costs (costly to protect). The human uses included 28 sources that were weighted in terms of their opportunity cost during expert workshops. Areas with high cost were avoided when selecting areas to meet conservation targets (Lagabrielle et al. 2010).

In this Chapter, the fish species distribution models (features hereafter) prepared in Chapter Three were used to investigate three aspects of conservation planning: 1) Report on the current conservation status of the features (a conservation assessment), and 2) investigate spatial solutions to meet unmet conservation targets (targets hereafter) (a conservation plan), and 3) highlight the shortcomings and overlap of conservation plans that are based on subsets of features. The present study is referred to as FishPLAN (the subset), in contrast to the overarching project which is called SeaPLAN (the full data set, which includes the fish data used in FishPLAN). SeaPLAN included distribution models for all the species used in FishPLAN, and thirteen extra species. These extra species were included based on expert recommendation. This criterion was removed from the present study (Chapter One), as it was not considered to be sufficiently objective. All species distribution models in SeaPLAN were modelled using CHARMs, while FishPLAN included a combination of CHARMs and Maxent models.

The aims were investigated under nine scenarios in which the initial protection status of the current MPAs and spatio-temporal differences in biodiversity distribution were varied in combination. The MPAs in KZN include three zones, A, B, and C that have different levels of protection status. The regulations that apply in the MPA zones are similar to Strict, categories defined in the IUCN's Categories System for Protected Areas (IUCN 2010). MPA zones A are strict sanctuary reserves that prohibit all extractive uses and disturbance. Pelagic or game fishing is allowed in MPA zones B, which therefore offer some protection to demersal fish (B. Mann pers. comm.). Limited protection is offered, however, by MPA zones B owing to the inevitable loss of species from catch and release fishing, and the higher disturbance rates than found in MPA zones A (Bullen et al. 2008; Mann et al. 2010). Although MPA zones C have some fishing regulations (e.g. Gazette 2004), fishing pressure is often higher inside than outside of these partially protected areas (B. Mann pers. comm.). MPA zones A, which strictly prohibit disturbance, and allow no fishing, are therefore the best form of protection to restore and conserve fish and other biota (Denny & Babcock 2004). In



this study the strict protection offered by A zones was applied to the B and C zones in different conservation planning scenarios.

The sum of the species distribution ranges that were modelled in Chapter Three showed seasonal spatio-temporal variation in the distribution patterns of marine fish species. The same three temporal scenarios (annual, summer and winter) were used in combination with the three protection scenarios (i.e. nine scenarios in total). It was expected that the differences in the biodiversity distribution patterns would result in differences in target achievement in existing MPAs, and therefore require different spatial solutions to meet unmet targets.

The conservation status assessment reports on target achievement for the biodiversity features considered in a conservation plan and identifies gaps in the existing network of MPAs (Lombard et al. 2004). Spatial conservation targets are often set to specify the proportion of the area required per biodiversity feature to provide adequate protection to conserve it into the future (persistence) (Svancara et al. 2005; Sink et al. 2010).

A minimum target of 20 - 30% of all biodiversity features and ecological processes was recommended during the World Summit on Sustainable Development (WSSD 2002), Durban South Africa. The 20 - 30% target was based on a study by Bohnsack et al. (2000), which argued that 20 - 30% of spatial protection would adequately protect spawner biomass potential (SBP) from species collapse, as it conserves more than the Maximum Sustainable Yield (MSY) requirements. Agardy et al. (2003) criticised the target as being *ad hoc* because its origin was based on localised studies of specific fisheries within particular habitats, and therefore does not represent ecological communities or a wide range of habitats.

Population viability analyses (PVA) have been used for data rich species with well understood ecological needs to determine adequate targets (e.g. Carroll et al. 2003; Goodman 2009). The lack of information on most marine species precluded the use of PVA-derived targets.

As yet, no definite resolution regarding quantitative targets has been reached. Although the author agrees with Agardy et al. (2003) that targets should be scientifically defensible, the lack of information should not delay conservation efforts. Policy-based targets are likely to be under estimates of the area required for adequate protection for persistence (Svancara et al. 2005). Until the necessary information and techniques become available to set scientifically defensible targets, a precautionary approach is recommended. In SeaPLAN a baseline (minimum) target was set using the WSSD (2000) minimum recommendations of 20%, to

which biological retention targets were added to increase the percentage required to adequately meet the conservation requirements for species that are endemic, rare or have vulnerable life histories (Lagabrielle et al. 2010). In this study, the sum of the baseline and retention targets was adjusted to compensate for differences in seasonal abundance, estimated from catch per unit effort data (CPUE) during winter and summer.

The importance of considering spatio-temporal changes in species distributions has been used in identifying offshore MPAs for pelagic stocks off Australia (Hobday & Hartmann 2006). Thirty-five pelagic fish stocks' distribution ranges were modelled to four seasons, using 60-day ocean climate data, and mobile MPA networks were identified for the four seasons. Grantham et al. (2008) showed that temporally closed areas are efficient to protect bycatch species and minimize cost to fishers in the South African pelagic longline fishery. Like pelagic fish species, spatio-temporal distribution and abundance of marine fish assemblages vary seasonally in KZN (van der Elst 1988). The differences in seasonal distribution patterns were modelled in Chapter Three. The conservation targets adjustments were based on the seasonal ratio of catch per unit effort (CPUE) per species, derived from the catch data evaluated in Chapter Two. The planning domain was limited to the 1000 m depth contour as none of the features extended beyond this depth zone. The planning domain was divided into 21546 one km<sup>2</sup> square planning units to assess target achievement in the existing network of MPAs, and derive spatial solutions to meet unmet conservation targets for the nine scenarios.

Conservation targets were adjusted for seasonal abundance. The differences in seasonal distribution patterns and seasonally adjusted targets were expected to produce different results during target assessment and spatial solutions. The target achievement shortcomings of a stationary reserve network for features that vary in distribution and abundance was evaluated by using the best performing stationary (annual) spatial solution with MPA zone A status to evaluate seasonal targets achievement.

One hundred efficient spatial solutions were calculated to meet currently unmet conservation targets, for each scenario. The spatial solutions calculated in Marxan uses simulated annealing, and every solution is not necessarily unique. The total number of times a particular planning unit was selected as part of the spatial solution sum of the 100 spatial solutions was used to create an irreplaceability surface that represents the conservation value of the planning unit to meet targets within the planning area. The irreplaceability surfaces were compared between the scenarios to establish if different spatial solutions were better suited to meet targets for temporally varying biodiversity patterns.

Conservation plans that consider only a focal subset of biodiversity may fail to identify critical areas for conservation, in particular where endangered species occur because of the mismatch in the distribution of different species and groups of biodiversity (Carroll et al. 2003; Cowling et al. 2004). The preliminary SeaPLAN project results were compared with the present study's results.

Conservation target setting, the conservation status assessment, and irreplaceability surfaces of the nine scenarios are discussed, and recommendations are based upon the outcome of the results.

## **Methods**

### **The planning domain**

The planning domain, the KZN Exclusive Economic Zone (EEZ), described in Chapter One, was limited to the 1000 m depth contour as none of the conservation features used extended any further offshore. A total of 21546 planning units were prepared at a scale of one km<sup>2</sup> to match the scale of the features and the maximum extent of the planning domain.

### **Features**

#### **Fish species distribution models**

The three temporal groups (annual, summer and winter) for which species distribution models (SDMs) were prepared in Chapter Three, were used as the conservation features for a conservation assessment and the conservation plans.

#### **Cost feature**

The opportunity cost GIS layer that was prepared for SeaPLAN was used in this study as a cost feature (Figure 2). The cost feature was constructed as a weighted surface from 28 human uses, e.g. trawling, recreational fishing, scuba diving, etc. The human uses were weighted based on their impact on marine biodiversity by a panel of experts during the design phase of the SeaPLAN project (Lagabrielle et al. 2010). The opportunity cost surface was scaled from 0 to 100, and resampled to match the scale and extent of this conservation plan. I used maximum value in the planning unit to determine the coarser resolution cost value, to adequately represent the high values of human uses restricted to the nearshore environment.

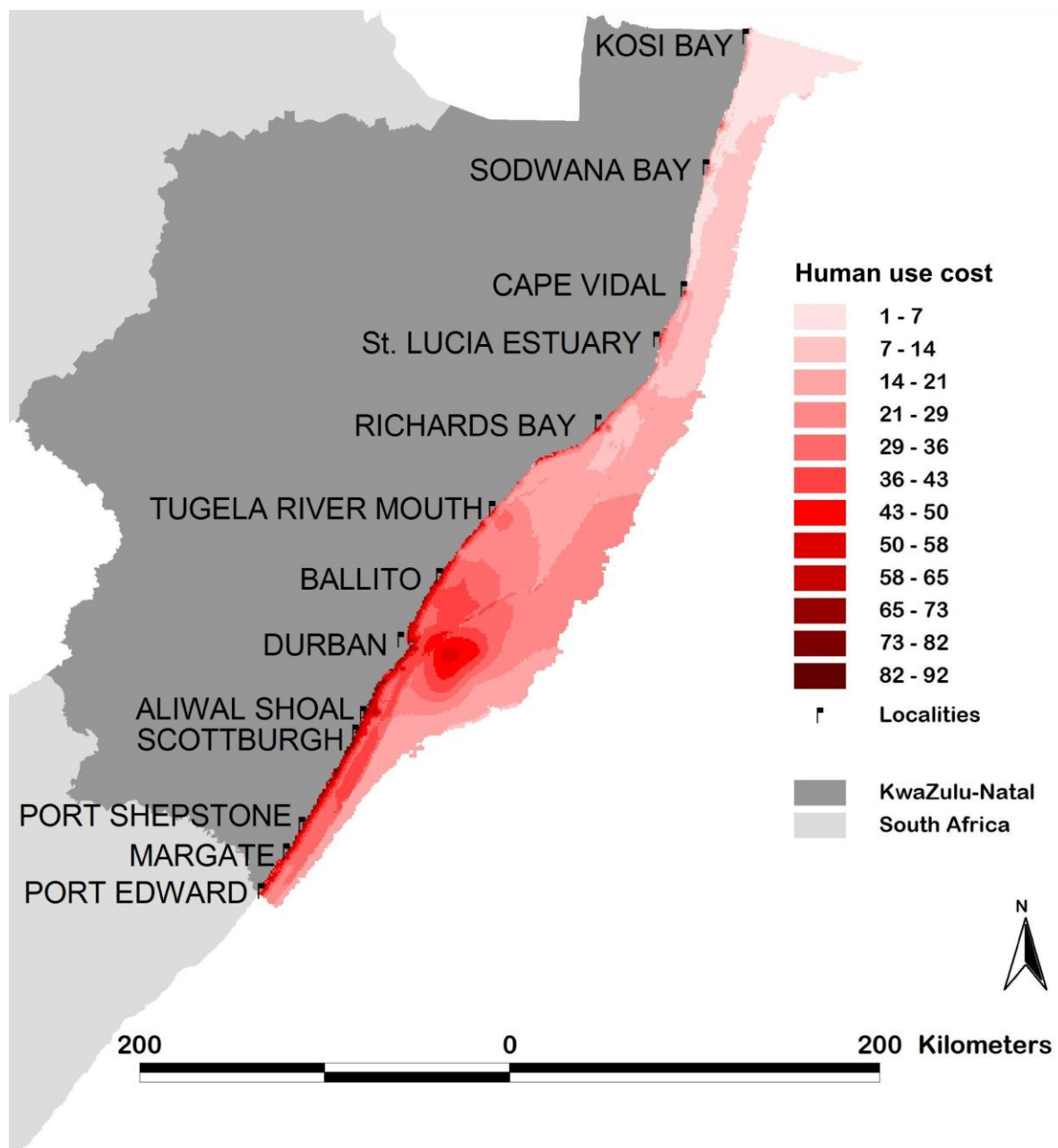


Figure 2. Standardised human use cost feature out of a possible 100 in KwaZulu-Natal.

## Scenarios

In three different starting point scenarios (Table 1), MPA zone A status (sanctuary zones where no fishing is allowed) was iteratively attributed to the existing MPA zones A; then to both existing zones A and B (i.e. both zones A and B were assumed to be sanctuaries); and finally to existing zones A, B and C (i.e. all three zones were assumed to be sanctuaries). This equates to the terminology: MPA A; MPA AB; and MPA ABC. These three scenarios were used as starting points in conservation planning runs or analyses (starting points are

initial areas that have their conservation predetermined before the run, i.e. they are initial reserves which contribute to meeting targets).

**Table 1. The three different starting point scenarios, showing which existing MPA zones were assumed to have the same status as existing A zones.**

	<i>Scenarios</i>		
	<i>MPA A</i>	<i>MPA AB</i>	<i>MPA ABC</i>
Assumes regulations of MPA zones A (sanctuaries) apply to:	Existing MPA zones A only	Existing MPA zones A and B	Existing MPA zones A, B and C

The three temporal data sets were the suite of species distribution models (SDMs) which were modelled per temporal division, summer, winter and annual, in Chapter Three. The nine scenarios were a combination of the three different starting point scenarios (Table 1), and three temporal data sets (Table 2).

**Table 2. Nine conservation planning scenarios with different starting points and temporal data sets.**

<i>Data set</i>	<i>Starting point scenarios</i>		
	<i>MPA A</i>	<i>MPA AB</i>	<i>MPA ABC</i>
Annual	Annual MPA A	Annual MPA AB	Annual MPA ABC
Summer	Summer MPA A	Summer MPA AB	Summer MPA ABC
Winter	Winter MPA A	Winter MPA AB	Winter MPA ABC

### Conservation targets

Conservation targets were prepared for the features using the values of the separate targets to determine the final conservation target:

- Baseline target

- Biological adjustment target
- Seasonal adjustments of abundance estimates used to inform the seasonal target adjustments

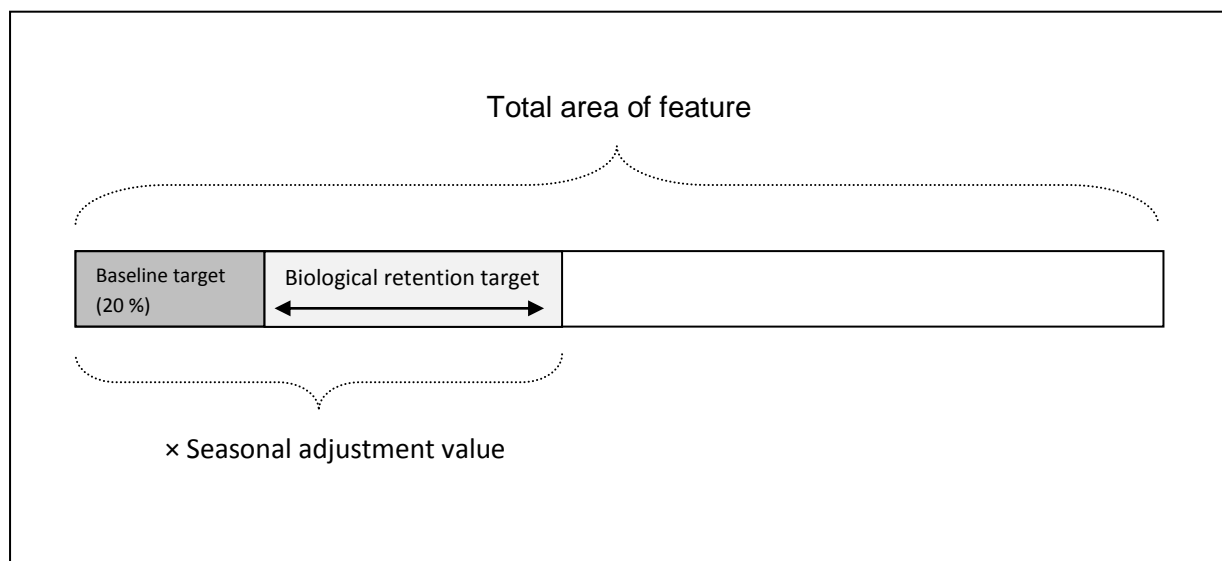


Figure 3. Flow diagram explaining target setting. A baseline target of 20% was augmented with a biological retention target, multiplied with the seasonal adjustment value

The baseline target was set at 20% based on the minimum targets set for biodiversity, as part of the Millennium Development Goals (MDGs) during the World Summit on Sustainable Development (WSSD 2002).

### Biological retention targets (y)

Conservation targets were increased for species with biological attributes that increase their vulnerability of extinction. The Biological characteristics were the following:

#### Localised distribution

- Range restricted endemics = 20%: Species occurring only in KZN, for which their total range is less than 25% percent of the KZN coastline length.
- East Coast = 10%: Species that are endemic to the Agulhas + KZN + Delagoa bioregions (see Chapter One).
- Southern Africa = 5%: Species that are confined to southern Africa; from the Kunene River in the west, to the northern Mozambique border.

#### Rare species

- Localised rare species = 10%. Localised rare species have only limited populations that can be conserved and require increased conservation effort (from Chapter One).

**Life history vulnerability**

- Species with vulnerable life history processes: spawning and reproduction areas = 20%. Spawning areas and areas associated with shark reproduction are a critical part of a fishes' life cycle. These areas are often also targeted in fishing operations, and therefore require increased protection.

**Seasonal adjustment value (m)**

The seasonal abundances were estimated for SeaPLAN fish species present during the two defined summer (January – June) and winter seasons (July – December) using catch per unit effort (CPUE) data from three sources: NMLS recreational shore and ski-boat data base; KZN Shark Board net and drum line catches; and whale shark counts from flights conducted by KZN Sharks Board, ORI, and KZN Wildlife.

CPUE was calculated for the NMLS recreational data base by totalling the catch per species which was divided by the hours spent fishing and the total number of fishermen (ski-boat data) over the two seasons from 1985 – 2009.

- $CPUE = ((\sum_t NMLS \text{ recreational ski \& shore data } (spp_i)) / \sum \text{Hours fishing}) / \sum \text{fishermen}$

where t = seasonal grouping of months, and i species is = species 1 to 66. Records without hours fishing were not used.

KZN Sharks Board catches were grouped per season, without further calculations of effort. Although net length has varied considerably over the years (Dudley & Simpfendorfer 2006), our calculations include all years, and I assumed that the two seasons were affected equally. Although the effect of net removal during the sardine run influences CPUE, the nets are predominantly removed during June and July, and hence affect the groupings of seasons equally (S. Dudley pers. comm.).

- $CPUE = \sum_t \text{Catches } (spp)$ , from KZN Sharks Board data

The number of counts per flight (CPF) conducted to count the number of whale sharks was used as an estimate of abundance.

- $CPUE = \sum_t CPF$ , where CPF is the number of sightings per flight

The seasonal abundance estimates per species were then divided by one another to yield a ratio (r).

- $r = [\text{summer abundance}] / [\text{winter abundance}]$

The data set values of ratio ( $r$ ) were normalised using (absolute value of the) 4<sup>th</sup> root transformation, owing to the highly skewed data as a result of outliers (highly seasonal species). Power transformation of the data set was explored to find a suitable transformation to normalise the data set (including log, square root, cube root and 4<sup>th</sup> root). Fourth root transformation is not commonly used in the literature, but was the most suitable, as the distribution of  $r$ -values was approximately a chi-squared distribution, and better approximated a normal distribution after transformation than the other methods investigated. Fourth root transformation has been used by Warwick (1988) to normalise similar highly skewed (chi-squared distributed) count data sets. The resulting multiplier values were between 0.53 and 1.86 (Figure 4).

- $m = (|{}^4\sqrt{r}|)$

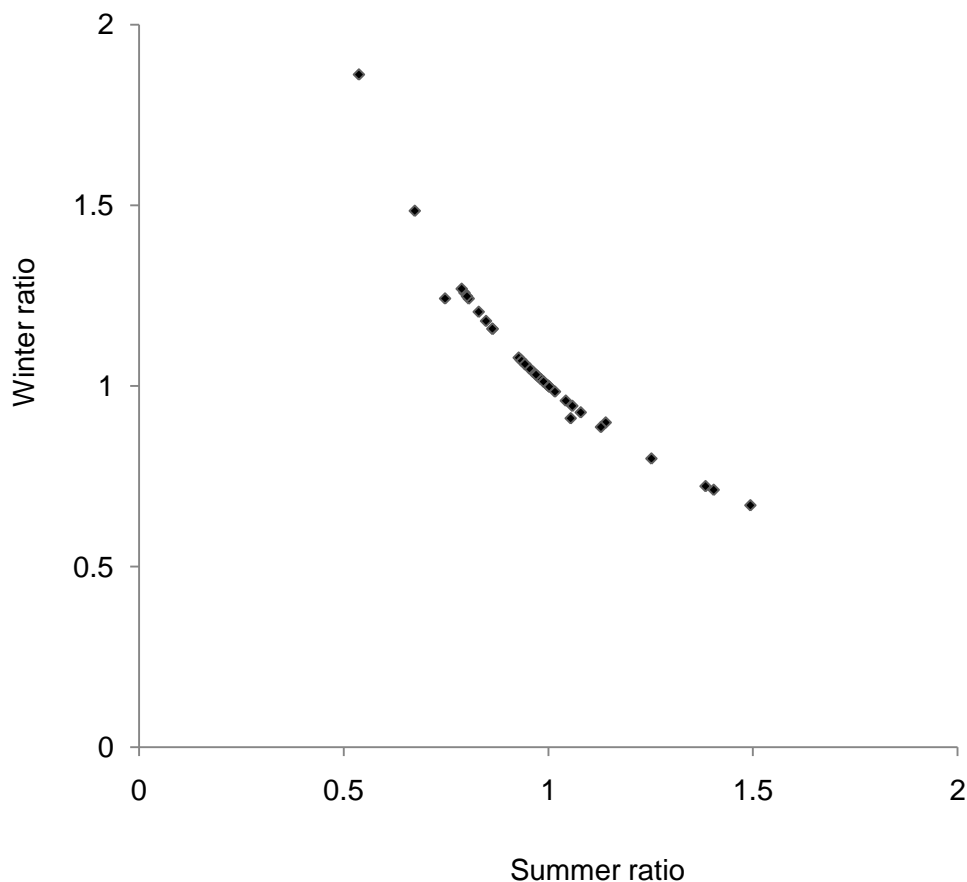


Figure 4. Seasonal adjustment multiplier value was calculated from 4<sup>th</sup> root transformed winter to summer ratio and applied to adjust seasonal targets for each feature.



The final target value was calculated by multiplying the seasonal adjustment multiplier with the sum of the baseline and biological retention targets. The multiplier value was kept at one for species without data, and for all annual data set target calculations.

- Target (spp) =  $(x + y) * m$

### **Seasonal target achievement**

Target achievement was calculated as the percentage of a given feature's conservation target that was met within the starting point reserve network. The current MPA network was used as the starting point, as explained above (Scenarios). The target achievement was repeated for all nine scenarios.

The targets set, and targets achieved, were compared for the corresponding seasonal features (that were included in both summer and winter).

Subsequent to running the conservation planning software Marxan (see below), the output of annual scenario MPA zone A was used to report on target achievement, for seasonal features, i.e. the best solution of the areas required for unmet conservation targets under annual scenario MPA zone A was used as the reserve network against which summer and winter target achievement was reported. The shortcomings in seasonal target achievement under this scenario are reported and discussed. The overall shortcoming of the season is calculated as the % target missing per feature multiplied with the total area of the feature.

### **Spatial conservation prioritisation**

Marxan was subsequently run to meet currently unmet conservation targets under the respective scenarios. The selection frequency outputs are presented as a measure of irreplaceability in terms of meeting conservation targets most efficiently. Irreplaceability is a measure of the value of each planning unit to meeting conservation targets, and the selection frequency is the number of times a planning unit is selected in the 100 runs that were performed for each scenario. Note that not all 100 runs produce different results necessarily.

The differences in seasonal selection frequencies were used to contrast seasonal solutions. The procedure was repeated for each of the temporal data sets. Marxan also produces the best solution, which is the minimum number of planning units that is required to meet all targets at the lowest possible cost. Marxan uses simulated annealing and an objective to produce its results. The best solution is achieved when the objective function is minimised.

### **GIS and conservation planning software**

Data were prepared using the GIS ESRI ArcView 3.2. The following three public domain software packages were used do the conservation assessment and run conservation plans:

- Marxan v2.11
- Map Window v4.5
- Zonae Cogito v1.24

### **Marxan parameter settings**

Calibration scenarios were run on the annual data set to determine a suitable boundary length modifier (blm) and species penalty factor (spf). Exponential calibration was followed by equal interval calibration to fine tune initial values used. Values that achieved minimum score, cost, and maximum target achievement were used. A blm of 0.5, and spf of 100 was determined to clump reserve selection, minimise cost and number of planning units required to meet unmet targets, while still achieving all targets.

### **Comparing FishPLAN and SeaPLAN**

The preliminary selection frequency for SeaPLAN is presented, along with the FishPLAN annual MPA zones ABC scenario, to explore differences (this FishPLAN scenario was the most similar scenario to the SeaPLAN one with respect to the features used, targets set, and the starting point reserve network used). The Marxan parameters were adjusted to match those used in SeaPLAN, and the results were rerun. The blm was adjusted to 100, and the species penalty factor to 1000000. The selection frequency was subtracted from the SeaPLAN selection frequency to show the differences and similarities between the two plans.

