TOWARDS AN ALTERNATIVE SPATIAL-BASED MANAGEMENT APPROACH FOR ESTUARINE FISHERIES IN SOUTH AFRICA, WITH A CASE STUDY FROM THE SUNDAYS ESTUARY

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ABSTRACT

Estuaries are productive habitats and biologically important ecosystems which serve as juvenile nursery areas and feeding grounds for adults from a host of fish species. They are, however, threatened habitats, increasingly exposed to human disturbance and exploitation. The stocks of several South African estuary-dependent linefish species are now considered as either overexploited or collapsed. It is clear that their dependence on estuaries would warrant the inclusion of these ecosystems into marine reserve planning exercises. Since traditional management strategies (e.g. bag and size limit restrictions) have proven ineffective for estuarine fisheries, there is a need for alternative management measures, such as spatial and temporal restrictions, to ensure increased survival of juveniles and recovery of adult breeding populations. This thesis explored the potential for an ecosystem-based approach through the application of a rapid sustainability assessment technique, and a spatial-based management approach for an important fishery species, using conservation planning software.

The Sundays Estuary, Eastern Cape, South Africa falls within the footprint of the Addo Elephant National Park, with a proposed expansion to include a marine protected area (MPA). However the estuaries resources were not considered during the planning of the proposed MPA. This study conducted an indicator-based sustainability assessment based on the principles of sustainable development. The results showed that present levels of exploitation, due to non-compliance and a lack of law enforcement are unsustainable. The sustainability of the Sundays Estuary had a low overall sustainability score of only 23.8%. With limited enforcement of estuarine fisheries regulations in South Africa, alternative management measures such as spatial regulations may provide a viable option forward.

The sustainability of fishery resources depends on the comprehensive understanding of the fishery resource. Acoustic telemetry is a technique that has been widely adopted to infer habitat and area use patterns of fish species. The second component of this study made use of high resolution telemetry data collected on juvenile dusky kob *Argyrosomus japonicus* movements within the Sundays Estuary to conduct a scenario-based approach using Marxan conservation planning software. The best solution given by Marxan, in the form of a protected area for the conservation of juvenile *A. japonicus* in the Sundays Estuary was identified in the middle (starting 7km from the mouth) to the upper reaches (approximately 16km from the mouth) of the estuary, ultimately providing protection to tagged individuals for 61% of their time in the estuary. Although Marxan presented a best solution, the Sundays Estuary's small size and shape, and minimal features used, was too simplistic to be included into a Marxan analysis. However, new methods and tools to analyse and plan spatial-based management options at this scale are currently being developed.

Using the Sundays Estuary as a case study, a decision tree was then developed as a protocol to assist management address the challenges of effective estuarine management depending on the unique biological and socio-economic characteristics of individual estuaries in South Africa.

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CHAPTER 1: GENERAL INTRODUCTION

1.1 BACKGROUND

Globally, fishing pressure has increased over the past century resulting from an increase in human population size and dependency on marine resources (Powels *et al*. 2000, Halpern *et al.* 2007, Bennett 2012). Technological advances in fishing gear (Roberts 2007), ineffective management regulations and non-compliance (Worm *et al.* 2009) have, in addition to increased fishing pressures, negatively impacted fish stocks. Consequently more than half the world's fish stocks are now either fully exploited and nearly a third overexploited (Worm *et al.* 2009, FAO 2010).

In South Africa, the harvesting of living resources started in the 17th century, with the only limit on harvesting being the availability of the resource itself (Bennett 2012). However, in the middle of the 20th century, recreational and commercial fisheries became increasingly important. Linefishing is defined as that "*activity where fish are harvested using a hook and line but excludes the use of set pelagic or demersal longline, which are managed as separate fisheries*" (Mann 2013), and the species regarded as linefish are of considerable social, economic and recreational value (Mann 2000). The South African linefishery consists of commercial, recreational and subsistence sectors which exploit over 200 fish species collectively (DAFF 2013).

Recreational angling in South Africa is a very popular activity which has seen large increases in the number of participants, fishing techniques and equipment over the past few decades (Brouwer 1997, Mann 2013). In 1995, it was estimated that 500 000 participants were active along the South African coastline. This number increased to approximately 900 000 in 2007 (Leibold & van Zyl 2008). The marine recreational fishery in South Africa is separated into four divisions: shore angling, deep sea angling, spearfishing and estuarine angling (van der Elst 1989). The infrastructure associated with this fishery in terms of tourism, boats, tackle and the bait industry makes it extremely valuable. It has been estimated that the total economic impact of the fishery is in excess of ZAR9 billion per annum (Leibold & van Zyl 2008).

In South Africa, estuarine angling is a very popular activity which has seen large increases in the number of participants and fishing techniques and equipment over the past few decades (Brouwer 1997, Mann 2013).

Estuarine angling has increased in recent years by a shift in effort from the coastal zone to estuaries following the ban on off-road vehicles on South African beaches in 2001(regulations promulgated in terms of the National Environmental Management Act No. 107 of 1998 [Government Gazette No. 22960]).The result was a shift in the distribution of fishing effort from beaches to estuaries, which led to an increase in fishing effort of juveniles in their estuarine nursery habitats and hence negative consequences for estuary-dependent fishery species (Potts *et al*. 2005). Estuarine fisheries have witnessed severe changes in size and catch composition, as well as a decline in the catch-per-unit-effort (CPUE) of many targeted estuary-dependent fishery species, namely dusky kob *Argyrosomus japonicus*, white steenbras *Lithognathus lithognathus*, spotted grunter *Pomadasys commersonnii* and Leervis *Lichia amia (*Van der Elst & Adkin 1991, Baird *et al*. 1996, Whitfield & Cowley 2010, Cowley *et al*. 2013).