## **Results**

### **Target setting**

Species with small endemic ranges, rare species and reproductive areas had the highest baseline and retention targets. After seasonal adjustment values were applied, features with highly seasonal trends had their targets changed accordingly. The change resulted in higher or lower targets for seasonal features, sometimes exceeding the targets of other rarer or endemic species (Figure 5). Seasonal adjustments were made to 39 features. All other features per data set retained their annual targets, which were based only on baseline and retention criteria. The difference between seasonal adjusted targets indicates the greater or lesser proportion of range required to meet conservation targets for the respective seasons.

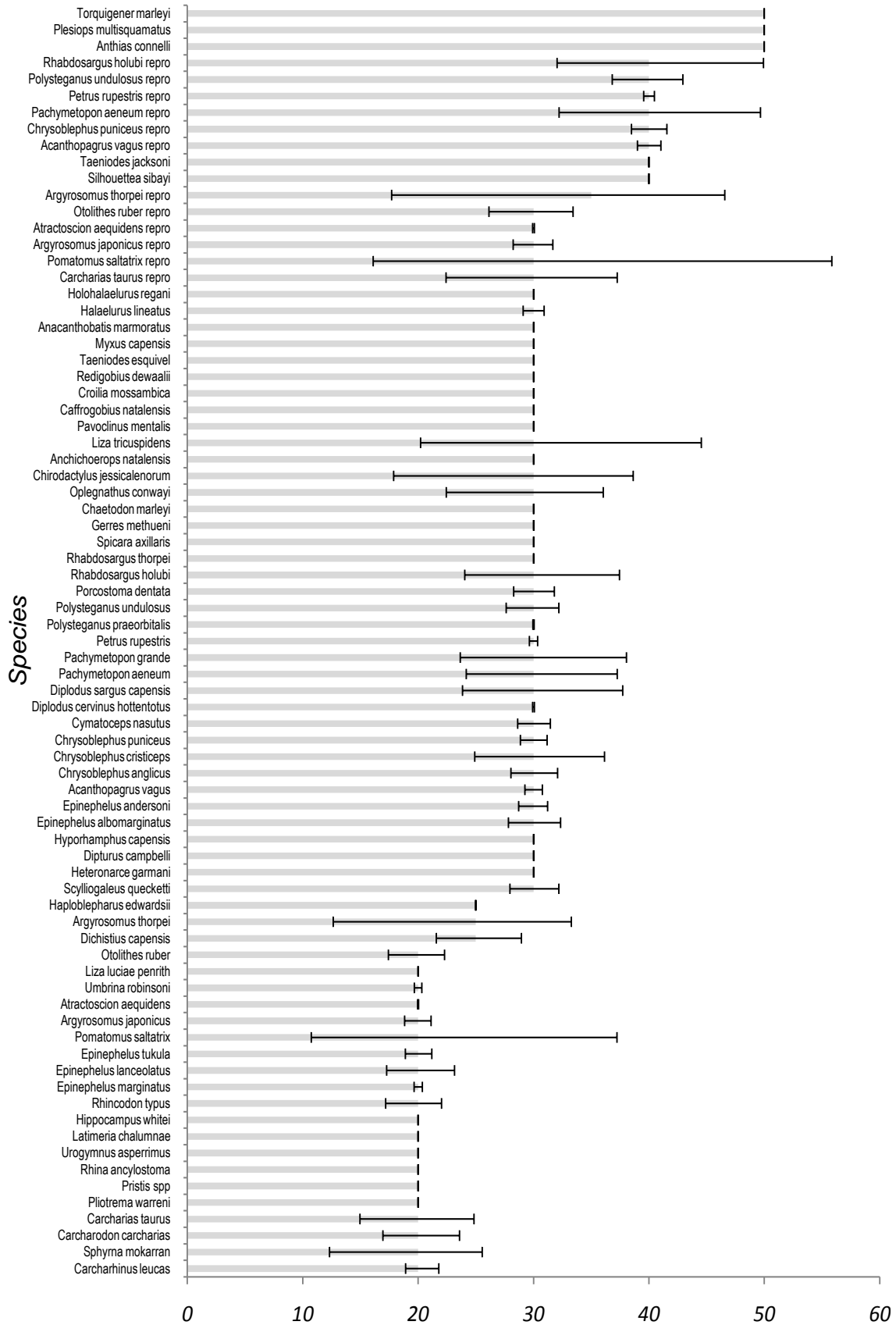


Figure 5. Annual conservation targets ( $\pm$  seasonal adjustments). The histogram indicates the features baseline + retention targets, while the black bars above and below the

histogram indicates the final target after seasonal adjustment was applied. Those species without black bars are species for which no seasonal adjustments were made. The suffix “repro” indicates areas associated with reproductive behaviour. Summer and winter changes are not equal, owing to 4<sup>th</sup> root transformation of seasonal abundance ratios.

Thirteen features had higher targets in summer, while 26 features had higher targets during winter (Figure 6). Most of the species with differences in their seasonal targets set were less than 10% of their total ranges; but four features had a difference > 10% of their ranges between summer and winter targets (i.e. higher summer targets): Squaretail kob (*Argyrosomus thorpei*) (20%) and Natal fingerfin (*Chirodactylus jessicalenorum*) (20%) had the greatest difference between summer and winter targets. During winter, 10 features had a difference >10% of their ranges between summer and winter targets (i.e. higher winter targets): Shad (*Pomatomus saltatrix*) (39%) and striped mullet (*Liza tricuspidens*) (24%) had the greatest differences (Figure 6).

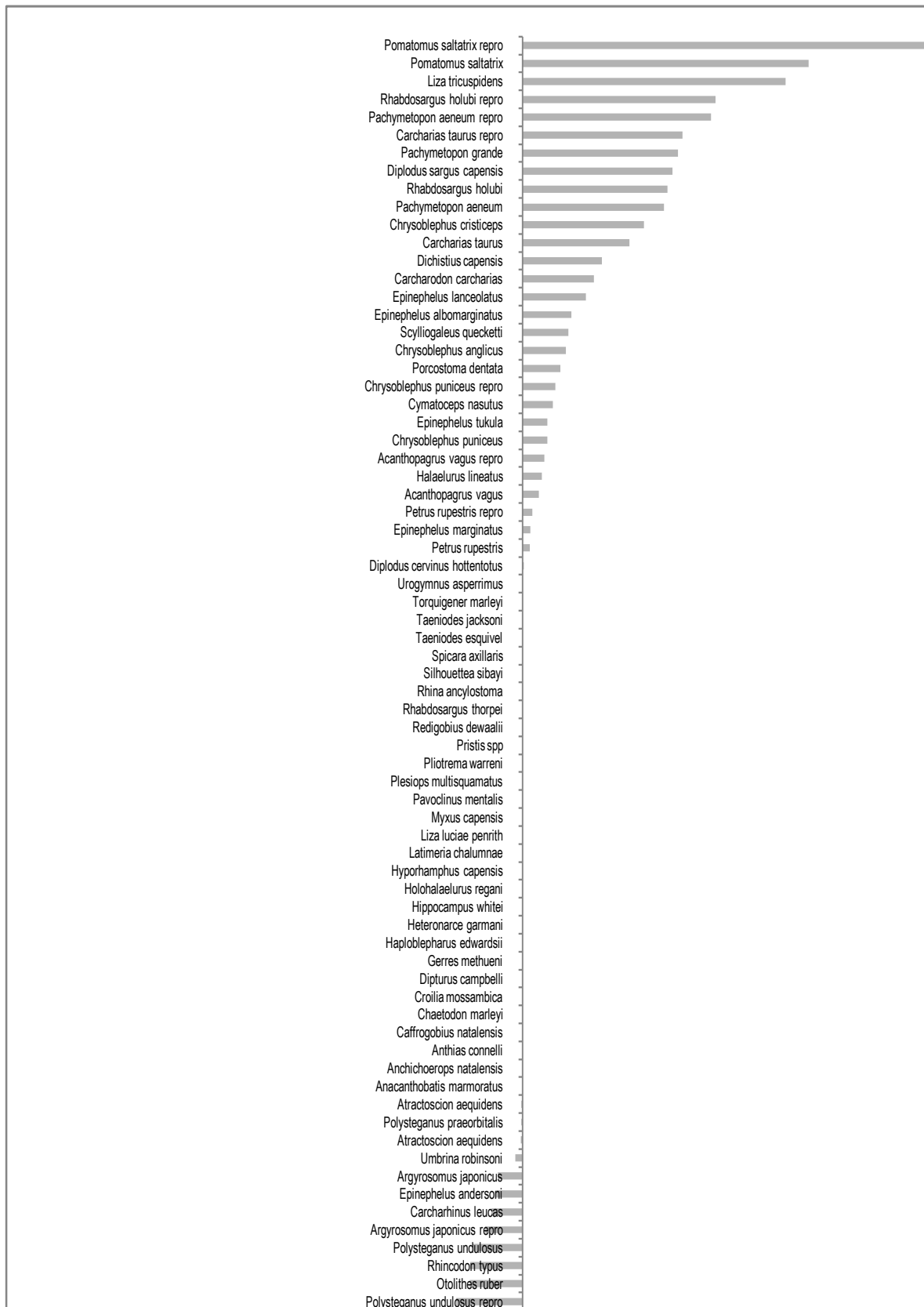


Figure 6. Difference between seasonal targets (percentage of range) per species, based on abundance estimates. Negative values indicate higher summer targets, and positive values indicate higher winter targets.

## Conservation assessment: target achievement of current MPAs

### Annual data sets

Using the annual data sets of all species, and allowing only existing MPA zones A to contribute to target achievement, I found the following results: slinger (*Chrysoblephus puniceus*), coelacanth (*Latimeria chalumnae*), potato bass (*Epinephelus tukula*), Sibaya goby (*Silhouettea sibayi*) (marine range), and the whale shark (*Rhincodon typus*) had the highest proportion of their conservation targets achieved (Figure 7). Slender puffer (*Torquigener marleyi*), squaretail kob (*A. thorpei*) and snapper kob (*Otolithus ruber*) did not have any conservation targets met by current MPAs.

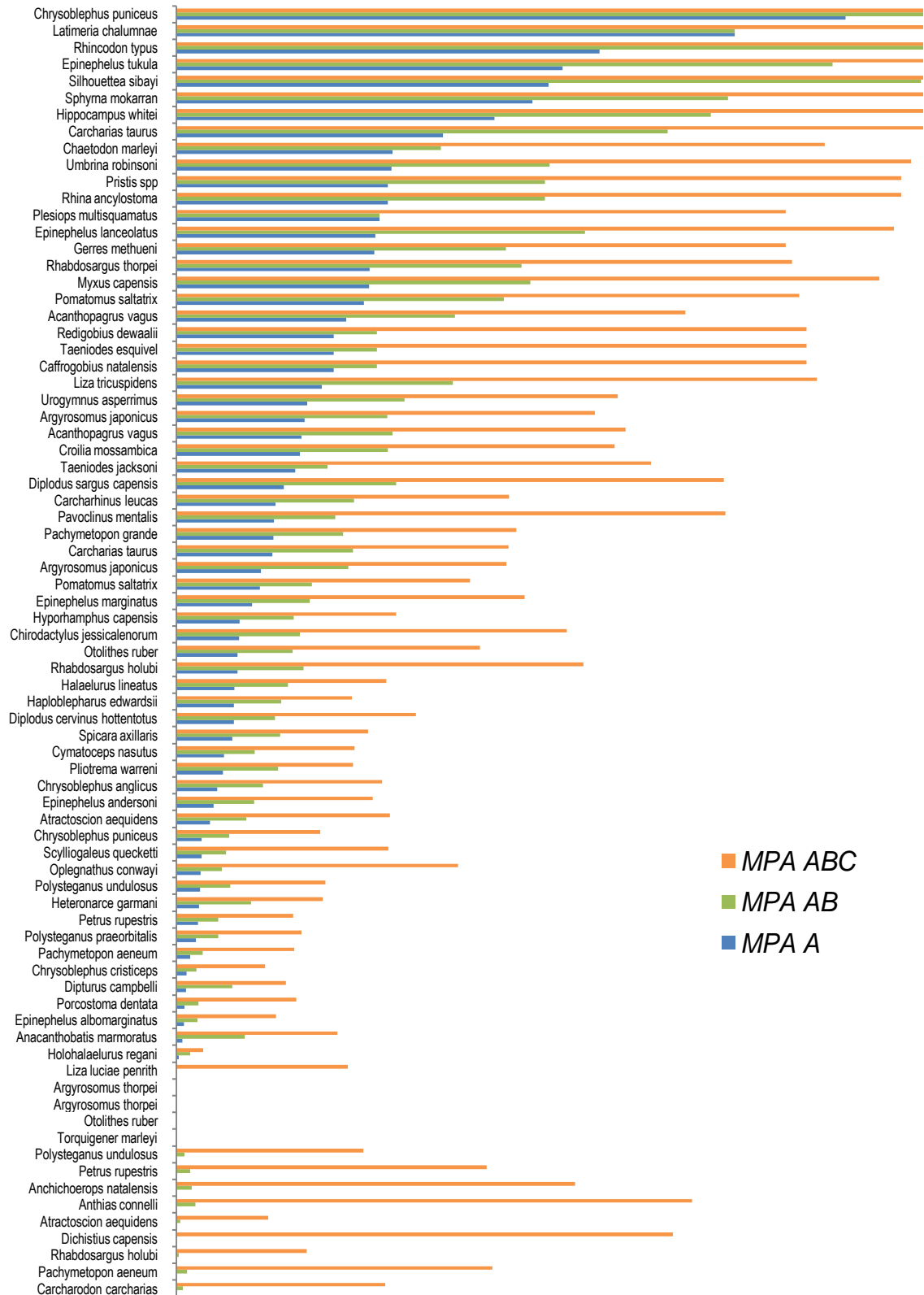


Figure 7. Annual data set target achievement under the three protection scenarios: (i) MPA zones A contribute to targets (MPA A); (ii) MPA zones A and B contribute to targets (MPA AB); and (iii) MPA zones A, B and C contribute to targets (MAP ABC).

As expected, greater percentages of targets were achieved when more zones contributed to targets (Figure 8 and Figure 9). Under scenario MPA A, 41 of the 77 features had 10% or less of their targets achieved, and no conservation targets were fully achieved (Figure 8). Under scenario MPA AB, 28 of 77 features had 10% or less of their targets achieved, and slinger (*C. puniceus*), Sibaya goby (*S. Sibayi*), whale shark (*R. typus*) conservation targets were fully achieved (Figure 8). In scenario MPA ABC, only five of 77 features had 10% of their targets achieved and eight features either achieved or over-achieved their targets: Coelacanth (*L. chalumnae*), potato bass (*E. tukula*), crowned seahorse (*Hippocampus whitei*), great hammerhead (*S. mokarran*), ragged-tooth shark (*Carcharias taurus*), and the aforementioned features in scenario MPA AB. In scenarios MPA A, AB and ABC, 70, then 67, and then only 42 of the 77 features had less than 50% of their targets achieved, respectively.

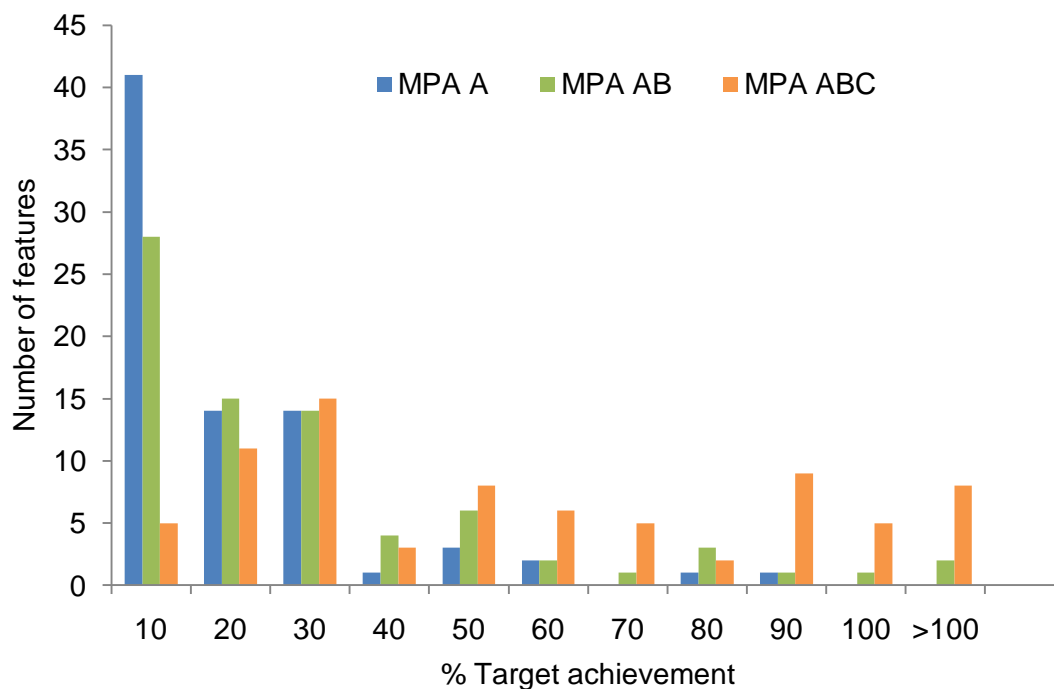


Figure 8. Number of annual features and percentage conservation target achieved by the three protection scenarios (i) to (iii) (see Figure 7).

### Seasonal target achievement

The average target achievement was highest during summer and lowest during winter (Figure 9). Individual target achievement followed a similar pattern for annual, summer and winter (Figure 7 in Appendix 7). As for the annual data set, higher percentages of seasonal targets were achieved as MPA zones B and C were added to the MPA network (Figure 9).



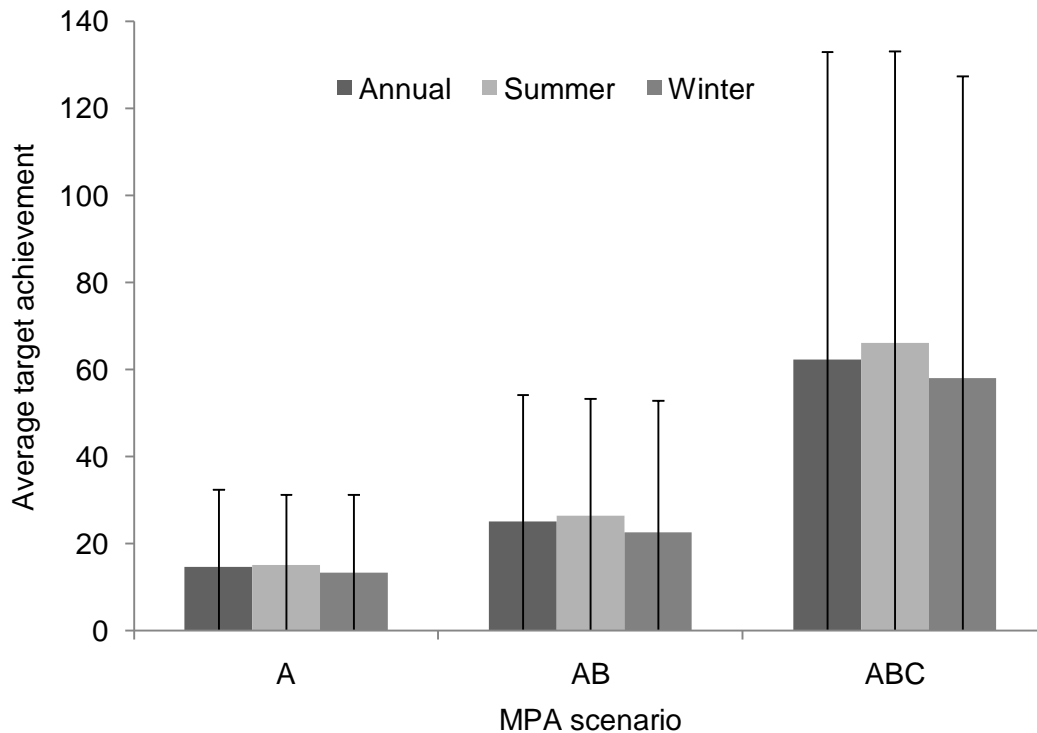


Figure 9. Average ( $\pm$  stdev) annual and seasonal targets achieved by the three protection scenarios (i) to (iii) (see Figure 7). Three data sets: Annual  $n = 77$ ; Summer = 71; Winter = 76.

The number of features with higher target achievement in the current MPAs was greater in the low abundance season than the high abundance season (Chi-square,  $p = 3.57E-17$ ), (Table 3). Differences in seasonal distribution ranges also contribute to difference in target set and achieved in 26 of the 39 features for each MPA scenario. No attempt was made to separate the relative influence of each.

**Table 3. Number of features with higher, no difference, or worse target achievement between high and low abundance season in current MPAs. Fewer features had their targets achieved during the high abundance season, than the low abundance season.**

<i>Target achievement in two abundance seasons</i>				
<i>MPA configurations</i>	<i>Low &gt; High</i>	<i>High = Low</i>	<i>High &gt; Low</i>	<i>Total</i>
MPA A	25	5	9	39
MPA AB	28	2	9	39
MPA ABC	29	1	9	39
TOTAL	82	8	27	117

### **Seasonal target achievement in the scenario with annual data and MPA zones A**

The scenario with annual data and MPA zones A produced a best solution (Figure 5 in Appendix 7) that met all targets. The best solution of the annual data set did not meet all the targets for the winter or summer data sets, in particular the Natal fingerfin (*Chirodactylus jessicalenorum*), Cape knifejaw (*Oplegnathus conwayi*) and squaretail kob (*A. thorpei*) during summer and shad (*Pomatomus saltatrix*), dageraad (*Chrysoblephus cristiceps*), blue hottentot (*Pachymetopon aeneum*), Cape stumpnose (*Rhabdosargus holubi*), striped mullet (*Liza tricuspidens*) and red steenbras (*Petrus rupestris*) during winter. The shortfall of target achievement of species range was totalled per season under the annual best solution. The total seasonal shortfall was greater during winter (1636 summed target achievement of total range shortfall) than during summer (198 summed target achievement of total range shortfall), i.e. the annual solution met much fewer of the winter targets than summer targets.

### **Selection frequency irreplaceability surface**

#### **Annual data set, using only MPA zones A as contributing to targets**

Areas surrounding MPA zones A were selected 80 – 90 times out of 100, as part of the a reserve network solution to meet unmet conservation targets (Figure 10). An area north of the Tugela River mouth, the Durban offshore reefs and the area between Aliwal Shoal and Scottburgh had high values for selection frequencies. A few smaller areas also had relatively high selection frequency values: Sodwana Bay, two small inshore areas between Richards Bay and the Tugela River mouth, and the offshore reefs between Margate and Port Edward.

The northern KZN inshore areas between Kosi Bay and south of St. Lucia, excluding the existing MPA zone As, Sodwana Bay and their immediate neighbouring planning units, had far lower selection frequency values.

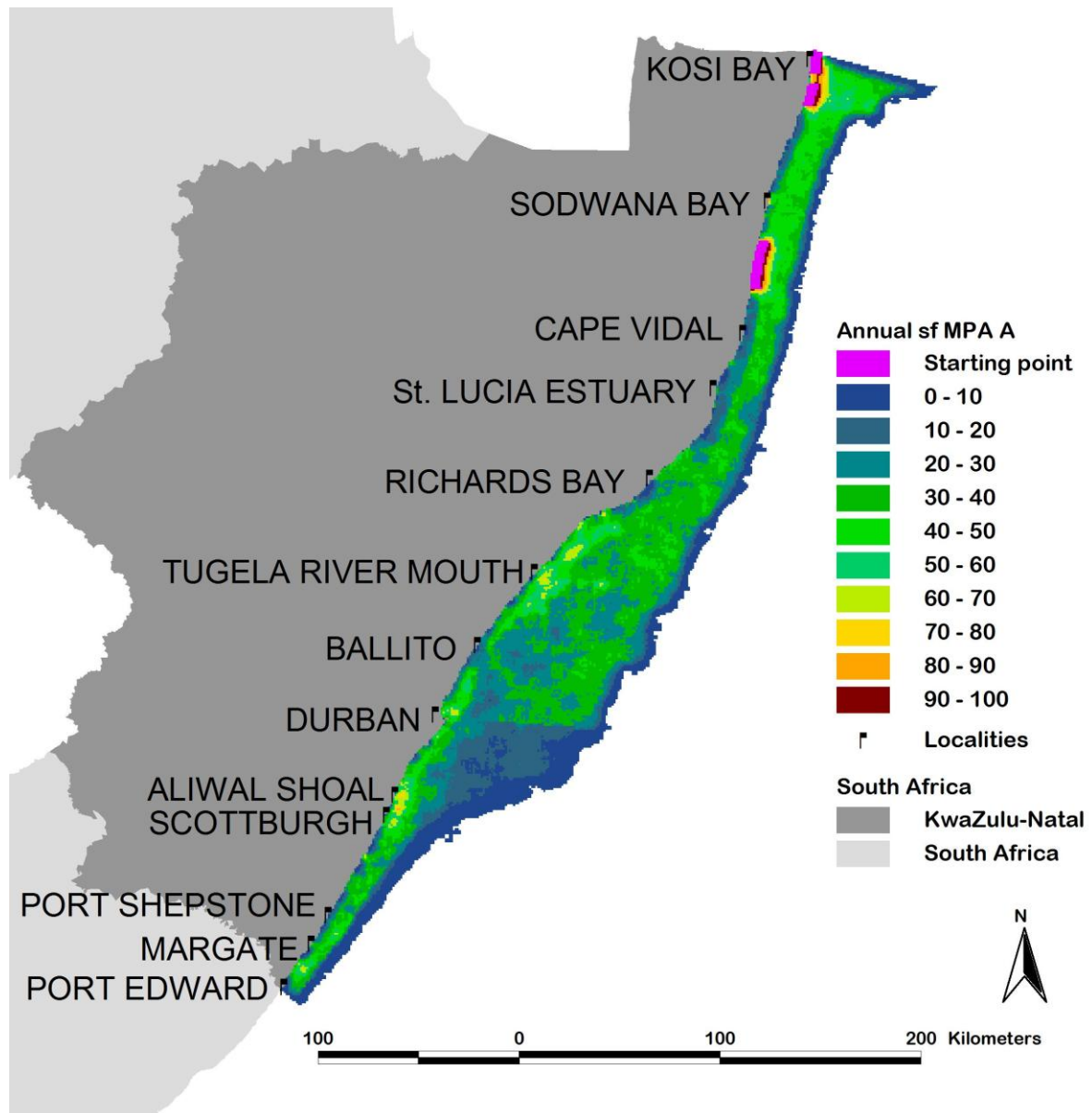


Figure 10. Annual selection frequency (sf) for scenario (i) where MPA zones A contribute to targets.

#### Annual data set, MPA zones A and B contribute to targets

The inclusion of MPA zones A and B as contributing to targets did not change the overall selection frequency pattern by much (Figure 11). The most notable change was in the values of planning units around Aliwal Shoal which were selected 90 – 100 times. This was as a

result of including the Produce wreck and Aliwal Crown MPA zones B as contributing to targets (see Figure 1).

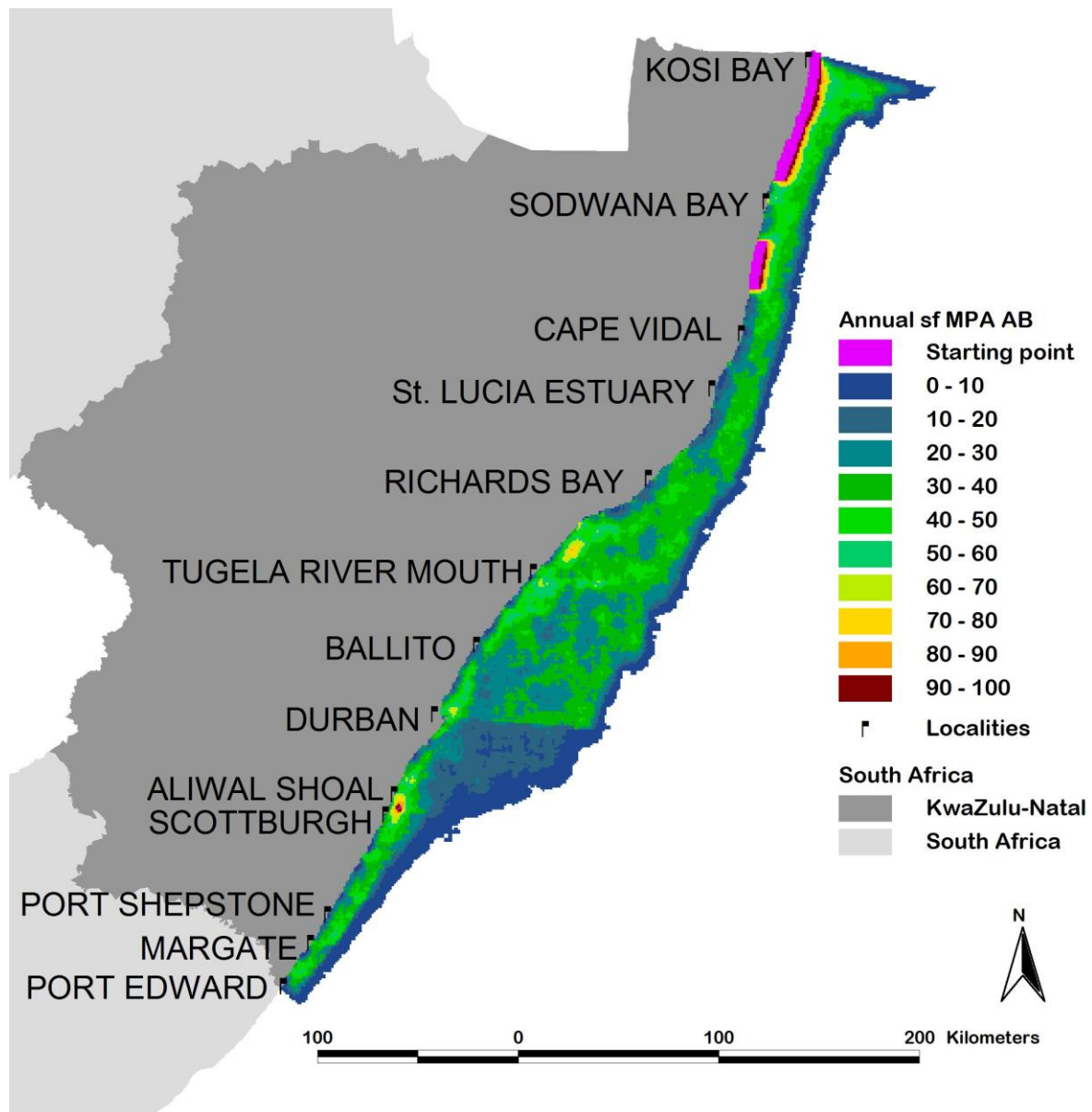


Figure 11. Annual selection frequency (*sf*) for scenario (ii) where MPA zones A and B contribute to targets.

#### Annual data set, MPA zones A, B and C contribute to targets

All MPA zones A, B and C contributed to targets in this scenario, and consequently, the extensive nearshore area in northern KZN acted as the starting point. This decreased the selection frequency of planning units to the immediate south, namely from Cape Vidal to south of Richards Bay (Figure 12). Two areas offshore from the Tugela River mouth had relatively high selection frequencies. The area around Aliwal Shoal was larger than under the two previous scenarios. The area around Trafalgar was included in the starting point

reserve network. Notably, the selection frequency of the area surrounding Durban decreased, with the higher selection frequencies of the aforementioned areas along the KZN south coast (Compare Figures 10 – 12). The selection frequency of the area north-and-offshore from the Tugela River mouth increased slightly by adding the existing MPA zones B and C to the starting reserve network (Compare Figures 10 – 12).

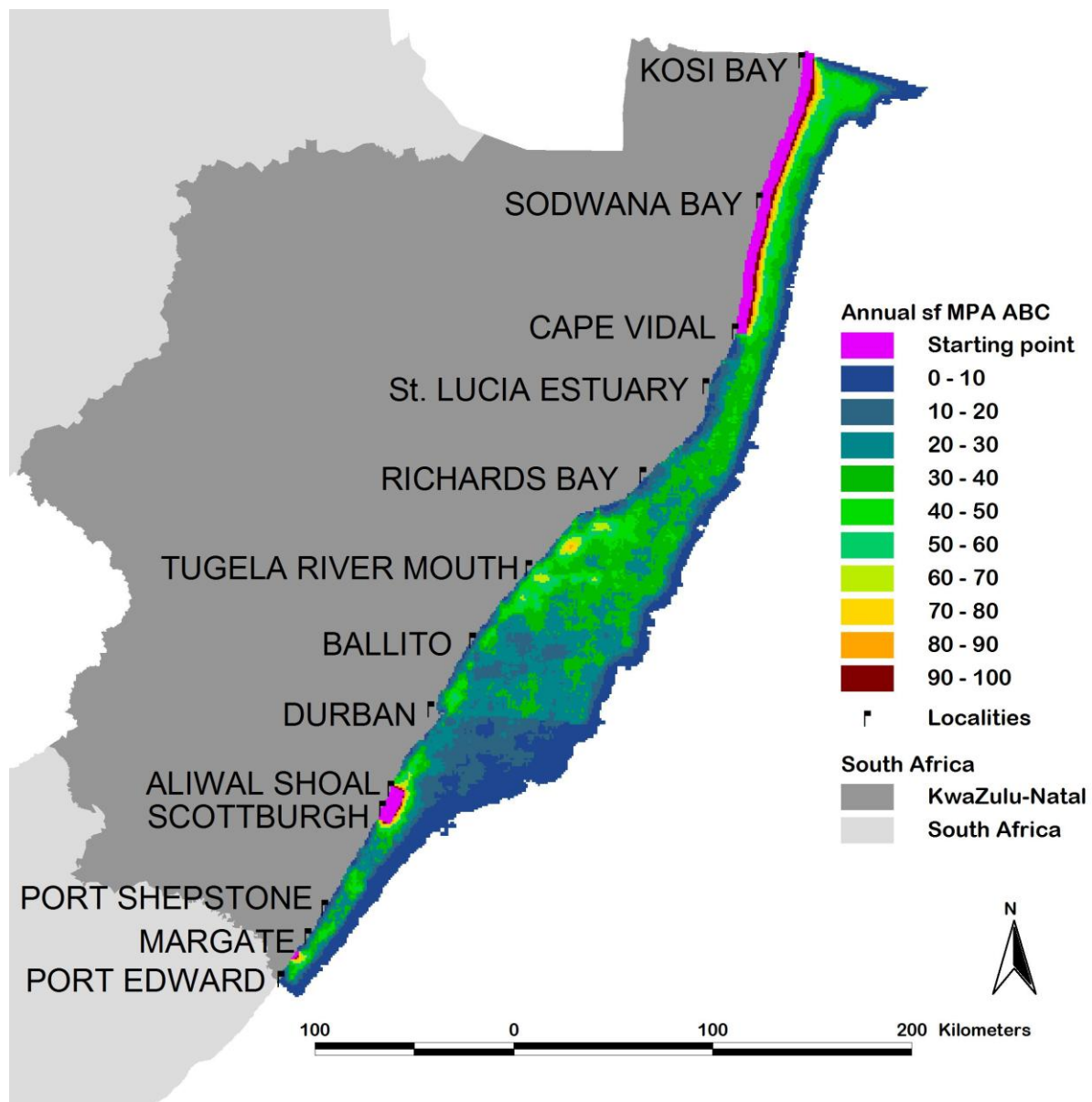


Figure 12. Annual selection frequency (sf) for scenario (iii) where MPA zones A, B and C contribute to targets.

### Differences in seasonal selection frequencies

The annual selection frequency, in the all three scenarios (i – iii), were more similar to that of summer (Appendix 7 – Figures 6 – 8) than during winter (Appendix 7 – Figures 9 – 11). The seasonal difference in selection frequencies between summer and winter data (with MPA

zones A, B and Cs as starting points) indicates that planning units in southern KZN were selected more times than planning units in northern KZN (Figure 13). This pattern follows a similar pattern to the difference in seasonal species richness distribution (Chapter Three, Figure 4). The area from the Tugela River mouth southwards had higher selection frequencies in winter than in summer. The Tugela River mouth, Durban offshore reefs, between Durban and Aliwal Shoal, and Port Shepstone were selected 18 – 42 times more during winter than summer. The inshore area from Kosi Bay to Sodwana Bay had slightly higher selection frequencies during summer, while the areas off St Lucia and south of Richards Bay were selected 6 – 30 times more during summer than winter.

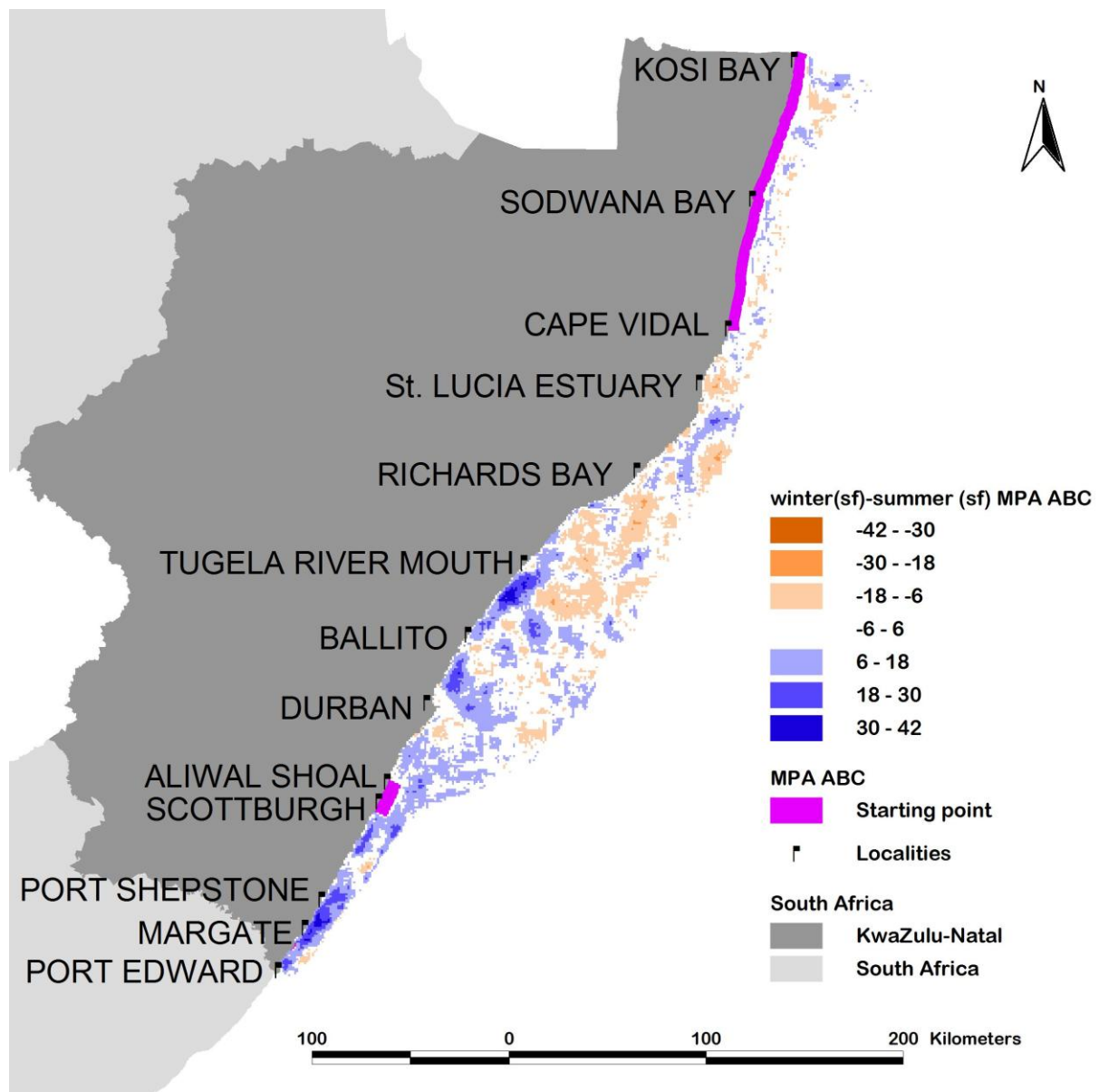


Figure 13. Seasonal difference of selection frequencies values for scenario (iii) where MPA zones A, B and C contribute to targets. Orange areas are selected more frequently in summer, whereas blue areas are selected more frequently in winter.

## SeaPLAN

The Marxan selection frequency map of areas required to meet SeaPLAN targets included offshore areas beyond the 1000 m depth contour (Figure 14), which were not included in the present study (FishPLAN). The SeaPLAN selection frequency output was highly clustered and showed clear potential areas required to meet conservation targets (this output is for targets for strict reservation (i.e. zone A type protection) only, see Lagabrielle et al. 2010). Large inshore areas with the highest selection frequency values were predominantly in the

north of KZN, and between Richards Bay and the Tugela river mouth. A few smaller areas along the south coast also had high selection frequencies, including Aliwal Shoal and the area between Margate and Port Edward.

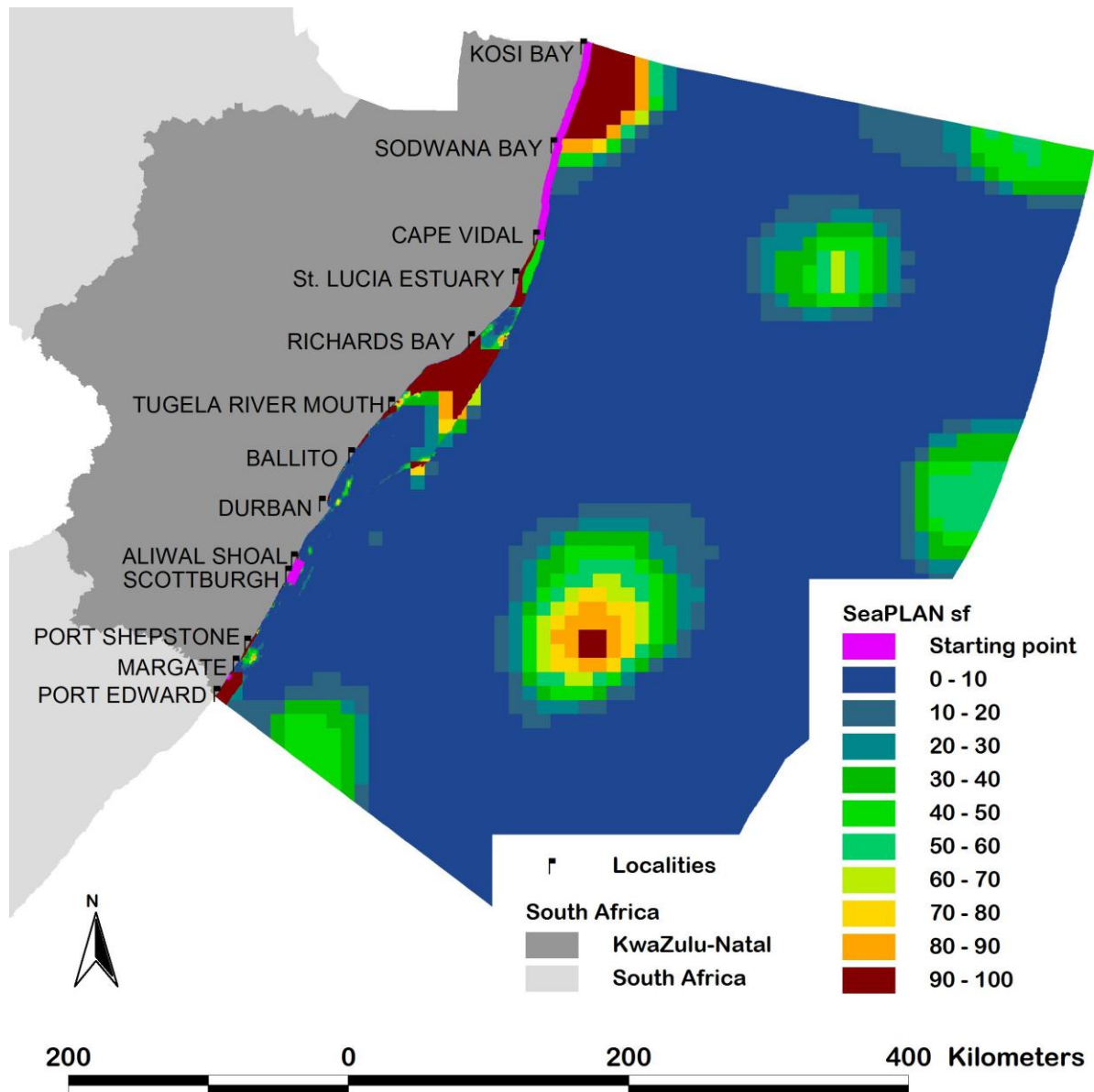


Figure 14. SeaPLAN selection frequency map of areas selected to meet targets (Lagabrielle et al. 2010).

The difference between the selection frequency of SeaPLAN, and that of FishPLAN, is shown in (Figure 15) the FishPLAN analysis was run on the annual data set, with MPA zones A, B and C as starting points).



There were areas of similarities and differences between the two outputs. Both plans had high values for the Aliwal Shoal area and around existing MPAs in northern KZN, especially between Sodwana Bay and Kosi Bay. Directly east from Richards Bay, and offshore from Ballito also had similar selection frequency values. FishPLAN did not consider the offshore environment. The large offshore orange and white areas therefore only show SeaPLAN's selection frequency. The output from FishPLAN, however, did not produce clear clusters of high selection frequencies like SeaPLAN did, i.e. there were more spatial options to meet unmet conservation targets in FishPLAN. The area just offshore from Kosi bay had higher values than that of FishPLAN, while the St. Lucia sanctuary zone (MPA zone A, between Sodwana and Cape Vidal) had higher values in FishPLAN. SeaPLAN had higher values for the large area south of Richards Bay, and Margate to Port Edward. FishPLAN had higher values from Durban to the Tugela river mouth, especially around the offshore Tugela Banks area.

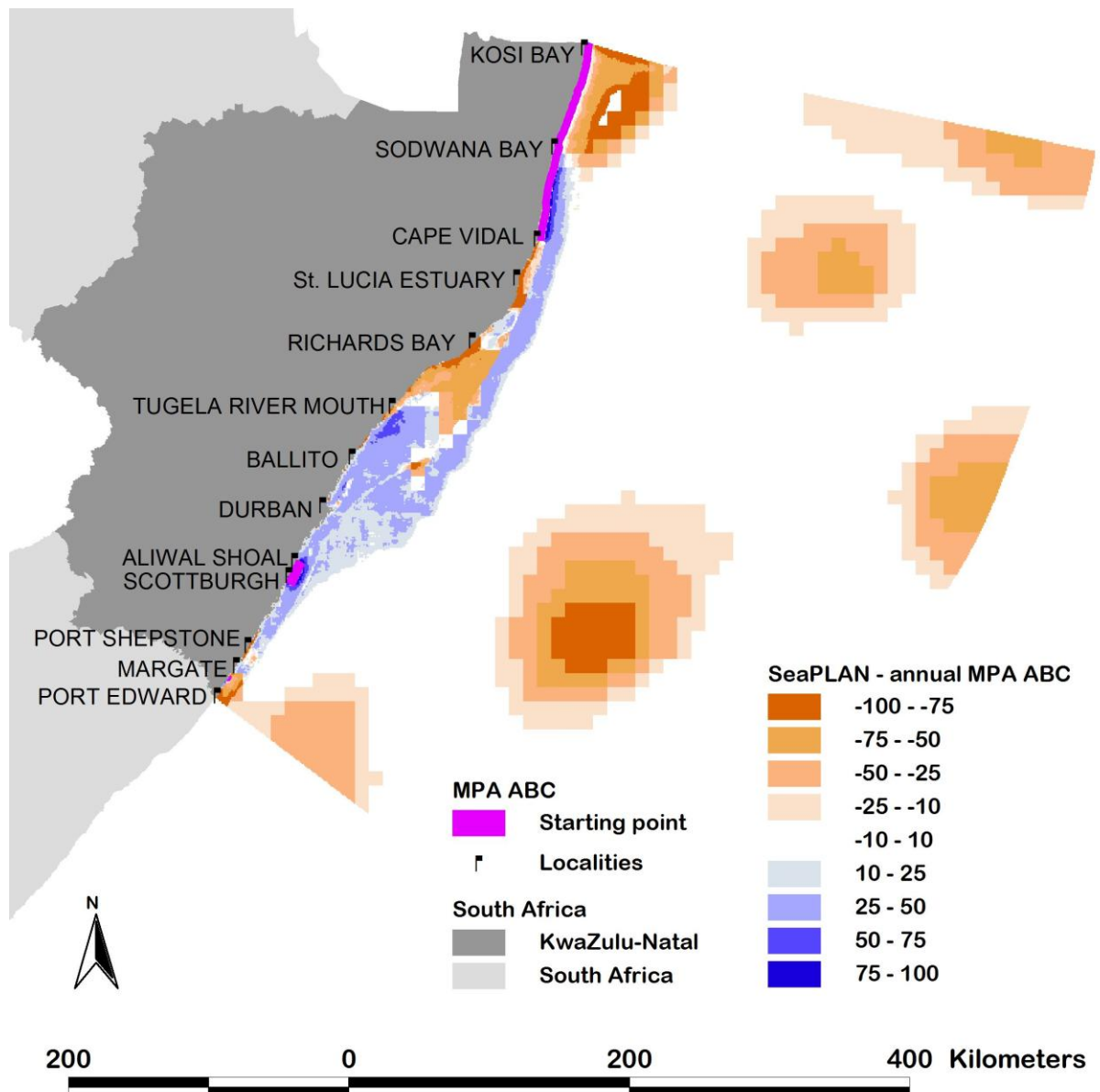


Figure 15. The difference between in the selection frequency of SeaPLAN and FishPLAN. Orange areas are selected more frequently by SeaPLAN, whereas blue areas are selected more frequently by FishPLAN.

## Discussion

### Seasonal conservation targets

Conservation targets were adjusted based on seasonal abundance estimates to provide adequate protection to species when present in highest abundance. The combination of seasonally varying distribution ranges (Chapter 3) and seasonally adjusted targets resulted in different conservation assessment outcomes under the current MPA network for the two seasons. Seasonal target achievement by current MPAs (zones A only) was higher during the season when the species was present in lower abundance (low season), as a result of the lower targets set, and as a result of changes in distribution ranges. For example, 21% of the target set for shad (*P. saltatrix*) in summer was achieved by current MPA zones A, but only 6% of the winter targets were achieved by these MPAs. In this instance, not only did seasonal distribution models (SDMs) differ from one another (Chapter Three), but the targets were different: 11% of range during summer, and 56% of range during winter. However, different seasonal targets were the only reason for differences in target achievement for species with no differences in seasonal ranges. For example, the same SDM was used for the great hammerhead shark (*Sphyrna mokarran*) during summer and winter, but targets were higher in summer (28% of range) than winter (14% of range), because this species is more abundant in KZN waters in summer. The assessment showed that only 35% of its target was achieved by current MPA zones A during summer, whereas 66% of the winter target was met. A greater MPA area is thus required to meet the unmet targets of both seasons, but particularly the summer season.

Conservation targets can therefore be adjusted to match the timing when mobile species are present in high abundance. Targets can therefore also be adjusted to match the timing of specific events, like aggregative spawning behaviour in fish species, e.g. red steenbras (*P. rupestris*) (van der Elst 1989). The use of temporal targets may be especially useful to vulnerable processes that occur throughout the year, but are in highest abundance during a specific time of the year, e.g. green turtle (*Chelonia mydas*) nesting (Gibson 1979).

### Conservation status assessment

The high number of features for which targets were unmet in all of the nine scenarios indicates that existing MPAs require considerable extension if they are to contribute to the conservation of the marine fish species selected for this study. Several features that had low target achievement were species with predominantly southerly distribution ranges, while species with predominantly northerly distribution ranges had higher target achievement. This

is hardly surprising because the majority of existing MPAs are located in northern KZN (Figure 1). Several of the species with predominantly southerly distribution ranges were winter endemic migrants, e.g. red steenbras (*P. rupestris*), geelbek (*Atractoscion aequidens*), and seventy four (*Polysteganus undulosus*), for which conservation concern has already been expressed (Mann 2000; Lombard et al. 2004; Lamberth & Joubert unpublished data). Target achievement was increased in scenarios which included the two southerly MPAs, Trafalgar and Aliwal Shoal, as starting points, indicating the important role that these MPAs play, and the need for their zonation to be changed from zones B and C to A, as well as the need for more MPA zones A in the southern extent of KZN.

The species selected for this study included several species that are in highest abundance during winter, and therefore target achievement by existing MPAs was lower in winter than in summer, or annual scenarios. The lower target achievement during winter indicates the greater need for MPAs that can meet targets for features with high winter abundance. Increased protection is required, especially to meet targets for winter migrants that occupy the south of KZN.

The current network of MPAs had better target achievement for summer scenarios, and poorer for winter scenarios, compared with the annual scenario. The efficiency with which conservation targets can be achieved for moving components of biodiversity (e.g. migratory fish) can therefore be better achieved with moving spatio-temporal MPA networks (Hobday & Hartmann 2006; Grantham et al. 2008; Game et al. 2009).

## **MPA zonation**

The current MPA zonation allows different activities inside zone boundaries. Catch and release of pelagic or game fish is allowed in MPA zones B, while the catch of demersal fish is prohibited (MLRA 1998). Although MPA zones B offer protection to demersal fish, the high mortality rate of caught and released fish limits the value of these zones to the protection of pelagic and game fish species (B. Mann pers. comm.). Fishing is allowed in MPA zones C, and these therefore do not contribute to fish conservation targets. If the relative contributions of each of the existing zones A, B and C to targets of all species had been taken into account (as can be done with Marzone software), the achievement of targets may have improved for some species, but this task was outside the time constraints of this study.

## Conservation plan

As expected, areas surrounding existing MPAs had high selection frequencies in the conservation planning analyses (because the software tries to minimise MPA boundary lengths and final MPA sizes). The areas with high selection frequencies outside of or far away from existing MPAs indicate that biodiversity in these areas is not currently represented in existing MPAs. The number of planning units required by the best solution (i.e. the planning units that were required to meet all targets while minimizing cost) comprised ca 25% of the planning area.

The similarity of areas that had high selection values in all nine scenarios indicates the need for increased protection around the Tugela River mouth, Aliwal Shoal and to a lesser extent, the Durban offshore reefs.

The Tugela (aka Thukela) River mouth has a unique sediment and oceanographic system in KZN, which is shaped by fluvial and oceanographic processes (Bosman et al. 2007). The increase of selection frequency of the area around the Tugela River mouth with the increase in MPA size under the three scenarios (i - iii) to (the far away) existing reserve network indicates its importance to meet targets (Figures 10 – 12), i.e. this area has a high irreplaceability value to conservation. The area is important for sciaenid fish species, especially as a spawning area for the square tail kob (*A. thorpei*) (Fennessy 1994a), and perhaps a stopover point for great white shark (*Carcharodon carcharias*) migrations (Bonfil et al. 2005). The area also supports a prawn trawling industry which takes high bycatch of several linefish elasmobranch species (Fennessy 1994a;b; Mkize 2006).

Aliwal shoal is an algal dominated subtropical reef, with structural similarities to the coral reefs north of Cape Vidal, but situated ca 350 km south of these reefs (Schleyer 2008; Olbers et al. 2009). Several migratory species use Aliwal Shoal for reproductive behaviour, e.g. spotted raggedtooth shark (*C. taurus*) (Van Tienhoven et al. 2007). Aliwal Shoal is a well known and heavily used dive site (Olbers et al. 2009), that is currently afforded minimum protection (Zones B and C, Gazette 2004). An increase in the protection regulations will thus require an intense stakeholder process.

Durban Bay has historically been likened with St Lucia in Northern KZN as a nursery area for several fish species, and is still the main spawning ground for riverbream (*Acanthopagrus vagus*) (Connell 2007). The offshore reefs support a diversity of permanent resident and migratory fish from the Cape and the tropics (NMLS unpublished data 2010). The Durban offshore reefs are subject to heavy fishing pressure, are adjacent to an international port, lie

in heavily used shipping lanes, and also support a crustacean trawling fishery, amongst many other pressures in the area (Figure 2).

Although the Durban area had a higher selection frequency than its immediate surrounding areas, the irreplaceability was lower than other areas further away, e.g. the aforementioned area offshore from Tugela River. The decrease in selection frequency of the area surrounding Durban, after MPA zones B and C were added to the existing reserve network, indicates that the features that occur here also occur elsewhere, i.e. the targets that can be met in this area may also be achieved by increasing MPA estates elsewhere in KZN. It is likely that the increased MPA estates around Aliwal Shoal and Trafalgar were responsible for decreasing the selection frequency around Durban as these areas shared several overlapping features. The higher cost of conservation in areas such as Durban may require identifying alternative areas that are able to meet the unmet targets. The rezoning of Aliwal Shoal to MPA A status is not necessarily the only option to meet unmet targets in the Durban area, but appeared to be highly efficient at doing so. Alternative areas that can meet conservation targets for features that occur in Durban need further investigation.

### **Seasonal differences in spatial solutions**

The difference in areas selected for seasonal reserve networks conform to the seasonal distribution of features (Chapter Three, Figure four). There were higher selection frequencies in the south of KZN than northern KZN during winter. Seasonal protection in the south of KZN during winter would thus benefit winter migrants. The species that are present in highest abundance during summer and occurring in the north of KZN would, however, benefit from increased summer protection in northern KZN, e.g. the great hammerhead shark (*S. mokarran*) and slinger (*C. puniceus*).

The seasonal variation of human use intensity of the marine and nearshore environment was not considered in this study. Several fishing operations have strict time periods over which they operate. For example, prawn trawling over the Tugela banks is closed in January and February (Fennessy 1994a), and October - November is closed for shad (*P. saltatrix*). Recreational fishing pressure is also generally considered to be lower during winter months (Brouwer et al. 1997; Mann et al. 1997). Changes in human uses have an influence on the opportunity cost of protecting an area, and this is a useful area for future research in the application of moving spatio-temporal MPAs. However, long-term accumulated benefits of permanent reserves to resident fish species and benthic communities may be lost if protection is seasonally removed from areas where these species occur. The long life cycles of resident fish would leave them vulnerable to fishing pressures, and benthic communities

may be destroyed if protection is seasonally removed. It may therefore be necessary to include both permanent and mobile protected areas as part of a reserve network.

Hobday and Hartmann (2006) have noted that practical considerations regarding changes in boundaries of protected areas need due consideration, for example, the communication to fishing fleets and management. The use of mobile spatio-temporal MPAs along the highly developed KZN coastline has more stakeholders than the pelagic or offshore environment, and the disruptions caused by sudden closures of areas to fishing and other activities is unlikely to be tolerated by the South African fishing community (Brouwer et al. 1997). A potential solution may be the use of seasonal alteration to zoning of new and existing MPAs (according to seasonal requirements).

### **Differences between SeaPLAN and FishPLAN**

A conservation plan that is based on a subset of the biodiversity features in an area of interest may identify some areas that are useful or even critical for overall biodiversity conservation. In FishPLAN there were several areas with high selection frequencies that corresponded with areas identified by the more complete SeaPLAN, for example, both plans agreed that the existing MPAs need to be extended offshore. This indicates that the subset of species may be a good surrogate for the data sets used in SeaPLAN, or that the rarity of some of the FishPLAN features drove selection frequencies in SeaPLAN (which also used many of these features). SeaPLAN considered 423 features, including habitats, ecological processes and species (Lagabrielle et al. 2010), of which 117 were fish SDMs. Forty-seven of the SDMs used in FishPLAN were also used in SeaPLAN and may have resulted in some similarities between the two plans. The differences between the two selection frequency maps can be attributed to the additional features used in SeaPLAN (e.g. whales, dolphins, turtles, shoreline habitats, pelagic habitats, and offshore ecological processes, etc.), whereas FishPLAN included only marine fish. The differences may also arise from the areas used as starting points of conservation planning analyses and the different parameters used in Marxan. In the SeaPLAN analysis presented here all existing MPA zones were used as a starting points, as well as Admiralty zones (state land along the coastline) (NEMA 1998).

The shortcomings of using focal groups of species as surrogates to represent biodiversity adequately have been shown for Indo Pacific coral reef species (Beger et al. 2007). Yet the use of surrogates is necessary because not all biodiversity is mapped (Faith et al. 2004; Pressey 2004). It then becomes a question of how much data, of what feature, is enough to adequately represent biodiversity in conservation plan. The simple answer appears to be: the more the better (Pressey 2004), as no land class (Lombard et al. 2003) or single species

(Beger et al. 2007) can act as a surrogate. Although the use of focal groups are useful to study aspects of conservation (Zacharias & Roff 2001), the use of the most complete and representative data set of biodiversity surrogates is recommended for conservation plans.

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# General conclusions

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This study of marine fish species formed part of a larger provincial marine systematic conservation plan for KwaZulu-Natal (KZN), SeaPLAN (Lagabriele et al. 2010). However, owing to budget and time constraints, not all ± 1430 fish species that occur in the region could be considered. A species prioritisation method was thus developed to identify the species that were most at risk of being missed by a conservation plan that is based mostly on habitat representation (i.e. SeaPLAN). This prioritisation method defined three underlying principles, on which seven criteria were based, in order to identify appropriate species. Each criterion had qualifying conditions. Using this method, 67 species were identified for inclusion into the study (which I called FishPLAN). Data available for fish species were evaluated for spatio-temporal accuracy, and distribution range models were developed to include greater detail and accuracy. Some novel ideas were tested: seasonal species distribution ranges were modelled; conservation targets were formulated to take seasonal abundance into account; and an assessment and a conservation plan were undertaken to evaluate the incorporation of seasonal data into planning.

The 67 species identified for FishPLAN represented a wide range of families (32), and several endemic and/or threatened sea breams (Sparidae, 15 of the 67 species). The difference between the species prioritisation method used in this study and that of fisheries-directed studies (e.g. Lamberth & Joubert unpublished data) was the inclusion of less conspicuous species that often lack fisheries data, e.g. the Gobiidae (gobies, 6 of the 67 species). Thirteen species were identified from local lists of conservation concern that did not appear on similar international lists (e.g. CITES 2009; IUCN 2009). These international lists are incomplete and species assessments are biased towards charismatic megafauna and species of special research interest (Possingham et al. 2002). The value of using local published works and lists of species of conservation concern (e.g. Mann 2000; Lombard et al. 2004; TOPs list 2007; Lamberth & Joubert unpublished data) should not be underestimated. Information on species dependent on specialised habitats was very limited, and allowed us to identify only four species using this criterion. There is a need for more information on habitat specialisation of species if these species are to be included in conservation plans. Criteria not investigated, like biodiversity distinctiveness (Vane-Wright et al. 1991), may have resulted in different species lists. The inclusion of ever more criteria, however, has the potential to delay conservation actions (Cowling et al. 2010).

The long-term marine fish data collection programmes in KZN that collect point locality data were assessed for spatiotemporal accuracy. Data were available for 39 of the 67 species identified in Chapter One. Data collection methods that do not currently meet the biodiversity data collection standards (TDWG 2009) restrict the use of the data to little beyond the goals of the data collection programmes. A low proportion (18%) of the data had a spatial uncertainty of one km or less and was suitable for fine-scale distribution modelling. However, a high level of temporal accuracy (>99% recorded to day-level) enabled grouping of data into seasonal divisions, and potential for fine-scale temporal application of the data.

Logistical difficulties of data being submitted and collected from recreational anglers and other large-scale data collection programmes may limit the scope for immediate improvements in these programmes (for example, the NMLS recreational programme). The use of handheld data recording and GPS devices is recommended for collecting data to increase the spatial accuracy and efficiency of data collection and transfer of data to a database. It is also recommended that data be collected from SCUBA divers where much of the northern KZN has fishing regulations that prohibit fishing, but allow diving. This would add data for several species for which there currently are no data, as well as provide a more representative spatial sample of data.

Species life cycle envelopes (SLICES) were developed to capture spatial differences in areas occupied during different life-cycle phases. This spatial distinction allows for increased conservation effort in areas and at times when vulnerable life stages occur, e.g. aggregative spawning behaviour in red steenbras (*Petrus rupestris*). Data for juvenile life cycle phases were severely limited, thereby limiting the use of SLICES for reproductive behaviour. Data for many marine fish and mammal species are sparse and many of the species are poorly understood (Kaschner et al. 2006).

Data limitations, like the short comings of point locality data, and incomplete knowledge of species' ranges, required an adaptive approach to distribution modelling that could use both quantitative and qualitative data. Two distribution modelling techniques were used: Maxent, which uses point locality, and CHARMS, which use descriptive range data. A combination of statistical and biological criteria was used to determine the most informative and correct model for each species. Maxent models provided more detailed information, but did not always meet statistical conditions, or did not always fit known ranges from literature descriptions.

Poor model performance was likely to be the result of under-sampling in areas that coincide with changes in oceanography, e.g. 10 km north and south of the Amatikulu River, and in

marine protected areas (MPAs) where fishing is prohibit in northern KZN. Again, SCUBA-diver data from northern KZN would supplement the sampling distribution along the KZN coastline.

The patterns of species richness developed from the seasonal models showed seasonal differences in richness patterns that conformed to known seasonal distributions of fish assemblages (van der Elst 1988). The southern parts of KZN had higher richness during winter, while northern parts had higher summer richness. Winter migrants that enter KZN from the South to breed and extend their feeding ranges are limited in their northern extent by strong ocean currents and water temperatures that form dispersal barriers. The southward dispersal of summer migrants is likely to be limited by low water temperatures. The importance of temporal considerations may therefore be useful to increase conservation planning efficiency, and was explored in Chapter Four.

Conservation targets were set using a baseline target of 20 %, as a minimum, based on the Millennium Development Goals (WSSD 2002). The general applicability of a single target for all biodiversity has yet to be scientifically proven (Agardy et al. 2003). Biological retention targets were added to baseline targets as a precautionary approach. Targets were further adjusted depending on seasonal abundance of a given species. This resulted in different seasonal target achievement under the current MPA configuration: seasonal target achievement was higher when species were present in lowest abundance and lower during high abundance.

Within the current network of MPAs, none of the species' targets were met by MPA zone As alone, and all species required increased protection. As expected, the scenarios that included MPA zones B and C (by theoretically rezoning them to A status) resulted in higher target achievement. The current protection offered by zones B and C is significantly poorer than zone A status (i.e. sanctuary) and is no substitute for non-extractive use (Hawkins et al. 1999; Denny & Babcock 2004; Stefansson & Rosenberg 2005). The zonation of MPAs in South Africa has not been applied consistently (Attwood et al. 1997) and is currently under review, with the aim to standardise it nationally. Aspects being considered are the number of zones and reformulating the regulations controlling activities allowed within them (Jackelman et al. 2007). It is hoped that new zonation will not further detract from the already inadequate protected area network.

The usefulness of mobile spatio-temporal MPAs has been shown for pelagic oceans (Hobday & Hartmann 2006; Grantham 2008; Game et al. 2009). The greater efficiency of a seasonal MPA network to protect seasonally varying distributions of biodiversity suggests

that it may be a useful tool to include in conservation management. The logistic and management constraints of mobile protected areas may limit their implementation at present, especially in coastal areas of high human use.

The difference in the spatial solutions between the FishPLAN and SeaPLAN indicate the importance of including a broad representation of biodiversity rather than a single target group of species. Single species conservation plans run the risk of identifying areas that are appropriate only for these species, and not to biodiversity as a whole (Lombard et al. 2003; Pressey 2004).

It is hoped that the methods used in this study to solve particular problems (e.g. species selection, data quality evaluation, model choice), and the approaches used to plan efficient spatio-temporal conservation (e.g. seasonal targets and distributions) will be of use in the development of conservation planning outside of the KZN province.

Knight et al. (2006) used an operational model to evaluate the effectiveness of conservation plans. Their study suggested several mechanisms to increase the effectiveness of conservation plans. The involvement of public stakeholders was identified as an important aspect to empower individuals and increase public stakeholder buy-in to conservation plans. Experts were consulted to help refine the distribution models during this study and added valuable buy-in from some of the toughest critics. The value of a conservation plan is greatly enhanced if its products are mainstreamed and the result of its implementation is reviewed and refined over time (Knight et al. 2006). The results from this study have built on, and refined the data that was available to SeaPLAN at that time. The species list, species database and the updated species distribution models have been made available to the Provincial conservation agency, Ezemvelo KZN Wildlife, for future use in conservation plans and management decisions. The products were also made available to the South African National biodiversity Institute (SANBI), including the species information database developed during this study. It is hoped that by involving public stakeholders and mainstreaming the products of this study, we will have partially filled the gap between research and implementation in marine conservation planning for KZN.

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Appendix 1: The criteria that KZN marine fish species qualified for, and the final list number for which each species qualified (see “List”).

<i>Family name</i>	<i>Scientific name</i>	<i>Common name</i>	<i>Endemic species</i>	<i>Species of conservation concern</i>	<i>Species with life history vulnerability</i>	<i>Highly resident species</i>	<i>Rare species</i>	<i>Species dependent on estuaries</i>	<i>Species dependent on specialised habitats</i>	<i>List</i>
Acanthuridae	<i>Acanthurus dussumieri</i>	pencilled surgeon	0	0	1	0	0	0	0	2
Acanthuridae	<i>Naso thorpei</i>	nohorn unicorn	1	0	0	0	0	0	0	1
Acanthuridae	<i>Zebrasoma gemmatum</i>	spotted tang	0	0	0	0	1	0	0	2
Acropomatidae	<i>Neoscombrops cynodon</i>	silver splitfin	1	0	0	0	0	0	0	1
Ambassidae	<i>Ambassis gynocephalus</i>	bald glassy	0	0	0	0	0	1	0	2
Ambassidae	<i>Ambassis natalensis</i>	smooth glassy	0	0	0	0	0	1	0	2
Ambassidae	<i>Ambassis productus</i>	longspine glassy	0	0	0	0	0	1	0	2
Aploactinidae	<i>Cocotropus monacanthus</i>	roughskin scorpionfish	0	0	0	0	1	0	0	1
Aploactinidae	<i>Ptarmus jubatus</i>	crested scorpionfish	0	0	0	0	1	0	0	2
Apogonidae	<i>Apogon nitidus</i>	bluespot cardinal	1	0	0	0	1	0	0	1
Ariidae	<i>Galichthys feliceps</i>	white seacatfish	0	0	0	1	0	1	0	2
Atherinidae	<i>Atherina breviceps</i>	Cape silverside	0	0	0	0	0	1	0	2
Atherinidae	<i>Atherinomorus lacunosus</i>	hardyhead silverside	0	0	0	0	0	1	0	2
Bembridae	<i>Parabembris robinsoni</i>	african deepwater flathead	1	0	0	0	0	0	0	2
Berycidae	<i>Centroberyx spinosus</i>	short alfonsino	1	0	0	0	0	0	0	2

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<i>Family name</i>	<i>Scientific name</i>	<i>Common name</i>	<i>Endemic species</i>	<i>Species of conservation concern</i>	<i>Species with life history vulnerability</i>	<i>Highly resident species</i>	<i>Rare species</i>	<i>Species dependent on estuaries</i>	<i>Species dependent on specialised habitats</i>	<i>List</i>
Blenniidae	<i>Alloblennius parvus</i>	dwarf blenny	0	0	0	0	1	0	0	1
Blenniidae	<i>Mimoblennius rusi</i>	rusi blenny	1	0	0	0	0	0	0	2
Blenniidae	<i>Omobranchus banditus</i>	bandit blenny	1	0	0	0	0	0	0	2
Blenniidae	<i>Omobranchus woodi</i>	kappie blennie	0	0	0	0	0	1	0	2
Bothidae	<i>Engyprosopon natalensis</i>	Natal flounder	1	0	0	0	0	0	0	1
Bothidae	<i>Laeops natalensis</i>	khaki flounder	1	0	0	0	0	0	0	1
Bothidae	<i>Pseudorhombus natalensis</i>	smalltooth flounder	1	0	0	0	0	0	0	2
Bythitidae	<i>Diplacanthopoma nigripinnis</i>	(bythitid - nil)	1	0	0	0	1	0	0	1
Callionymidae	<i>Draculo celetus</i>	dainty dragonet	0	0	0	0	1	0	0	2
Callionymidae	<i>Synchirops monacanthus</i>	deepwater dragonet	0	0	0	0	1	0	0	2
Carangidae	<i>Caranx ignobilis</i>	giant kingfish	0	0	0	1	0	1	0	2
Carangidae	<i>Caranx sexfasciatus</i>	bigeye kingfish	0	0	0	0	0	1	0	2
Carangidae	<i>Caranx papuensis</i>	brassy kingfish	0	0	0	1	0	0	0	2
Carangidae	<i>Caranx sem</i>	blacktip kingfish	0	0	0	1	0	0	0	2
Carangidae	<i>Lichia amia</i>	garrick	0	0	1	0	0	1	0	2



Appendix 1: The criteria that KZN marine fish species qualified for, and the final list number for which each species qualified (see “List”).

Family name	Scientific name	Common name	Endemic species	Species of conservation concern	Species with life history vulnerability	Highly resident species	Rare species	Species dependent on estuaries	Species dependent on specialised habitats	List
Carangidae	<i>Scomberoides lysan</i>	doublespotted queenfish	0	0	0	0	0	1	0	2
Carangidae	<i>Seriola lalandi</i>	giant yellowtail	0	0	0	0	1	0	0	2
Carangidae	<i>Trachinotus africanus</i>	african pompano	0	0	1	0	0	0	0	2
Carangidae	<i>Trachinotus botla</i>	largespotted pompano	0	0	0	1	0	0	0	2
Carcharhinidae	<i>Carcharhinus brachyurus</i>	copper shark	0	0	1	0	0	0	0	2
Carcharhinidae	<i>Carcharhinus leucas</i>	bull shark	0	1	1	0	0	0	0	3
Carcharhinidae	<i>Carcharhinus limbatus</i>	blacktip shark	0	1	0	0	0	0	0	2
Carcharhinidae	<i>Carcharhinus longimanus</i>	oceanic whitetip shark	0	1	0	0	0	0	0	2
Carcharhinidae	<i>Carcharhinus sealei</i>	blackspot shark	0	0	1	0	0	0	0	2
Carcharhinidae	<i>Carcharhinus obscurus</i>	dusky shark	0	0	1	0	0	0	0	2
Carcharhinidae	<i>Hemipristis elongatus</i>	snaggletooth fossil shark	0	1	0	0	0	0	0	1
Carcharhinidae	<i>Paragaleus leucomatus</i>	whitetip weasel shark	1	0	0	0	1	0	0	1
Carcharhinidae	<i>Scylliogaleus queckettii</i>	flapnose houndshark	1	1	1	0	0	0	0	3
Centracanthidae	<i>Spicara australis</i>	picarel	1	0	0	0	1	0	0	3
Centracanthidae	<i>Spicara axillaris</i>	windtoy	1	0	0	0	1	0	0	3

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Family name	Scientific name	Common name	Endemic species	Species of conservation concern	Species with life history vulnerability	Highly resident species	Rare species	Species dependent on estuaries	Species dependent on specialised habitats	List
Cepolidae	<i>Owstonia simoterus</i>	bandfish	1	0	0	0	1	0	0	1
Cetomimidae	<i>Cetomimus indagator</i>	whalefish	1	0	0	0	1	0	0	1
Chaetodontidae	<i>Chaetodon mitratus</i>	oblique banded butterflyfish	0	0	0	0	1	0	0	2
Chaetodontidae	<i>Chaetodon bennetti</i>	archer butterflyfish	0	0	0	0	1	0	0	3
Chaetodontidae	<i>Chaetodon marleyi</i>	doublesash butterflyfish	1	1	0	0	0	0	0	2
Cheilodactylidae	<i>Chirodactylus jessicalenorum</i>	Natal fingerfin	1	0	1	0	0	0	0	3
Chimaeridae	<i>Hydrolagus africanus</i>	african chimaera	0	0	0	0	1	0	0	2
Chlamydoselachidae	<i>Centrophorus niaukang</i>	Taiwan gulper shark	0	0	1	0	1	0	0	2
Clinidae	<i>Clinus spatulatus</i>	bot river klipfish	0	0	0	0	0	1	0	2
Clinidae	<i>Clinus superciliosus</i>	super klipfish	0	0	0	0	0	1	0	2
Clinidae	<i>Pavoclinus laurenti</i>	rippled klipfish	1	0	0	0	0	0	0	2
Clinidae	<i>Pavoclinus mentalis</i>	bearded klipfish	1	0	0	0	1	0	0	3
Clupeidae	<i>Gilchristella aestuaria</i>	estuarine roundherring	0	0	0	0	0	1	0	2
Clupeidae	<i>Sardinops sagax</i>	pilchard	0	0	1	0	0	0	0	2
Congrogadidae	<i>Halimuraena shakai</i>	zulu snakelet	0	0	0	0	1	0	0	2

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Congrogadidae	<i>Natalichthys leptus</i>	pencil snakelet	1	0	0	0	1	0	0	1
Congrogadidae	<i>Natalichthys ori</i>	Natal snakelet	1	0	0	0	1	0	0	1
Congrogadidae	<i>Natalichthys sam</i>	nail snakelet	1	0	0	0	1	0	0	1
Creedidae	<i>Apodocreeidia vanderhorsti</i>	longfin burrower	1	0	0	0	0	0	0	2
Cynoglossidae	<i>Cynoglossus attenuatus</i>	fourline tonguefish	1	0	0	0	0	0	0	2
Cynoglossidae	<i>Cynoglossus marleyi</i>	threeline tonguefish	1	0	0	0	0	0	0	2
Cynoglossidae	<i>Symphurus ocellatus</i>	tonguefish	1	0	0	0	0	0	0	2
Dasyatidae	<i>Himantura draco</i>	dragon stingray	1	0	0	0	1	0	0	1
Dasyatidae	<i>Taeniura meyeni</i>	round ribbontail ray	0	1	0	0	0	0	0	2
Dasyatidae	<i>Urogymnus asperrimus</i>	porcupine ray	0	1	0	0	1	0	0	3
Dichistiidae	<i>Dichistius capensis</i>	galjoen	0	1	1	0	0	0	0	3
Dichistiidae	<i>Dichistius multifasciatus</i>	banded galjoen	1	0	0	0	0	0	0	2
Dinoperidae	<i>Dinoperca petersi</i>	cave bass	0	0	1	1	0	0	0	2
Echeheidae	<i>Phtheichthys lineatus</i>	slender remora	0	0	0	0	1	0	0	2
Eleotridae	<i>Butus butis</i>	duckbill sleeper	0	0	0	0	0	1	0	2

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Eleotridae	<i>Eleotris fusca</i>	dusky sleeper	0	0	0	0	0	1	0	2
Eleotridae	<i>Eleotris mauritanus</i>	widehead sleeper	0	0	0	0	1	1	0	2
Eleotridae	<i>Eleotris melanosoma</i>	broadhead sleeper	0	0	0	0	1	1	0	2
Eleotridae	<i>Hypseleotris dayi</i>	golden sleeper	1	0	0	0	0	0	0	2
Elopidae	<i>Elops machnata</i>	springer	0	0	0	0	0	1	0	2
Engraulidae	<i>Thryssa vitrirostris</i>	orangemouth glassnose	0	0	0	0	0	1	0	2
Gerreidae	<i>Gerres macracanthus</i>	longspine pursemouth	0	0	0	0	0	1	0	2
Gerreidae	<i>Gerres filamentosus</i>	threadfin pursemouth	0	0	0	0	0	1	0	2
Gerreidae	<i>Gerres methueni</i>	evenfin pursemouth	1	0	0	0	0	1	0	3
Gobiesocidae	<i>Lissonanchoa lusherii</i>	streaky clingfish	1	0	0	0	0	0	0	1
Gobiesocidae	<i>Pherallodus smithi</i>	mini clingfish	1	0	0	0	1	0	0	1
Gobiidae	<i>Caffrogobius caffer</i>	banded goby	1	0	0	0	0	0	0	2
Gobiidae	<i>Caffrogobius gilchristi</i>	prison goby	0	0	0	0	0	1	0	2
Gobiidae	<i>Caffrogobius natalensis</i>	baldy	1	0	0	0	1	1	0	3
Gobiidae	<i>Caffrogobius nudiceps</i>	barehead goby	0	0	0	0	0	1	0	2

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Gobiidae	<i>Croilia mossambica</i>	naked goby	1	0	0	0	0	1	0	3
Gobiidae	<i>Drombus simulus</i>	pinafore goby	1	0	0	0	0	0	0	2
Gobiidae	<i>Favonigobius melanobranchus</i>	blackthroat goby	0	0	0	0	0	1	0	2
Gobiidae	<i>Favonigobius reichei</i>	tropical sandgoby	0	0	0	0	0	1	0	2
Gobiidae	<i>Glossogobius biocellatus</i>	sleepy goby	0	0	0	0	0	1	0	2
Gobiidae	<i>Glossogobius callidus</i>	river goby	0	0	0	0	0	1	0	2
Gobiidae	<i>Gobiopsis pinto</i>	snakehead goby	1	0	0	0	0	0	0	2
Gobiidae	<i>Heteroleotris margaretae</i>	smooth scale goby	1	0	0	0	0	0	0	2
Gobiidae	<i>Monishia william</i>	kaalpens goby	1	0	0	0	0	0	0	2
Gobiidae	<i>Mugilogobius durbanensis</i>	Durban goby	1	0	0	0	0	0	0	2
Gobiidae	<i>Oligolepis acutipennis</i>	sharptail goby	0	0	0	0	0	1	0	2
Gobiidae	<i>Oligolepis keiensis</i>	speartail goby	0	0	0	0	0	1	0	2
Gobiidae	<i>Oxyurichthys lemayi</i>	lace goby	1	0	0	0	0	0	0	2
Gobiidae	<i>Pandaka silvana</i>	dwarf goby	0	0	0	0	0	1	0	2
Gobiidae	<i>Paragobiodon echinocephalus</i>	redhead goby	0	0	0	0	0	0	1	2

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Gobiidae	<i>Paragobiodon modestus</i>	warthead goby	0	0	0	0	0	0	1	2
Gobiidae	<i>Paragobiodon xanthosomus</i>	emerald goby	0	0	0	0	0	0	1	2
Gobiidae	<i>Periophthalmus koelreuteri africanus</i>	african mudhopper	0	0	0	0	0	1	0	2
Gobiidae	<i>Periophthalmus sobrinus</i>	bigfin mudskipper	0	0	0	0	0	1	0	2
Gobiidae	<i>Pleurosicya annandalai</i>	scalenape goby	0	0	0	0	1	0	1	2
Gobiidae	<i>Psammogobius knyaensis</i>	Knysna sandgoby	0	0	0	0	0	1	0	2
Gobiidae	<i>Pteleotris lineopinnis</i>	sad glider	1	0	0	0	0	0	0	1
Gobiidae	<i>Redigobius bikolanus</i>	bigmouth goby	0	0	0	0	0	1	0	2
Gobiidae	<i>Redigobius dewaalii</i>	checked goby	1	0	0	0	0	1	0	3
Gobiidae	<i>Silhouettea sibayi</i>	barebreasted goby	1	0	0	0	0	1	0	3
Gobiidae	<i>Taeniodes esquivel</i>	bulldog eelgoby	1	0	0	0	0	1	0	3
Gobiidae	<i>Taeniodes jacksoni</i>	bearded eelgoby	1	0	0	0	0	1	0	3
Haemulidae	<i>Pomadasys commersonnii</i>	spotted grunter	0	0	0	1	0	1	0	2
Haemulidae	<i>Pomadasys laurentino</i>	manylined grunter	0	0	0	0	1	0	0	2
Haemulidae	<i>Pomadasys multimaculatum</i>	cock grunter	0	0	1	0	0	0	0	2

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Haemulidae	<i>Pomadasys olivaceum</i>	piggy	0	1	0	0	0	0	0	1
Haemulidae	<i>Pomadasys striatum</i>	striped grunter	1	0	0	0	0	0	0	2
Haemulidae	<i>Pomadasys furcatum</i>	grey grunter	0	0	0	1	0	0	0	2
Haemulidae	<i>Pomadasys kaakan</i>	javelin grunter	0	0	0	1	0	0	0	2
Hemiramphidae	<i>Hyporhamphus capensis</i>	Cape halfbeak	1	0	0	0	0	1	0	3
Hoplichthidae	<i>Hoplichthys acanthopleurus</i>	spiny flathead	0	0	0	0	1	0	0	2
Istiophoridae	<i>Xiphias gladius</i>	swordfish	0	1	0	0	0	0	0	2
Labridae	<i>Anchichoerops natalensis</i>	Natal wrasse	1	1	1	0	1	0	0	3
Labridae	<i>Bodianus anthiodes</i>	lyretail hogfish	0	0	0	0	1	0	0	2
Labridae	<i>Cirrhilabrus exquisitus</i>	exquisite wrasse	0	1	0	0	0	0	0	2
Labridae	<i>Macropharyngodon bipartitus</i>	divided wrasse	0	1	0	0	1	0	0	1
Labridae	<i>Thalassoma genivittatum</i>	redcheek wrasse	0	1	0	0	1	0	0	1
Lamnidae	<i>Carcharodon carcharias</i>	great white shark	0	1	1	0	0	0	0	3
Lamnidae	<i>Isurus paucus</i>	longfin mako	0	1	0	0	0	0	0	2
Latimeriidae	<i>Latimeria chalumnae</i>	coelacanth	0	1	0	0	1	0	0	3

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Leiognathidae	<i>Leiognathus equula</i>	common ponyfish	0	0	0	0	0	1	0	2
Lutjanidae	<i>Lutjanus sanguineus</i>	blood snapper	0	0	1	0	0	0	0	2
Lutjanidae	<i>Lutjanus argentimaculatus</i>	river snapper	0	0	0	1	0	0	0	2
Lutjanidae	<i>Lutjanus rivulatus</i>	speckled snapper	0	0	0	1	0	0	0	2
Mobulidae	<i>Manta birostris</i>	manta ray	0	0	1	0	0	0	0	2
Monodactylidae	<i>Monodactylus argenteus</i>	round or natal moony	0	0	0	0	0	1	0	2
Monodactylidae	<i>Monodactylus falciformis</i>	oval moony	0	0	0	0	0	1	0	2
Mugilidae	<i>Crenimugil crenilabis</i>	fringelip mullet	0	0	1	0	0	1	0	2
Mugilidae	<i>Liza alata</i>	diamond mullet	0	0	0	0	0	1	0	2
Mugilidae	<i>Liza dumerilii</i>	groovy mullet	0	0	0	0	0	1	0	2
Mugilidae	<i>Liza luciae Penrith</i>	St. Lucia mullet	0	1	0	0	0	1	0	3
Mugilidae	<i>Liza macrolepis</i>	large-scale mullet	0	0	0	0	0	1	0	2
Mugilidae	<i>Liza tricuspidens</i>	striped mullet	1	0	0	0	0	1	0	3
Mugilidae	<i>Mugil cephalus</i>	flathead mullet	0	0	1	0	0	1	0	2
Mugilidae	<i>Myxus capensis</i>	freshwater mullet	1	1	0	0	0	1	0	3



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Mugilidae	<i>Valamugil cunnesius</i>	longarm mullet	0	0	0	0	0	1	0	2
Mugilidae	<i>Valamugil robustus</i>	robust mullet	0	0	0	0	0	1	0	2
Muraenidae	<i>Gymnothorax johnsoni</i>	whitespotted moray	1	0	0	0	0	0	0	2
Muraenidae	<i>Gymnothorax melatremus</i>	blackspot moray	0	0	0	0	1	0	0	2
Muraenidae	<i>Gymnothorax meleagris</i>	guineafowl moray	0	0	0	0	1	0	0	2
Myliobatidae	<i>Rhinoptera javanica</i>	flapnose ray	0	1	0	0	0	0	0	2
Narkidae	<i>Heteronarce garmani</i>	Natal electric ray	1	0	0	0	1	0	0	3
Odontaspidae	<i>Carcharias taurus</i>	spotted ragged-tooth shark	0	1	1	0	0	0	0	3
Ogcocephalidae	<i>Dibranchius stellulatus</i>	none	0	0	0	0	1	0	0	1
Ophichthidae	<i>Muraenichthys xorae</i>	orangehead worm eel	1	0	0	0	0	0	0	2
Ophidiidae	<i>Ophidion smithi</i>	Smith's cuskeel	0	0	0	0	1	0	0	1
Oplegnathidae	<i>Oplegnathus conwayi</i>	Cape knifejaw	1	0	1	0	0	0	0	3
Oplegnathidae	<i>Oplegnathus paeolopsis</i>	mozambique knifejaw	0	0	0	0	1	0	0	2
Oplegnathidae	<i>Oplegnathus robinsoni</i>	natal knifejaw	0	0	1	0	0	0	0	2
Parascorpididae	<i>Parascorpius typus</i>	jutjaw	1	0	0	0	0	0	0	2

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Pegasiidae	<i>Pegasus volitans</i>	longtail seamothe	0	0	0	0	1	0	0	2
Percophidae	<i>Osopsaron natalensis</i>	duckbill	1	0	0	0	1	0	0	1
Percophidae	<i>Pteropsaron Heemstrai</i>	duckbill	1	0	0	0	1	0	0	1
Pinguipedidae	<i>Parapercis maritzi</i>	gold-barred sandperch	1	0	0	0	0	0	0	2
Platycephalidae	<i>Grammoplites portugesus</i>	thorny flathead	1	0	0	0	0	0	0	2
Plesiopidae	<i>Plesiops multisquamatus</i>	spotted longfin	1	0	0	0	1	0	0	3
Pleuronectidae	<i>Paralichthodes algoensis</i>	Measels flounder	1	0	0	0	0	0	0	2
Plotosidae	<i>Plotosus nkunga</i>	eel-catfish	1	0	0	0	0	0	0	2
Pomacanthidae	<i>Apolemichthys kingi</i>	tiger angelfish	0	0	0	0	1	0	0	2
Pomacanthidae	<i>Pomacanthus chrysurus</i>	goldtail angelfish	0	0	0	0	1	0	0	2
Pomacentridae	<i>Abudefduf natalensis</i>	fourbar damsel	1	0	0	0	0	0	0	2
Pomacentridae	<i>Chromis dasygenys</i>	bluespotted chromis	1	0	0	0	0	0	0	2
Pomatomidae	<i>Pomatomus saltatrix</i>	shad	0	1	1	0	0	1	0	3
Pristidae	<i>Pristis microdon</i>	largetooth sawfish	0	1	1	0	0	0	0	3
Pristidae	<i>Pristis zijsron</i>	narrow snout sawfish	0	1	1	0	0	0	0	3

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Pristiophoridae	<i>Pliotrema warreni</i>	sixgill sawshark	0	1	1	0	0	0	0	3
Pseudocarchariidae	<i>Pseudocarcharias kamoharai</i>	crocodile shark	0	1	1	0	0	0	0	1
Rajidae	<i>Anacanthobatis marmoratus</i>	spotted legskate	1	0	0	0	1	0	0	3
Rajidae	<i>Cruriraja triangularis</i>	triangular legskate	1	0	0	0	0	0	0	2
Rajidae	<i>Dipterus springeri</i>	roughbelly skate	0	0	0	0	1	0	0	1
Rajidae	<i>Dipturus campbelli</i>	blackspot skate	1	1	0	0	1	0	0	3
Rajidae	<i>Raja lanceorostrata</i>	rattail skate	1	0	0	0	0	0	0	1
Rajidae	<i>Rostroraja alba</i>	spearnose skate	0	1	0	0	0	0	0	2
Rhincodontidae	<i>Rhincodon typus</i>	whale shark	0	1	1	0	0	0	0	3
Rhinobatidae	<i>Rhina ancylostoma</i>	bowmouth guitarfish	0	1	0	0	1	0	0	3
Rhinobatidae	<i>Rhinobatos annulatus</i>	lesser guitarfish	0	1	0	0	0	0	0	2
Rhinobatidae	<i>Rhinobatos leucospilus</i>	greyspot guitarfish	1	0	0	0	0	0	0	2
Rhinobatidae	<i>Rhynchobatus djiddensis</i>	giant guitarfish	0	1	0	0	0	0	0	2
Scaridae	<i>Chlorurus cyanescens</i>	blue humpheaded parrotfish	0	0	0	0	1	0	0	2
Scaridae	<i>Scarus festivus</i>	lunate parrotfish	0	0	0	0	1	0	0	2

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Sciaenidae	<i>Argyrosomus inodorus</i>	silver kob	0	1	0	0	0	0	0	2
Sciaenidae	<i>Argyrosomus japonicus</i>	dusky kob	0	1	1	0	0	1	0	3
Sciaenidae	<i>Argyrosomus thorpei</i>	squaretail kob	0	1	1	1	0	0	0	3
Sciaenidae	<i>Atractoscion aequidens</i>	geelbek	0	1	1	0	0	0	0	3
Sciaenidae	<i>Johnius dussumieri</i>	small kob	0	1	0	0	0	0	0	2
Sciaenidae	<i>Otolithes ruber</i>	snapper kob	0	1	1	0	0	0	0	3
Sciaenidae	<i>Umbrina robinsoni</i>	baardman	0	1	0	1	0	0	0	3
Sciorhinidae	<i>Haploblepharus edwardsii</i>	puffadder shyshark	0	1	0	1	0	0	0	3
Sciorhinidae	<i>Haploblepharus fuscus</i>	brown shyshark	0	1	0	0	0	0	0	2
Scomberidae	<i>Scomberomorus commerson</i>	king mackerel	0	1	0	0	0	0	0	2
Scomberidae	<i>Thunnus maccoyii</i>	southern bluefin tuna	0	1	0	0	0	0	0	2
Scomberidae	<i>Thunnus obesus</i>	bigeye tuna	0	1	0	0	0	0	0	2
Scomberidae	<i>Thunnus albacares</i>	yellowfin tuna	0	1	0	0	0	0	0	2
Scorpaenidae	<i>Choriodactylus natalensis</i>	threestick stingfish	1	0	0	0	0	0	0	2
Scorpaenidae	<i>Pterois mombasae</i>	deepwater firefish	0	0	0	0	1	0	0	2

Appendix 1: The criteria that KZN marine fish species qualified for, and the final list number for which each species qualified (see "List").

Family name	Scientific name	Common name	Endemic species	Species of conservation concern	Species with life history vulnerability	Highly resident species	Rare species	Species dependent on estuaries	Species dependent on specialised habitats	List
Scorpaenidae	<i>Rhinopias frondosa</i>	popeyed scorpionfish	0	0	0	0	1	0	0	2
Scorpaenidae	<i>Scorpaenopsis gilchristi</i>	Gilchrist's scorpionfish	0	0	0	0	1	0	0	1
Scorpaenidae	<i>Sebastapistes tinkhami</i>	darkspotted scorpionfish	0	0	0	0	1	0	0	2
Scorpididae	<i>Neoscorpis lithophilus</i>	stonebream	1	0	0	0	0	0	0	2
Scyliorhinidae	<i>Halaelurus lineatus</i>	banded catshark	1	1	0	0	0	0	0	3
Scyliorhinidae	<i>Holohalaelurus punctatus</i>	spotted catshark	0	1	0	0	0	0	0	2
Scyliorhinidae	<i>Holohalaelurus punctatus</i>	whitespotted izak	0	1	0	0	0	0	0	2
Scyliorhinidae	<i>Holohalaelurus regani</i>	honeycomb izak	1	1	0	0	0	0	0	3
Scyliorhinidae	<i>Poroderma pantherium</i>	blackspotted catshark	0	0	0	0	1	0	0	2
Serranidae	<i>Anthias connelli</i>	harlequin goldie	1	0	0	0	1	0	0	3
Serranidae	<i>Cephaloppholis aurantia</i>	golden rockcod	0	0	0	0	1	0	0	2
Serranidae	<i>Epinephelus albomarginatus</i>	captain fine	1	1	1	0	0	0	0	3
Serranidae	<i>Epinephelus marginatus</i>	yellowbelly rock-cod	0	1	1	1	0	0	0	3
Serranidae	<i>Epinephelus andersoni</i>	catface rockcod	1	1	1	1	0	0	0	3
Serranidae	<i>Epinephelus lanceolatus</i>	brindle bass	0	1	1	0	0	0	0	2

Appendix 1: The criteria that KZN marine fish species qualified for, and the final list number for which each species qualified (see “List”).

<i>Family name</i>	<i>Scientific name</i>	<i>Common name</i>	<i>Endemic species</i>	<i>Species of conservation concern</i>	<i>Species with life history vulnerability</i>	<i>Highly resident species</i>	<i>Rare species</i>	<i>Species dependent on estuaries</i>	<i>Species dependent on specialised habitats</i>	<i>List</i>
Serranidae	<i>Epinephelus malabaricus</i>	malabar rockcod	0	0	0	1	0	0	0	3
Serranidae	<i>Epinephelus rivulatus</i>	halfmoon rockcod	0	0	0	1	0	0	0	2
Serranidae	<i>Epinephelus tukula</i>	potato bass	0	1	1	1	0	0	0	3
Soleidae	<i>Heteromycteris capensis</i>	Cape sole	0	0	0	0	0	1	0	2
Soleidae	<i>Solea bleekeri</i>	blackhand sole	0	0	0	0	0	1	0	2
Soleidae	<i>Synaptura marginata</i>	shallow water sole	1	0	0	0	0	0	0	2
Soleidae	<i>Zebrias regani</i>	zebra sole	1	0	0	0	0	0	0	2
Sparidae	<i>Acanthopagrus vagus</i>	riverbream	1	0	1	1	0	1	0	3
Sparidae	<i>Argyrops spinifer</i>	king soldier bream	0	0	1	0	0	0	0	2
Sparidae	<i>Boopsoidea inornata</i>	fransmadam	1	0	0	0	0	0	0	2
Sparidae	<i>Chrysoblephus anglicus</i>	englishman	1	1	0	0	0	0	0	2
Sparidae	<i>Chrysoblephus cristiceps</i>	dageraad	1	1	0	0	0	0	0	3
Sparidae	<i>Chrysobelphus lophus</i>	false englishman	1	0	0	0	0	0	0	3
Sparidae	<i>Chrysoblephus puniceus</i>	slinger	1	1	0	0	0	0	0	3
Sparidae	<i>Cymatoceps nasutus</i>	black musselcracker	1	1	0	1	0	0	0	3

Appendix 1: The criteria that KZN marine fish species qualified for, and the final list number for which each species qualified (see “List”).

Family name	Scientific name	Common name	Endemic species	Species of conservation concern	Species with life history vulnerability	Highly resident species	Rare species	Species dependent on estuaries	Species dependent on specialised habitats	List
Sparidae	<i>Diplodus cervinus hottentotus</i>	zebra	1	0	0	1	0	0	0	3
Sparidae	<i>Diplodus sargus capensis</i>	blacktail	1	0	0	1	0	0	0	3
Sparidae	<i>Pachymetopon aeneum</i>	blue hottentot	1	1	0	0	0	0	0	3
Sparidae	<i>Pachymetopon grande</i>	bronze bream	1	0	1	0	0	0	0	3
Sparidae	<i>Pagellus bellottii natalensis</i>	sand soldier	0	1	0	0	0	0	0	2
Sparidae	<i>Petrus rupestris</i>	red steenbras	1	1	1	0	0	0	0	3
Sparidae	<i>Polyamblyodon germanum</i>	german	1	0	0	0	0	0	0	2
Sparidae	<i>Polysteganus coeruleopunctatus</i>	blueskin seabream	0	1	0	0	0	0	0	2
Sparidae	<i>Polysteganus praeorbitalis</i>	scotsman	1	1	1	1	0	0	0	3
Sparidae	<i>Polysteganus undulosus</i>	seventy four	1	1	1	0	0	0	0	3
Sparidae	<i>Porcostoma dentata</i>	dane	1	1	0	0	0	0	0	3
Sparidae	<i>Pterogymnus lanarius</i>	panga	0	1	0	0	0	0	0	2
Sparidae	<i>Rhabdosargus holubi</i>	Cape stumpnose	1	0	0	0	0	1	0	3
Sparidae	<i>Rhabdosargus sarba</i>	Natal stumpnose	0	0	0	1	0	1	0	2
Sparidae	<i>Rhabdosargus thorpei</i>	bigeye stumpnose	1	0	0	0	0	1	0	3

Appendix 1: The criteria that KZN marine fish species qualified for, and the final list number for which each species qualified (see “List”).

<i>Family name</i>	<i>Scientific name</i>	<i>Common name</i>	<i>Endemic species</i>	<i>Species of conservation concern</i>	<i>Species with life history vulnerability</i>	<i>Highly resident species</i>	<i>Rare species</i>	<i>Species dependent on estuaries</i>	<i>Species dependent on specialised habitats</i>	<i>List</i>
Sparidae	<i>Sarpa salpa</i>	strepie	0	0	1	0	0	0	0	2
Sphyraenidae	<i>Sphyraena barracuda</i>	great barracuda	0	0	0	1	0	1	0	2
Sphyrnidae	<i>Sphyrna lewini</i>	scalloped hammerhead	0	1	0	0	0	0	0	2
Sphyrnidae	<i>Sphyrna mokarran</i>	great hammerhead	0	1	0	0	1	0	0	3
Sphyrnidae	<i>Sphyrna zygaena</i>	smooth hammerhead	0	1	0	0	0	0	0	2
Syngnathidae	<i>Doryrhamphus birainatus</i>	narrowstripe pipefish	1	0	0	0	0	0	0	2
Syngnathidae	<i>Hippichthys heptagonus</i>	belly pipefish	0	0	0	0	0	1	0	2
Syngnathidae	<i>Hippichthys spicifer</i>	bellybarred pipefish	0	0	0	0	1	1	0	2
Syngnathidae	<i>Hippocampus whitei</i>	crowned seahorse	0	1	0	0	1	0	0	2
Syngnathidae	<i>Hippocampus camelopardalis</i>	giraffe seahorse	0	1	0	0	0	0	0	2
Syngnathidae	<i>Hippocampus histrix</i>	thorny seahorse	0	1	0	0	0	0	0	2
Syngnathidae	<i>Hippocampus kuda</i>	yellow seahorse	0	1	0	0	0	0	0	2
Syngnathidae	<i>Hippocampus trimaculatus</i>	longnose seahorse	0	1	0	0	0	0	0	3
Syngnathidae	<i>Microphis fluviatilis</i>	freshwater pipefish	0	0	0	0	0	1	0	2
Syngnathidae	<i>Microphis brachurus</i>	short-tail pipefish	0	0	0	0	1	1	0	2



Appendix 1: The criteria that KZN marine fish species qualified for, and the final list number for which each species qualified (see "List").

<i>Family name</i>	<i>Scientific name</i>	<i>Common name</i>	<i>Endemic species</i>	<i>Species of conservation concern</i>	<i>Species with life history vulnerability</i>	<i>Highly resident species</i>	<i>Rare species</i>	<i>Species dependent on estuaries</i>	<i>Species dependent on specialised habitats</i>	<i>List</i>
Syngnathidae	<i>Syngnathus acus</i>	longsnout pipefish	0	0	0	0	0	1	0	2
Syngnathidae	<i>Syngnathus watermeyeri</i>	river pipefish	0	0	0	0	0	1	0	2
Teraponidae	<i>Terapon jarbua</i>	thornfish	0	0	0	0	0	1	0	2
Tetraodontidae	<i>Canthigaster smithae</i>	bicoloured toby	0	0	0	0	1	0	0	2
Tetraodontidae	<i>Chelonodon pleurospilus</i>	blaasop beauty	1	0	0	0	0	0	0	2
Tetraodontidae	<i>Takifugu oblongus</i>	lattice blaasop	0	0	0	0	1	0	0	2
Tetraodontidae	<i>Torquigener marleyi</i>	slender puffer	1	0	0	0	1	0	0	3
Trichiuridae	<i>Trichiurus lepturus</i>	cutlass	0	1	0	0	0	0	0	2
Trichonotidae	<i>Trichonotus marleyi</i>	sand diver	1	0	0	0	0	0	0	2
Triglidae	<i>Trigloporus lastoviza africanus</i>	african gurnard	1	0	0	0	0	0	0	1

Appendix 2: The species from List Three that are of conservation concern and the qualifying conditions met.

<i>Scientific name</i>	<i>Common name</i>	<i>IUCN Red List Status</i>	<i>CITES listed</i>	<i>TOPs Listed</i>	<i>NSBA listed</i>	<i>Overexploited</i>	<i>Published concern</i>
<i>Pristis zijsron</i>	narrow snout sawfish	CR	Appendix 1	P	1	0	0
<i>Latimeria chalumnae</i>	coelacanth	CR	Appendix 1	P	0	0	0
<i>Pristis microdon</i>	large tooth sawfish	CR	Appendix 2	E	1	0	0
<i>Holohalaelurus regani</i>	honeycomb izak	CR			0	0	1
<i>Sphyrna mokarran</i>	great hammerhead	EN			0	0	1
<i>Epinephelus marginatus</i>	yellowbelly rock-cod	EN			0	0	0
<i>Liza luciae Penrith</i>	St. Lucia mullet	EN			0	0	0
<i>Rhincodon typus</i>	whale shark	VU	Appendix 1		0	0	0
<i>Carcharodon carcharias</i>	great white shark	VU	Appendix 1	P	0	0	0
<i>Epinephelus lanceolatus</i>	brindle bass	VU		P	1	0	0
<i>Myxus capensis</i>	freshwater mullet	VU		P	0	0	0
<i>Scylliogaleus queckettii</i>	flapnose houndshark doublesash	VU			0	0	0
<i>Chaetodon marleyi</i>	butterflyfish	VU			0	0	0
<i>Rhina ancylostoma</i>	bowmouth guitarfish spotted ragged-tooth	VU			0	0	0
<i>Carcharias taurus</i>	shark	VU			0	0	0
<i>Epinephelus albomarginatus</i>	captain fine	VU			0	0	0
<i>Urogymnus asperrimus</i>	porcupine ray	VU			0	0	0

Appendix 2: The species from List Three that are of conservation concern and the qualifying conditions met.

<i>Scientific name</i>	<i>Common name</i>	<i>IUCN Red List Status</i>	<i>CITES listed</i>	<i>TOPs Listed</i>	<i>NSBA listed</i>	<i>Overexploited</i>	<i>Published concern</i>
<i>Anchichoerops natalensis</i>	Natal wrasse		Appendix 2	P	0	0	0
<i>Hippocampus whitei</i>	crowned seahorse		Appendix 2		0	0	0
<i>Polysteganus undulosus</i>	seventy four			P	1	0	1
<i>Epinephelus tukula</i>	potato bass			P	1	0	0
<i>Epinephelus andersoni</i>	catface rockcod	NT		VU	0	0	0
<i>Argyrosomus japonicus</i>	dusky kob				1	0	0
<i>Chrysoblephus cristiceps</i>	dageraad				1	0	0
<i>Chrysoblephus puniceus</i>	slinger				1	0	0
<i>Cymatoceps nasutus</i>	black musselcracker				1	0	0
<i>Polysteganus praeorbitalis</i>	scotsman				1	0	0
<i>Petrus rupestris</i>	red steenbras				1	0	0
<i>Argyrosomus thorpei</i>	squaretail kob				0	1	1
<i>Atractoscion aequidens</i>	geelbek				0	1	0
<i>Chrysoblephus anglicus</i>	englishman				0	1	0
<i>Porcostoma dentata</i>	dane				0	1	0
<i>Pomatomus saltatrix</i>	shad				0	1	0
<i>Dichistius capensis</i>	galjoen				0	1	0

Appendix 2: The species from List Three that are of conservation concern and the qualifying conditions met.

<i>Scientific name</i>	<i>Common name</i>	<i>IUCN Red List Status</i>	<i>CITES listed</i>	<i>TOPs Listed</i>	<i>NSBA listed</i>	<i>Overexploited</i>	<i>Published concern</i>
<i>Pachymetopon aeneum</i>	blue hottentot				0	1	0
<i>Carcharhinus leucas</i>	bull shark	NT			0	0	1
<i>Pliotrema warreni</i>	sixgill sawshark	NT			0	0	1
<i>Dipturus campbelli</i>	blackspot skate	NT			0	0	1
<i>Haploblepharus edwardsii</i>	puffadder shyshark				0	0	1
<i>Halaelurus lineatus</i>	banded catshark				0	0	1
<i>Umbrina robinsoni</i>	baardman				0	0	1
<i>Otolithes ruber</i>	snapper kob				0	0	1

Appendix 3: Endemic species that qualified for List Three (n = 43).

<i>Scientific name</i>	<i>Common name</i>	<i>Endemicity</i>
<i>Torquigener marleyi</i>	slender puffer	Range restricted
<i>Silhouettea sibayi</i>	barebreasted goby	Range restricted
<i>Anthias connelli</i>	harlequin goldie	KZN provincial endemic
<i>Taeniodes jacksoni</i>	bearded eelgoby	KZN provincial endemic
<i>Spicara australis</i>	picarel	KZN provincial endemic
<i>Plesiops multisquamatus</i>	spotted longfin	KZN provincial endemic
<i>Croilia mossambica</i>	naked goby	Delagoa bioregion endemic
<i>Holohalaelurus regani</i>	honeycomb izak	Natal & Delagoa bioregions endemic
<i>Epinephelus andersoni</i>	catface rockcod	Natal & Delagoa bioregions endemic
<i>Dipturus campbelli</i>	blackspot skate	Natal & Delagoa bioregions endemic
<i>Anacanthobatis marmoratus</i>	spotted legskate	Natal & Delagoa bioregions endemic
<i>Oplegnathus conwayi</i>	Cape knifejaw	Natal & Agulhas bioregion endemic
<i>Petrus rupestris</i>	red steenbras	Natal & Agulhas bioregion endemic
<i>Heteronarce garmani</i>	Natal electric ray	Natal & Agulhas bioregion endemic
<i>Chrysoblephus cristiceps</i>	dageraad	Natal & Agulhas bioregion endemic
<i>Diplodus sargus capensis</i>	blacktail	Natal & Agulhas bioregion endemic
<i>Cymatoceps nasutus</i>	black musselcracker	Natal & Agulhas bioregion endemic
<i>Pachymetopon aeneum</i>	blue hottentot	Natal & Agulhas bioregion endemic
<i>Diplodus cervinus hottentotus</i>	zebra	Natal & Agulhas bioregion endemic
<i>Pavoclinus mentalis</i>	bearded klipfish	Natal & Agulhas bioregion endemic
<i>Polysteganus praeorbitalis</i>	scotsman	Natal & Agulhas bioregion endemic
<i>Polysteganus undulosus</i>	seventy four	Natal & Agulhas bioregion

Appendix 3: Endemic species that qualified for List Three (n = 43).

<i>Scientific name</i>	<i>Common name</i>	<i>Endemicity</i>
		endemic
<i>Caffrogobius natalensis</i>	baldy	Natal & Agulhas bioregion endemic
<i>Rhabdosargus holubi</i>	Cape stumpnose	Natal & Agulhas bioregion endemic
<i>Scylliogaleus queckettii</i>	flapnose houndshark	Natal & Agulhas bioregion endemic
<i>Anchichoerops natalensis</i>	Natal wrasse	Natal & Agulhas bioregion endemic
<i>Pachymetopon grande</i>	bronze bream	Natal & Agulhas bioregion endemic
<i>Chirodactylus jessicalenorum</i>	Natal fingerfin	East coast endemic
<i>Chrysoblephus anglicus</i>	englishman	East coast endemic
<i>Chrysoblephus puniceus</i>	slinger	East coast endemic
<i>Chaetodon marleyi</i>	doublesash butterflyfish	East coast endemic
<i>Epinephelus albomarginatus</i>	captain fine	East coast endemic
<i>Gerres methueni</i>	evenfin pursemouth	East coast endemic
<i>Halaelurus lineatus</i>	banded catshark	East coast endemic
<i>Acanthopagrus vagus</i>	riverbream	East coast endemic
<i>Liza tricuspidens</i>	striped mullet	East coast endemic
<i>Myxus capensis</i>	freshwater mullet	East coast endemic
<i>Porcostoma dentata</i>	dane	East coast endemic
<i>Redigobius dewaalii</i>	checked goby	East coast endemic
<i>Rhabdosargus thorpei</i>	bigeye stumpnose	East coast endemic
<i>Spicara axillaris</i>	windtoy	East coast endemic
<i>Taeniodes esquivel</i>	bulldog eelgoby	East coast endemic
<i>Hyporhamphus capensis</i>	Cape halfbeak	East coast endemic

## Appendix 4

### Data provided by the institutions and organisations

<i>Data set</i>	<i>SAIAB</i>	<i>IZIKO</i>	<i>NMLS</i>	<i>ORI/WWF-SA tagging</i>	<i>KZNSB</i>	<i>Boat Launch Site Monitoring data</i>	<i>Fisheries independent data</i>
Data type	Recreational, coordinated research	Recreational and coordinated research	Recreational and commercial	Recreational fishing, tag and recapture	Bycatch from anti-shark nets and drum lines	Recreational, ski boat	Recreational shore-based catches
Species focus	Wide	Wide	Important linefish	Line fish, > 30 cm and 500 g	Large marine species, mostly elasmobranchs, cetaceans, marine turtles	Game and demersal fish, line fish	Important line and bait fish
Research aims	Systematics and taxonomy	Systematics and taxonomy, Biology	Fish stock assessments, fisheries management	Biology, migrations routes, stock assessments	Biology, migratory patterns, anti-shark impacts	Catch assessments, stock assessments	Assessment of under and miss-reporting of NMLS
Data contributors	Various, researchers, ad hoc public submissions	Ad hoc public collections, coordinated research collections, data and specimens accessioned from outside the organisation	Various fishing operations; recreational only KZN, commercial SA	Recreational shore and ski-boat fishing, including coordinated tagging efforts in Cape Vidal and De Hoop	KZNSB	Recreational, ski-boat and all other craft-based capture data	Recreational shore-based catches
Full extent of data	World Wide, focussed on Southern Africa	World Wide, focussed on Southern Africa	Southern Africa, focussed SA	South Africa, focussed in Western Cape and Cape Vidal	KZN	KZN	KZN
Data housed	SAIAB	Iziko Museum Cape Town	MCM, also available through ORI	ORI	KZNSB	ORI	ORI
Spatial information	Coordinates, locality descriptions, place names	Coordinates, locality descriptions, place names	Beach locality code, code matching locality names	Beach locality code, code matching locality names	Beach locality code, code matching locality names, net number	Beach locality code, code matching locality names	Beach locality code, code matching locality names
Estimated spatial uncertainty	50 m - 360 km, see Figure 3	50 m - 360 km, see Figure 3	1 - 20 km, see Table 5	1 - 20 km	1 km	ca 20 km	1 km
Other errors or limitations	Historical data have lower spatial resolution than post 2000. Pre 2006 data have less environmental information.	Historical data have larger uncertainty associated than newer data, which may be accompanied by GPS coordinated	Poor species identification	Non-reporting of tags, species identification	no data north of Richards Bay	Very poor species identification	Small data set, with long time periods between data sets.

Quality assessment	Quality control for accuracy of coordinates, species identification		checked on entry by ORI, and MCM	Quality control of data submitted by ORI, species identification in the case of a recapture	KZNSB scientists review odd records*	Checked for obvious records that are misplaced	ORI
Reference data			National Marine Linefish System and the data type prepared by MCM. OR Reference provided by ORI report,	ORI data reports		ORI data reports	ORI data reports
Processing steps required	Quality control for accuracy of coordinates	Quality control for accuracy of coordinates	Coordinates have to be associated	Coordinates have to be associated.	Coordinates have to be associated	Coordinates have to be associated	Coordinates have to be associated
Online facility available from	SAIAB (limited information), OBIS (limited information, but includes coordinates), FishBase	OBIS, limited information	OBIS, as MCM-Line and MCM-DEM data sets, limited information	Tagging News , and more detail on request	NA	NA	NA
Online data last updated	SAIAB - daily; OBIS 2008, FishBase 2004	OBIS	2006	NA	NA	NA	NA

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## Appendix 5

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This appendix provides the technical information and specifications of data preparation, GIS data processing, and basic metadata. This appendix is not intended to be an exhaustive account of exactly every step taken in data processing, but to supplement the technical data processing components not covered in the methods. Acronyms are also not redefined. The Appendix and Methods section are to be read together to provide more details to assist future duplication of the methods used.

### Software

Idrisi Taiga v16.5

ESRI ArcView 3.2

Maxent 3.3.3.a

### Georeferencing of point locality data

The GIS operation used to georeference point locality data without coordinates is described below. The coordinates for data that projected overland as a result of being recorded to the shoreline or because of being recorded to a beach locality code (see Chapter Two) were adjusted 500 m perpendicular to the coastline in an offshore direction. Adjusting the shore-based data by 500 m offshore was justified because distance moved was less than the spatial resolution of a single cell.

### Beach code locality system

Data recorded to the beach locality code system form the bulk of the available data (see Chapter Two), and were georeferenced over the marine environment of KZN in two steps.

This included NMLS shore and boat based, ORI/WWS-SA, and KZN Sharks Board data. The 561 beach locality codes in KZN were initially georeferenced to their positions along the coastline. The beach locality codes indicate a km interval along the shore, starting at the number 3565 in the North (Kosi Bay), and ending at 4125 in the South (Umtamvuna River mouth) in KZN. I used GIS software (ArcView 3.2) and a script, divide2.avx, to divide the KZN high water line used in SeaPLAN (Lagabrielle et al. 2010) into 561 equidistant intervals of one km and placed a point at the start of each interval, to duplicate the beach locality code system. The 561 points were assigned the matching beach locality codes and attributed with coordinates in the GIS software (using SANTI Tools GetXY), such that each beach locality was now associated with a georeferenced point on a map of KZN.

### **Adjusting coordinates offshore**

Secondly, the coordinates of all shore-based data were adjusted such that all data projected over the marine environment for modelling purposes. This included both beach locality code data (NMLS shore angling, NMLS shore patrol, and ORI/WWS-SA, KZN Sharks Board) and previously georeferenced data (SAIAB, Iziko) which projected over land. In ArcView 3.2 GIS, I generated a buffer of 1000 m around each overland data point, then clipped it by planning region (KZN EEZ), then determined its centroid using XTools. This process moved the data ca 500 m perpendicular to the coastline in an offshore direction. The coordinates were determined for all records, using the SANTI TOOLS GetXY extension in ArcView.

The coordinates for boat-based data that were recorded to beach locality codes, with distances offshore, were determined by adding the distance offshore to the longitude of the beach locality code.

### **Technical processing of remotely-sensed data (RSD)**

Global RSD data (SST, Chl, Cdom, Kd) for monthly climatologies were downloaded in HDF format and converted to raster files, using the Idrisi Import HDF4 module. HDF files were imported as plane coordinates (not projected into Idrisi). The extents were calculated from the number of columns and rows per image and projected to WGS84 (Latlong.ref). Extents were adjusted by adding half the cell resolution to extent values (see Table 1).

**Table 1. The extent calculations applied to project RSD into Latlong in Idrisi.**

<i>Environmental variable</i>	<i>Adjust by half image resolution for extent<sup>a</sup></i>	<i>X, left corner</i>	<i>Y, left corner</i>	<i>Sensor</i>
Cdom	0.041666667/2= 0.0208333335	+180.0208333335	-179.9791666665	MODIS
Chl	0.041666667/2= 0.0208333335	+180.0208333335	-179.9791666665	MODIS
Kd	0.041666667/2= 0.0208333335	+180.0208333335	-179.9791666665	MODIS
SST	0.043945313/2 = 0.0219726565	+180.0219726565	-179.9780273435	Pathfinder <sup>b</sup>

<sup>a</sup>Shift from zero was because Idrisi georeferences to upper left corner values of cells, while hdfs files are georeferenced to centres of cells (see <http://hdfeos.net/forums/showthread.php?t=302>).

<sup>b</sup>Note all MODIS RSD values are the same, but are different to AVHRR pathfinder (SST).

The data were 'windowed' (clipped) in Idrisi to the planning area (KZN EEZ). Monthly climatologies were grouped into annual, summer and winter raster file groups, as the seasonal cut-offs specify in the Methods section of Chapter Three. Minimum and maximum values were calculated for the 4.6 km<sup>2</sup> resolution raster groups for all RSD variables using the Idrisi min and max modules. The data were reprojected to match the planning area's extent and resolution and projection to WGS84, central line 31 degrees (W31.ref). During reprojection from 4.6 km<sup>2</sup> to 1 km<sup>2</sup>, data values were interpolated from neighbouring values. Missing values in the gaps between marine and shoreline data were also interpolated from neighbouring values. The resulting images all shared the tabulated metadata features as required for Maxent (see Table 2).

**Table 2. Scale and projection metadata for GIS layers used in Maxent.**

<i>Metadata</i>	<i>Values</i>
Columns	634
Rows	703
Min X	-80000.0
Max X	554000.0
Min Y	-3676000.0
Max Y	-2972000.0
X resolution	1000. 0
Y resolution	1000.0
Units	Meters
Ref. System	WGS 84 UTM31 S

## **Maxent**

### **Environmental variables**

All background values were defined in the Idrisi metadata for export purposes to ESRI ARC Raster (ASCII).

### **Point locality data**

All unique values were grouped into annual, summer and winter data sets. Unique values were identified as values with unique species identifier, and X and/or Y coordinates. All values were then exported to three respective CSV files, annual, summer, and winter.

### **Background bias file**

All models require information on the background environmental variables, also termed pseudo-absences in presence-only modelling methods (Phillips et al. 2009). It has been shown that the predictive strength of distribution modelling is substantially improved by using a target group background selection to sample environmental variable values (Phillips et al. 2009). The area from which the software randomly selects background information was limited to the area from the shore to the 1000 m depth contour, in order to exclude as many undersampled areas as possible, and also areas where it is known that the species does not occur (Phillips & Dudík 2008; Phillips et al. 2009).

**Rationale**

The RSD values used in this study are known to affect species distributions (Agenbag et al. 2003; Hiddink & Ter Hofstede 2008; Skov et al. 2008) but predominantly at the mesoscale level (Phillips et al. 2006). The role of RSD is artificially reduced as a result of the small provincial scale used in this study. The background was limited to the area inside of the 1000 m depth contour to compensate for the loss of RSD contribution to Maxent SDMs.

The 1000m depth contour was used as it captured enough environmental variability of the RSD environmental parameters, and included all sample points. The literature suggested that none of the SeaPLAN species occur in water deeper than 1000 m, except for the whale shark (van der Elst & Vermeulen 1986; Compagno et al. 1989; van der Elst 1989; van der Elst & Thorpe 1989; Mann 2000; Heemstra & Heemstra 2004). Ten thousand background data points are recommended for Maxent models (Phillips et al. 2009). The background files were only further reduced for severely under sampled species because too small an area has the effect of reducing the number of background data points below 10000.

**Estimating spatial bias**

CHARMs represent the species' known ranges. The point locality data were overlaid onto the CHARM to estimate the sampling bias for each species. Areas lacking point locality data, where the species is known to occur, were turned into background bias files. The background bias files included MPAs A and B categories for species that predominantly occur in the north of the planning area. Several species that occur in that area suffer from under sampling owing to the large MPAs (especially restricted bottom fish). Similarly, the area 10 km around the Amatikulu River mouth and at 5 nm offshore lacked data for several shark species that do occur here. Some species of shark have data derived only from shark nets, which cover the south and central areas as far as Richards Bay. Shark nets are restricted to the nearshore environment (Dudley & Simpfendorfer 2006), hence several species are undersampled in the north of KZN and offshore. The bias file was restricted to the first 1500 m offshore and extended to Cape Vidal.

Modelling the distribution of the whale shark was treated separately, as the sampled area for this species is well known. The data were almost exclusively flight data that are collected within a 5 km band along the shoreline during flights.

## **Resampling**

'Cross validation' was the preferred resampling method because of the added advantage of using the full data set for evaluating the resulting model which may be particularly useful for those species with few records (Phillips et al. 2006).

## **Regularization parameter**

A low regularization parameter acts to constrain the prediction closer to the points, while higher values relax the fit (Phillips et al. 2006). During the third run, the regularization parameter was applied to the models that under predicted. This approach was favoured over using a very small background bias file which ran the risk of too low variability in some of the environmental parameters, like dissolved particulate organic matter (Cdom), to be informative for the software (Phillips et al. 2006). Regularization parameters were adjusted for species with bias files that limited the offshore distribution, e.g. bull shark, mostly shark net data, bias file limited distribution to 1500 m offshore. The referenced distance offshore of the species was divided with a constant value of 15 for all shark net biased data, e.g. 150 m depth maximum yields a regularization parameter of 10. Species that were biased by the removal of Cape Vidal North or MPAs were investigated but the results showed that further regularization did not appear necessary.

In order to improve fit, the regularization parameter was calculated in proportion to which Maxent SDMs were under predicting area of occurrence. The regularization parameter was adjusted based on the difference between the distance predicted offshore for presence and the 'known' distance based on the literature. Maxent models were converted to presence-absence images by applying a minimum cell value threshold. The threshold was set to the value of minimum cell value at training presence localities. The difference between the depth at which the CHARM and Maxent SDMs end was calculated. The regularization value was then set as the difference multiplied by three.

## **Environmental variable contribution to model**

Maxent plots response curves (as part of the Maxent output) of the change in probability of occurrence in response to change of each variable, respectively. The response curves do not consider relative contribution of the other variables and therefore indicate nothing more than response to change in variable value. Highly correlated variables are therefore at risk of appearing more important than they actually are. To discern the relative contribution of all variables, Maxent calculates relative variable contribution as a percentage out of 100, as follows:

The contribution of each variable to the model is calculated by adding or subtracting the change in regularised training gain to the contribution of the corresponding variable after each iteration of the training algorithm. As a second estimate, the sequence of the values of the variables are changed (iteratively for each variable), and the resulting change in AUC evaluated. The changes in AUC are normalized, and presented as a percentage, and allow for comparison of relative variable contribution.

Maxent also uses jack-knifing to estimate variable contribution in two ways to calculate the change in AUC and regularized training gain: Running the same model but (1) excluding one variable at a time, and (2), using only one variable at a time. The change in regularized training gain and AUC value indicates the relative variable contribution to the overall model.

Phillips et al. (2006) warns users that the contribution of variables that are highly correlated cannot be evaluated effectively using either of these methods. Using highly correlated variables should however be avoided as far as possible (Phillips et al. 2006).

## **CHARMS**

The environmental layers were converted to Boolean presence/absence (1/0) raster images, using Idrisi Taiga v16.4 and multiplied to create a presence/absence raster image for total and preferred range for each species. The minimum information required to construct a CHARM is latitude range (distribution limits) and depth range.

CHARMs were modelled at one km<sup>2</sup> owing to the environmental variable with the poorest resolution being rock reef (mapped at one km<sup>2</sup> in SeaPLAN, Lagabrielle et al. (2010)). Qualitative range descriptions used to construct CHARMs depend only on the resolution of the environmental variables.

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## Appendix 6

### Summed summer and summed winter SDMs

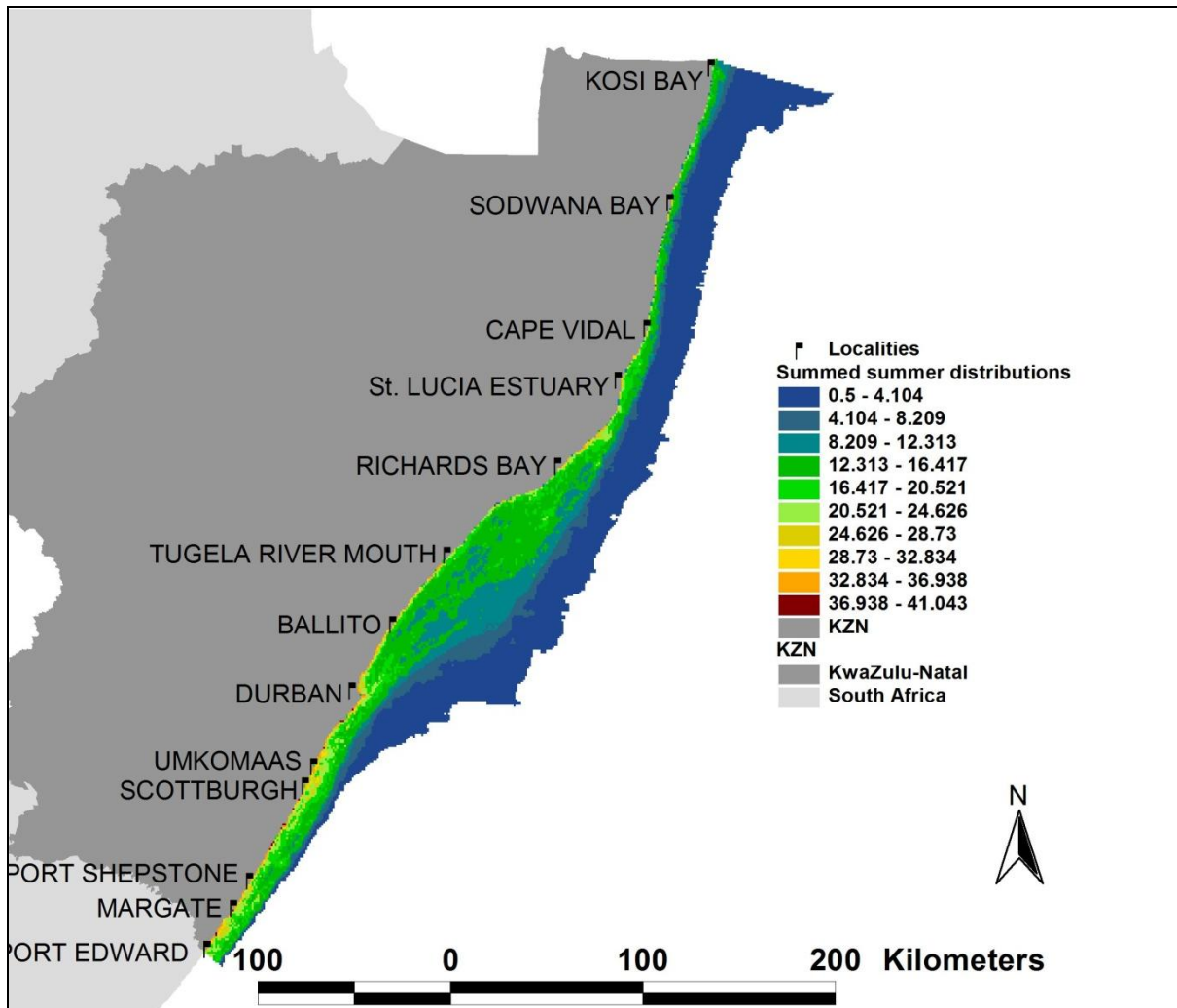


Figure 1. Summed summer SDMs in KZN. The map shows lower values in the KZN south and central coast than the summed winter map (below). Although values on the north coast are lower than on the south and central coasts, they are higher than the in the summed winter map. Nearshore areas along the south coast (Durban and Richards Bay) have the highest values.

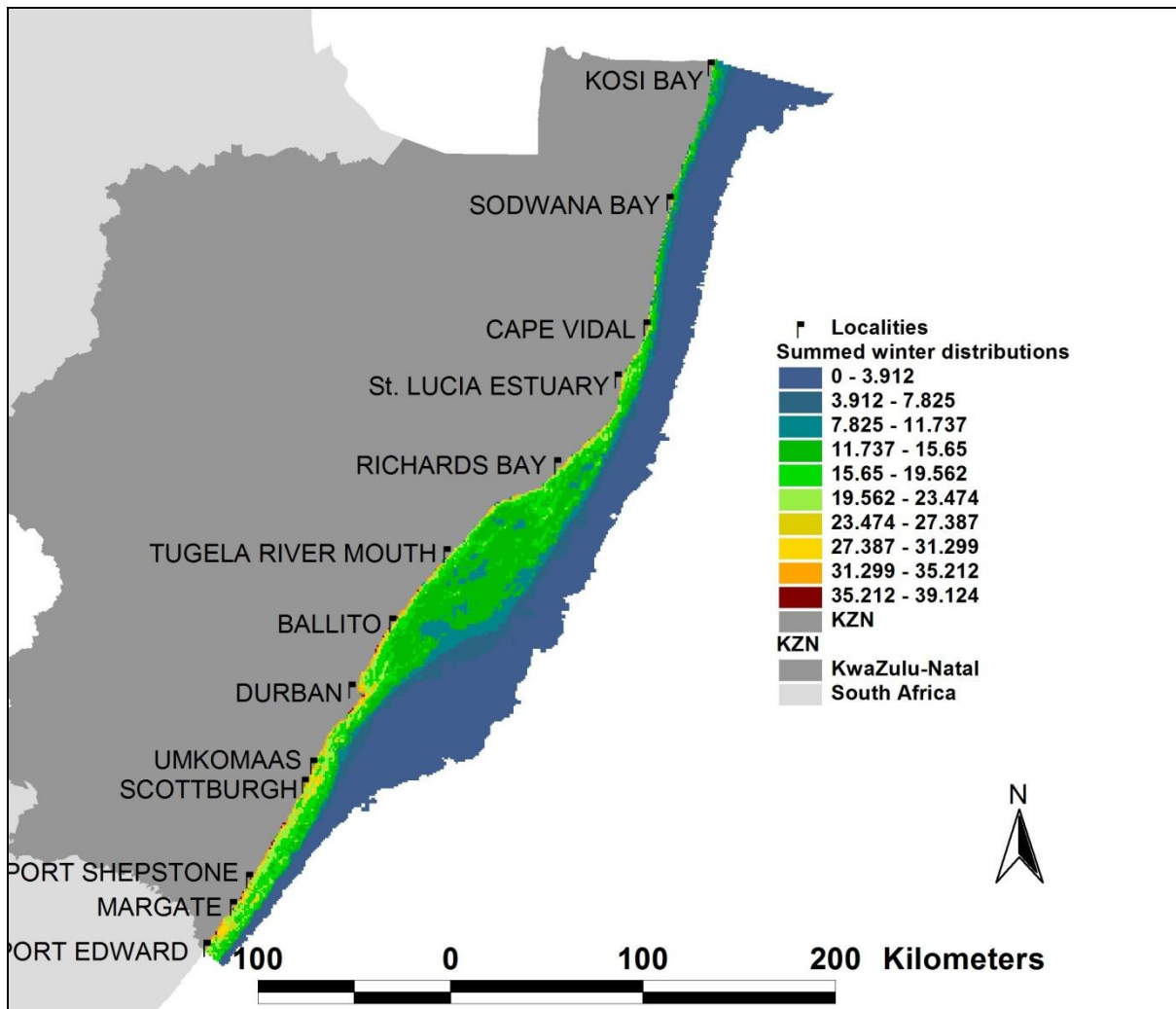


Figure 2. Summed winter SDMs, showing high values in the south and central coast, and lower values on the north coast than the summer map (above). The values on the north coast are much lower than for the summer map. Although the pattern is similar to the summer map, winter has higher values that extend over the offshore reefs in the south coast and Durban reefs area. This may be reflecting the presence of winter migrants from the southern Cape, that spawn over offshore reefs, e.g. geelbek (*Atractoscion aequidens*).

## Appendix 7

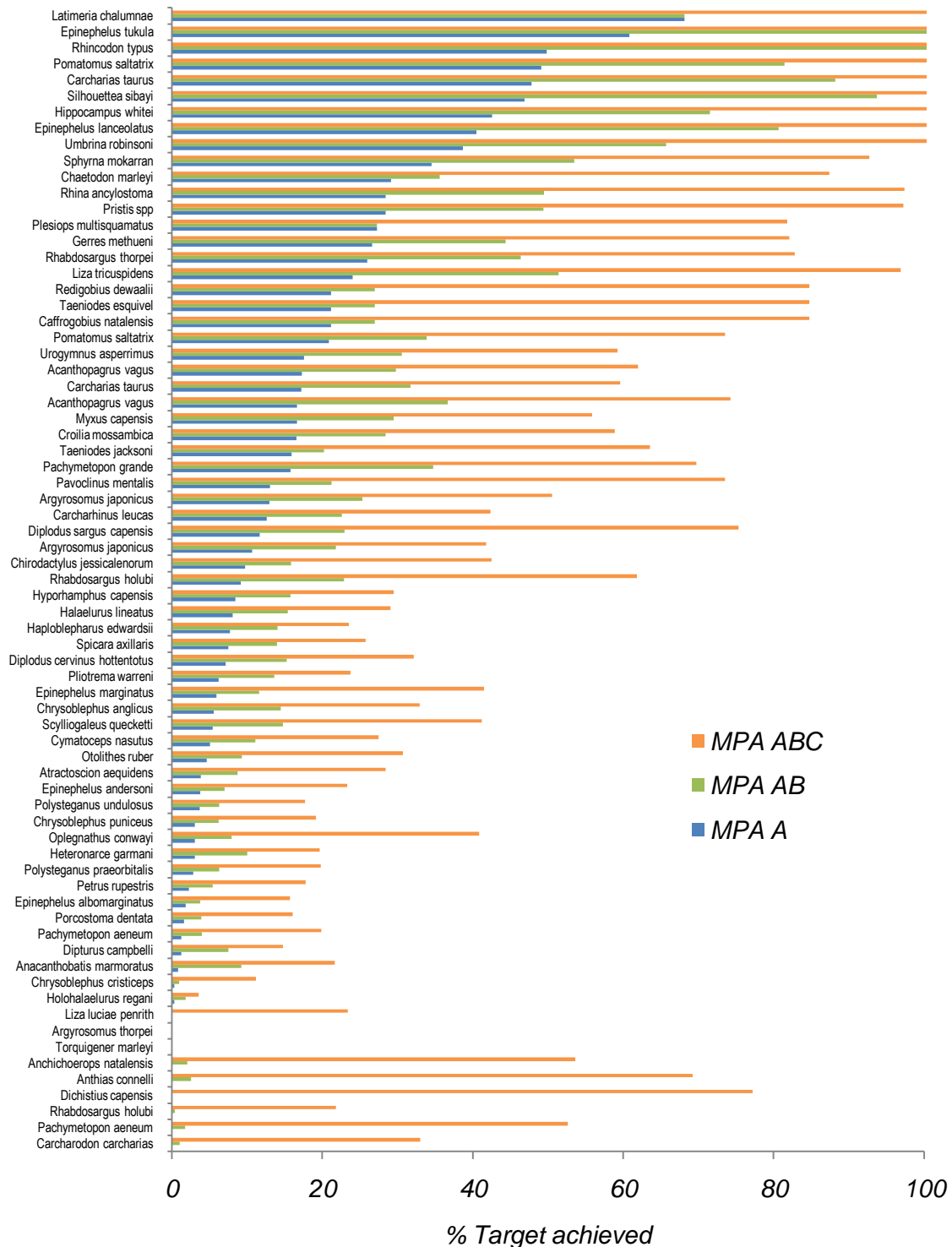


Figure 1. Summer data set target achievement under the three protection scenarios: (i) MPA zones A contribute to targets (MPA A); (ii) MPA zones A and B contribute to targets (MPA AB); and (iii) MPA zones A, B and C contribute to targets (MAP ABC). Species with

predominantly southerly distribution ranges had the lowest target achievement, while species with predominantly northerly distribution ranges had higher target achievement.

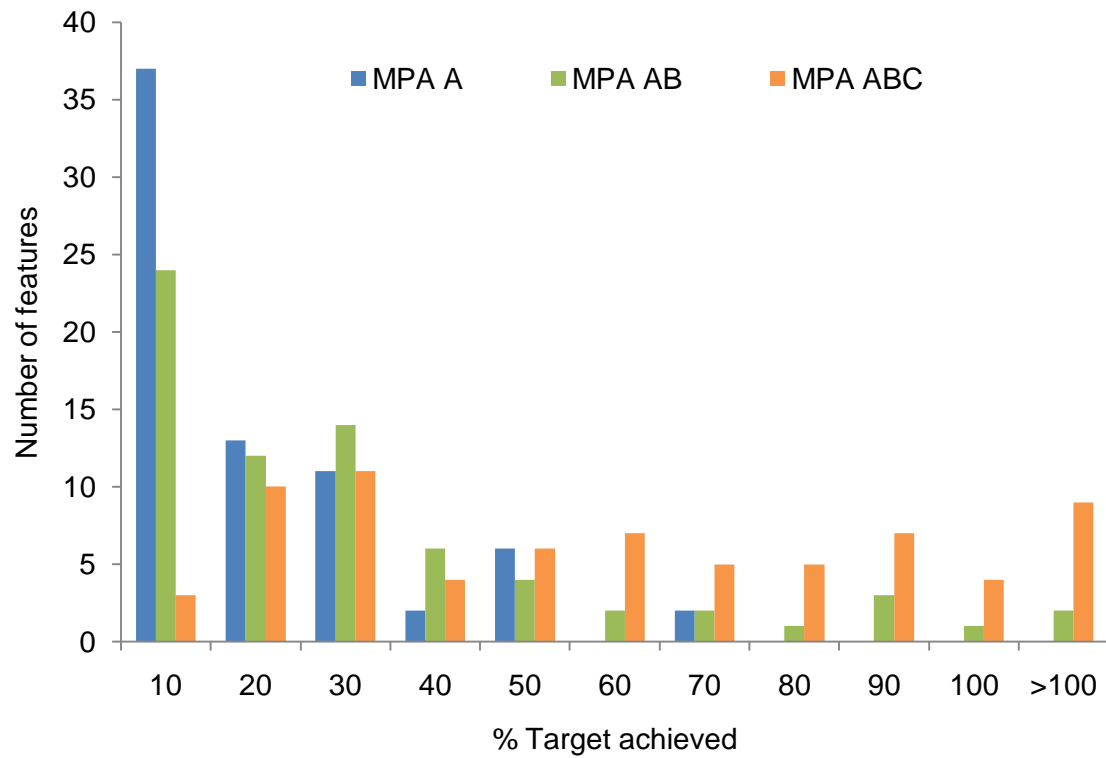


Figure 2. Number of summer features and percentage conservation target achieved by the three protection scenarios (i) to (iii) (see Figure 1).

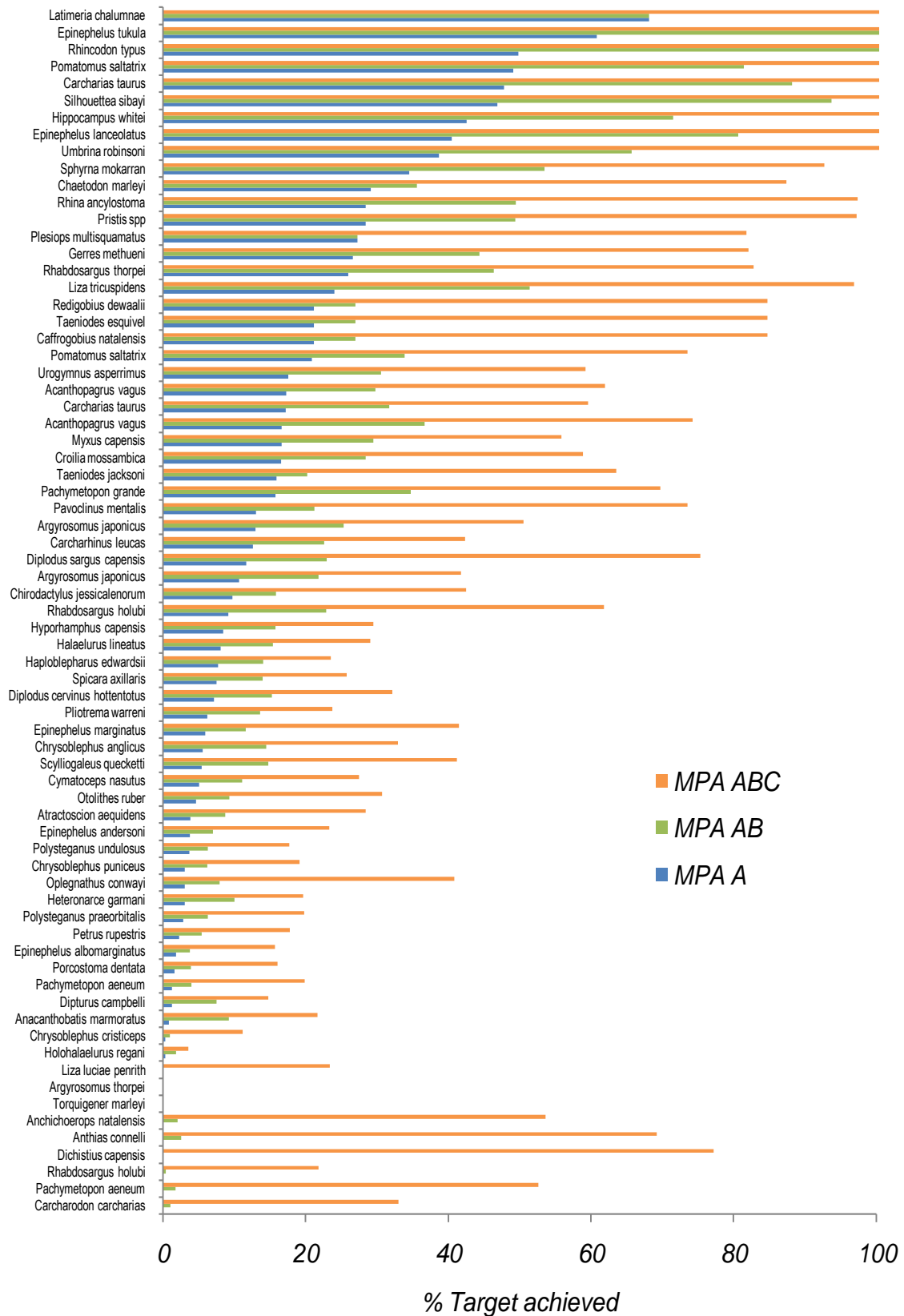


Figure 3. Winter data set target achievement under the three protection scenarios (i) to (iii) (see Figure 1).

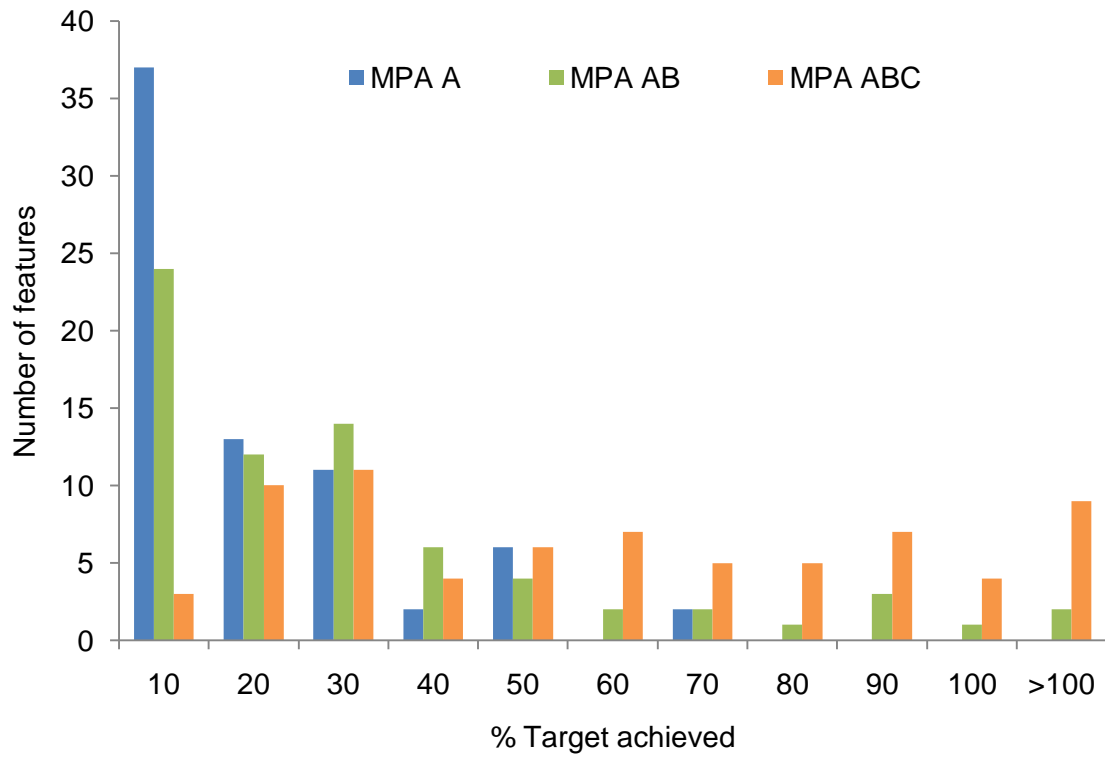


Figure 4. Number of winter features and percentage conservation target achieved by the three protection scenarios (i) to (iii) (see Figure 1).

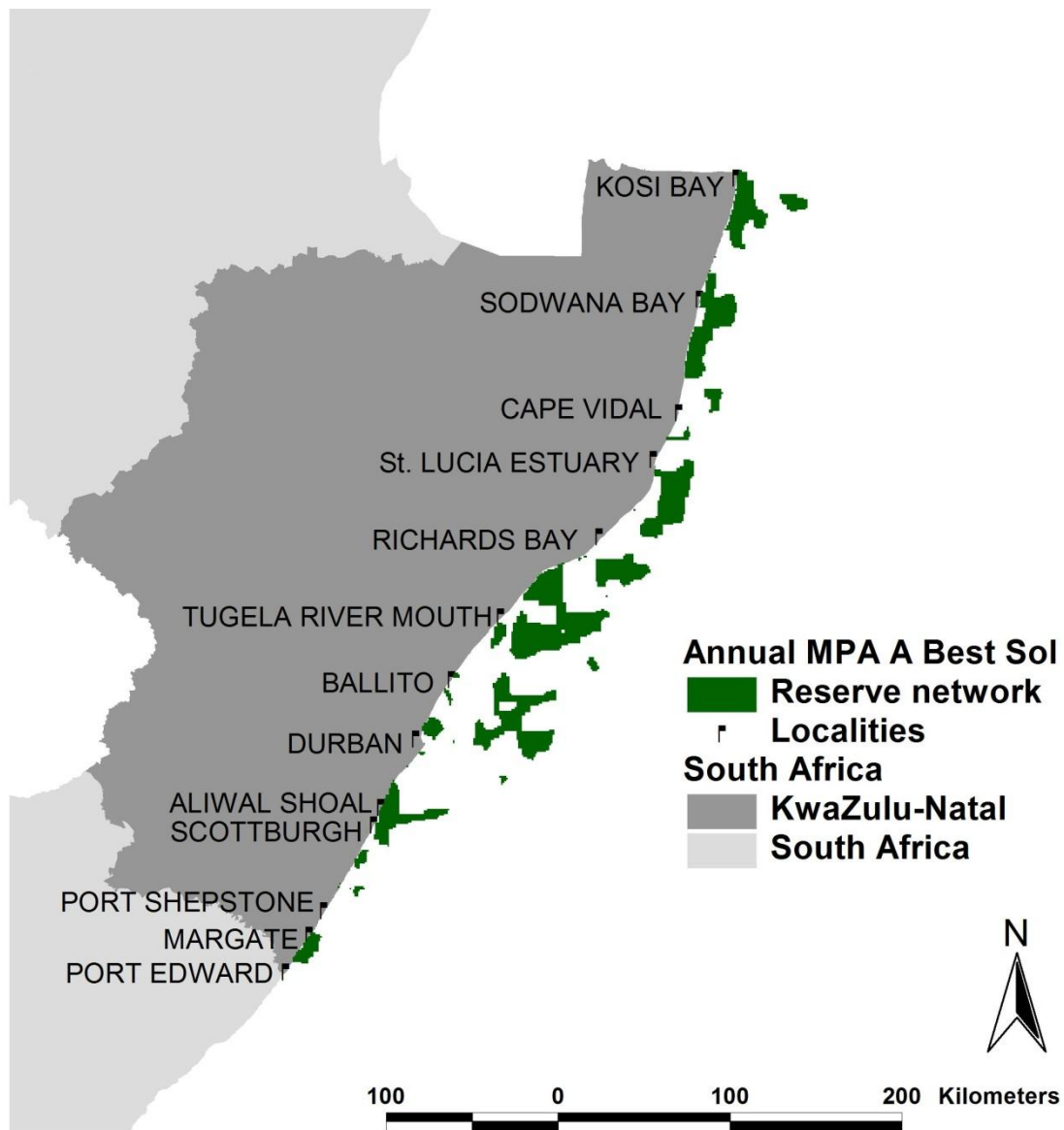


Figure 5. The best solution reserve network produced for the annual data set, using only MPA zones A as starting points. This network was used to calculate seasonal target achievement.

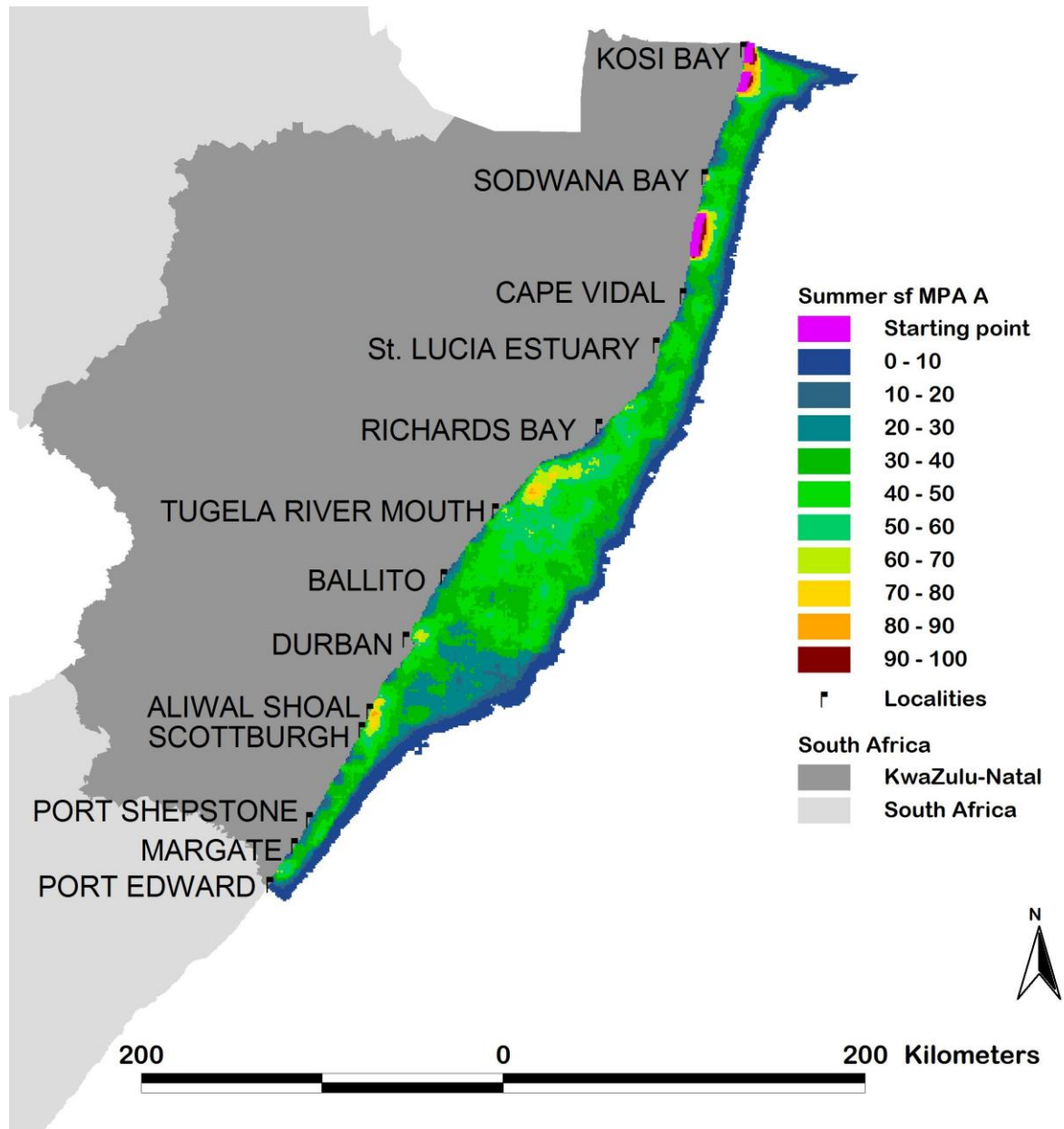


Figure 6. Summer selection frequency (sf) for scenario (i) where MPA zones A contribute to targets.



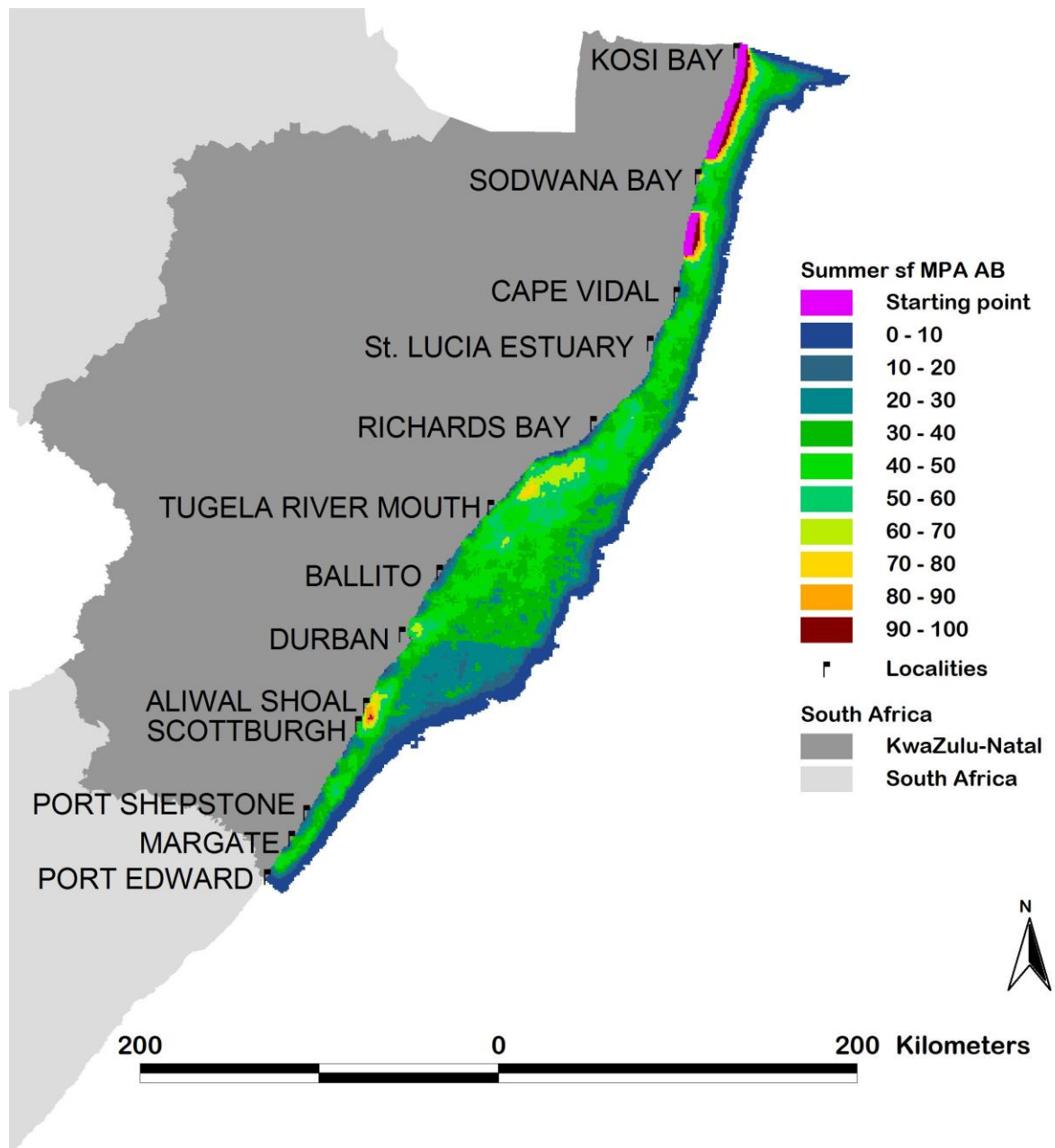


Figure 7. Summer selection frequency (sf) for scenario (ii) where MPA zones A and B contribute to targets.

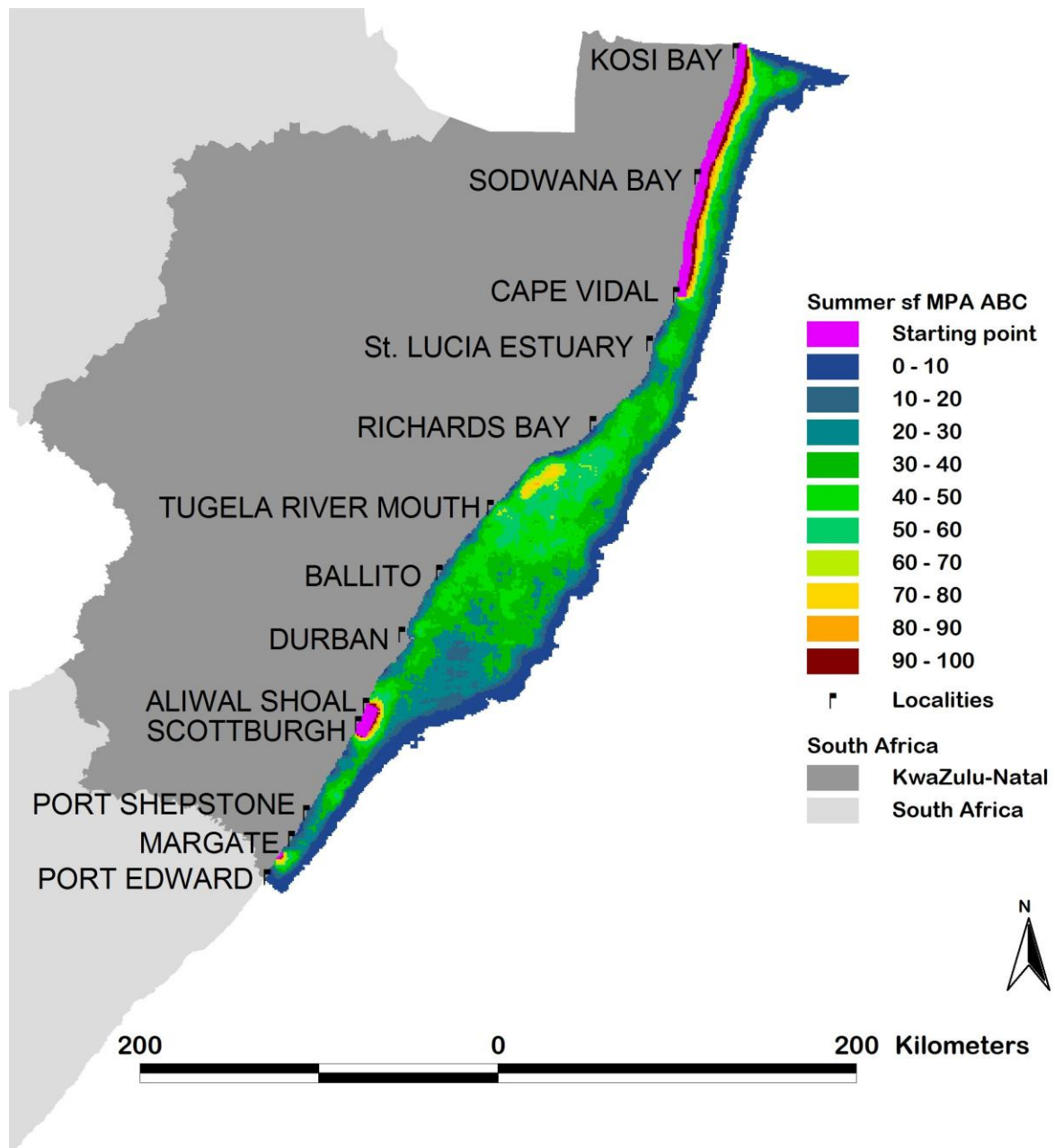


Figure 8. Summer selection frequency (sf) for scenario (iii) where MPA zones A, B and C contribute to targets.

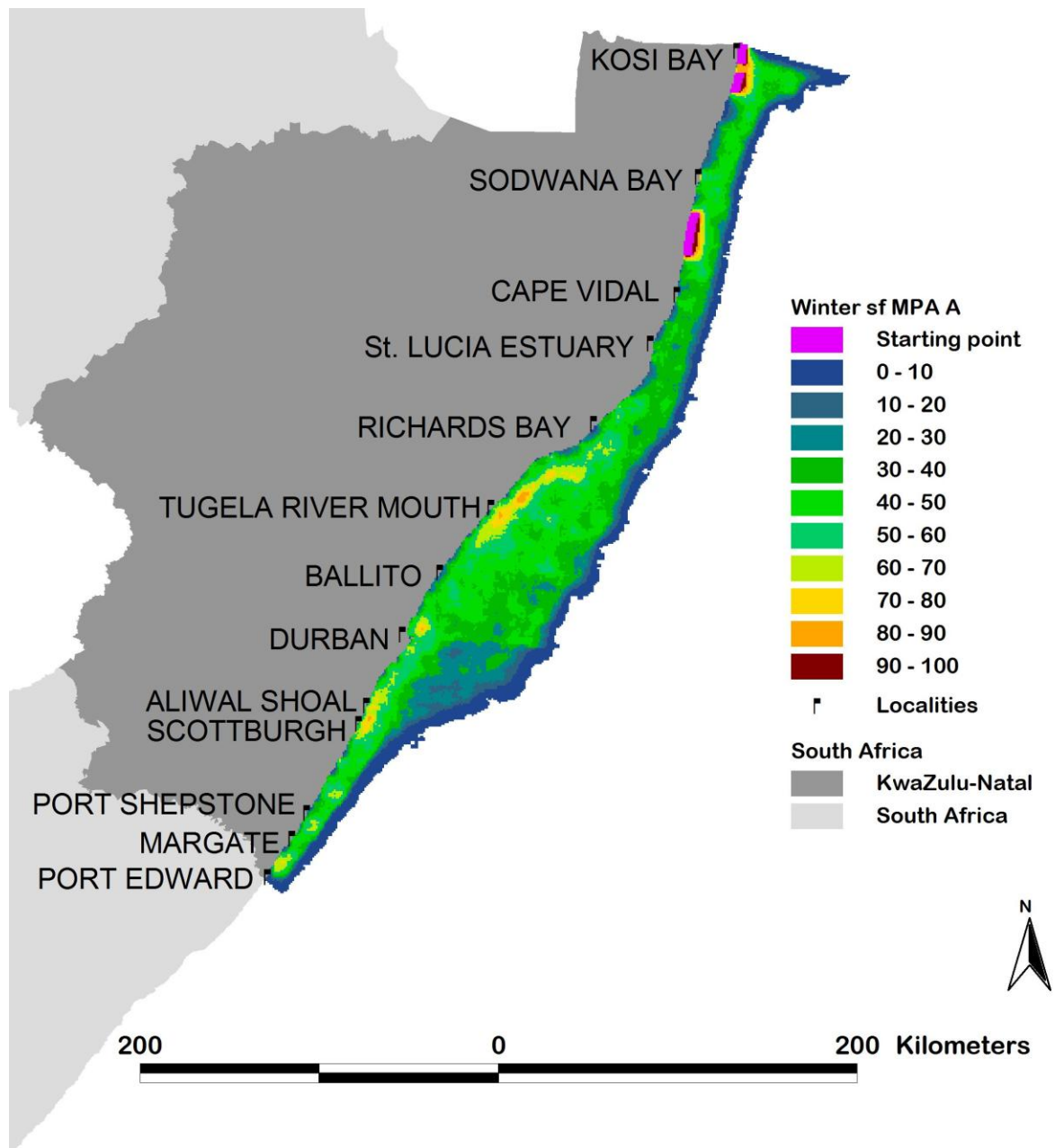


Figure 9. Winter selection frequency (sf) for scenario (i) where MPA zones A contribute to targets.

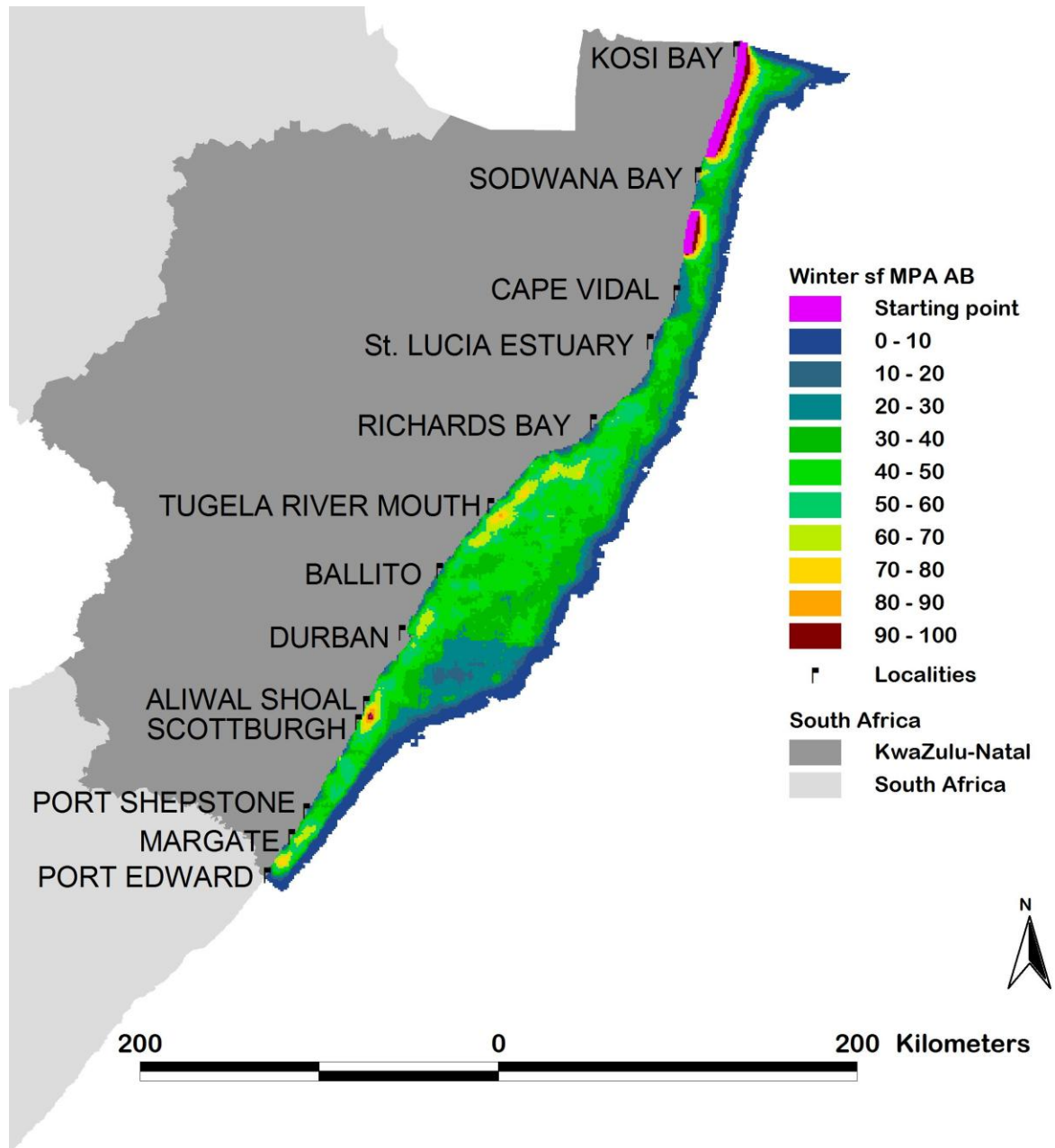


Figure 10. Winter selection frequency (sf) for scenario (ii) where MPA zones A and B contribute to targets.

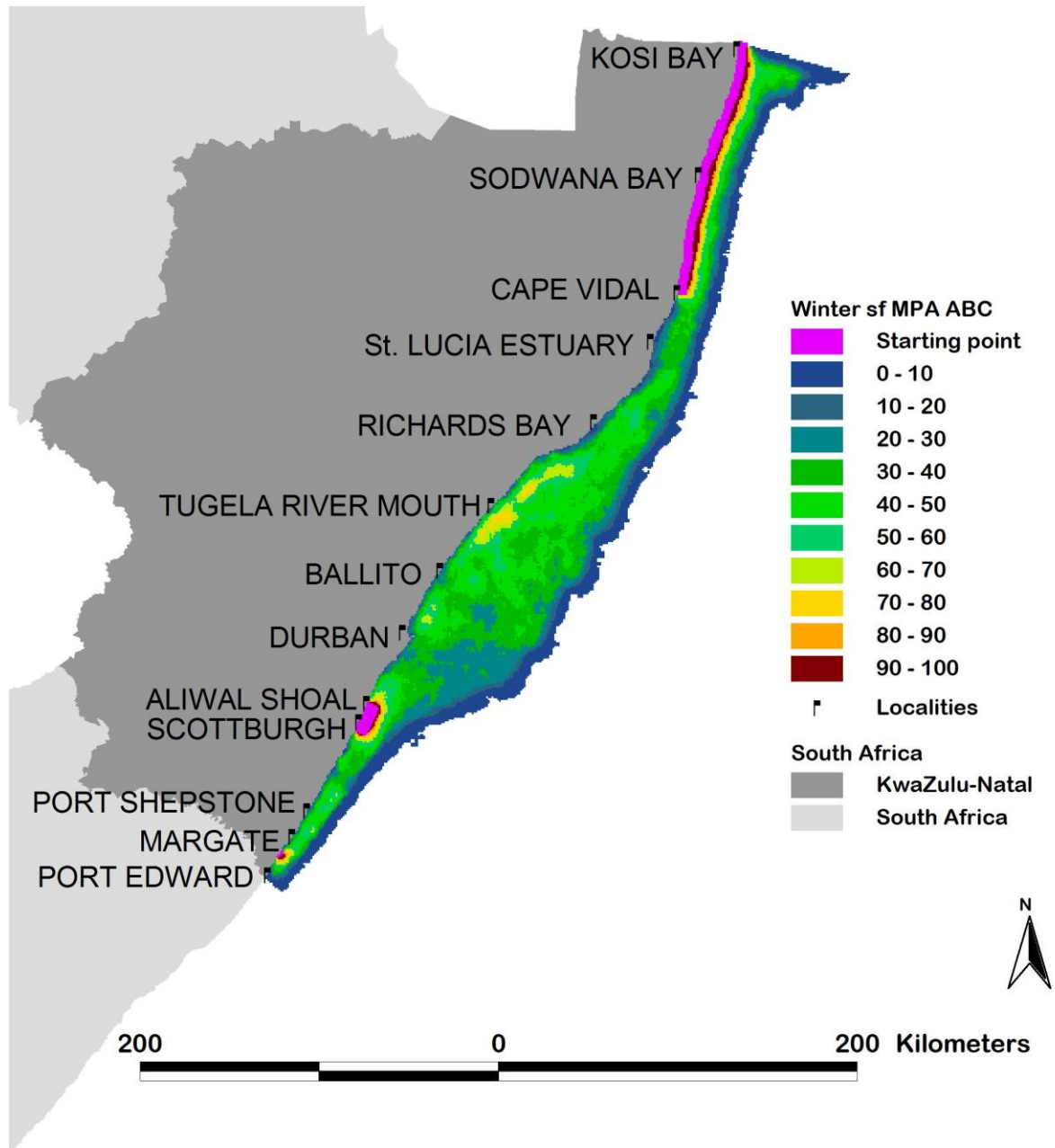


Figure 11. Winter selection frequency (sf) for scenario (iii) where MPA zones A, B and C contribute to targets.

**DEPARTMENT OF ACADEMIC ADMINISTRATION  
EXAMINATION SECTION – NORTH CAMPUS**

PO Box 77000  
Nelson Mandela Metropolitan University  
Port Elizabeth 6013  
Tel. +27 (0) 41 504 3206 / 504 3392  
Fax. +27 (0) 41 504 9206 / 504 3064



**Nelson Mandela  
Metropolitan  
University**

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**DECLARATION BY STUDENT**

**NAME:** Philip Haupt

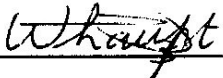
**STUDENT NUMBER:** 208093941

**QUALIFICATION:** MSc, Botany Department, Science Faculty

**TITLE:** The use of fish species in a marine conservation plan for  
KwaZulu-Natal

**DECLARATION:**

In accordance with Rule G4.6.3, I hereby declare that the above-mentioned treatise/dissertation/thesis is my own work and that it has not previously been submitted for assessment to another University or for another qualification.

**SIGNATURE:** 

**DATE:** 20 December 2010