Estuaries are critical components of coastal zones (Constanza *et al.* 1997, Turpie & Gross 2015), as they serve asessential nursery habitats for a number of recreational, subsistence and commercial fishery species (Beck *et al*. 2001, Cowley *et al.* 2011). Despite the economic and social benefits that estuaries provide, there are increasing pressure on estuaries through direct use of resources and developments on their margins and within their catchments, which pose a threat to their value (Cowley *et al*. 2013).

The Minister of Environmental Affairs announced in 2000 that there was a crisis in the South African linefishery. This led to a reduction of daily bag limits for many species and reduced effort in the offshore boat-based commercial fishery (regulations promulgated in terms of the National Environmental Act No. 107 of 1998 [Government Gazette No. 19519]). The current management regulations including size and bag limits that have been introduced to conserve linefish stocks have been unsuccessful thus far and current management regulations for estuarine fisheries are inadequate (Taylor *et al*. 2010, Whitfield & Cowley 2010, Mann 2013, Cowley *et al.* 2013). This is attributed to the inadequacy of several factors, including limited law enforcement, lack of compliance, no fishery monitoring, ill-informed users and no public awareness campaigns, all of which threaten the sustainability of estuarine fishery resources (Griffiths 2000, Mann 2013).

1.2 MANAGEMENT OF MARINE AND COASTAL FISHERIES

Traditionally, fisheries management has focused on a single sector approach using stock assessments; however, this approach is often ineffective because it ignores ecosystem components and biological and human interaction (Pikitch *et al.* 2004). There is now a growing awareness of the cumulative effects of anthropogenic activities on marine ecosystems (Halpern *et al.* 2008, Douvere 2010), which led to a paradigm shift in fisheries management in the 1900s. This has been referred to as ecosystem based management (EBM) and takes ecosystems goods and services into consideration during the assessment and management of marine ecosystems (FAO 2003). EBM addresses three broad domains relating to resource management, in order to ensure sustainable resource utilisation; these include environmental/biological (the resource), social (those using, or relying on the resource) and institutional (decision makers, management authorities or the associated legislation) (Pajak 2000).

Spatial trends in resource use and of the resource can assist in developing spatial management plans and is a central component of EBM. Marine spatial planning is one of the key tools that can be used to facilitate the implementation of EBM, and incorporates a full range of anthropogenic drivers on the marine environment (Chalmers 2012).

In the past, management of the South African linefishery had not been well recognised (Griffiths 2000). The Marine Living Resources Act (regulations promulgated in terms of the Marine Living Resource Act No. 18 of 1998 [Government Gazette No. 27453])) called for the establishment of operational management plans for linefish species in South Africa (Mann 2013). To address these needs, a Linefish Management protocol (LMP) (Griffiths *et al*. 1999) was implemented in 1999 to provide a standardised method for assessing the status of linefish stocks, through the use of per-recruit analyses and age-structured production modelling (Griffiths 1999, Bennett 2012). Within the LMP, management plans for all linefish species were developed, in which the species-specific type of data, stock assessment analysis and quantifiable biological reference points were used (Bennett 2012). Certain regulations were put in place to manage fishing pressure on resources. Some of which included size and bag limits, closed areas and seasons through the establishment of Marine Protected Areas (MPAs). However, the LMP failed to advocate an Ecosystem-based approach to the linefish management regulations established.

The declaration of MPAs is facilitated by the Protected Areas Act (Act no. 57 of 2003) and the concept of MPAs is well established in South Africa (Attwood et al. 2007, Whitfield & Cowley 2010). Approximately 23% of the coastline is designated as MPAs, with 9% being fully protected in no-take zones (Lombard *et al.* 2004). However, a detailed assessment of the MPA network revealed that because of the *ad hoc* designation of MPAs in the past, different habitats and biodiversity were poorly represented (Attwood *et al.* 1997, Lombard *et al*. 2004, Chalmers 2012). As part of the National Protected Areas Expansion Strategy (NPAES), South Africa has aimed to integrate terrestrial, riverine, estuarine, inshore and offshore protected areas through the development of MPA networks (DEAT 2010).

If MPAs are found on sustainable EBM frameworks, they can provide possible alternative management solutions to the failing conventional measures in South Africa.

1.3 MANAGEMENT OF SOUTH AFRICAN ESTUARINE FISHERIES

South African estuaries are unique in their biological and physiological characteristics. Additionally, the resource use activities, reasons for resource use and local socio-economic situations differ for each estuary (Cowley *et al*. 2013). As a result, estuarine management should be estuary-specific to account for the unique characteristics of each estuary. The current level of protection afforded to South African estuaries and estuarine fisheries is both regionally and nationally poor. Owing to the lack of law enforcement and non-compliance of fishery participants, current management regulations have been ineffective (Whitfield & Cowley 2010). Consequently, there is an urgent need to improve the unsuccessful management regulations of the heavily targeted linefish species, which are dependent on these systems (Chalmers 2012). However, limited resources are dedicated to enforcing these regulations, and as a result, alternative management measures such as spatial and temporal regulations, which require fewer resources to enforce, may be a viable option (Childs 2013).

1.4 MOTIVATION AND APPROACH FOR THE CURRENT STUDY

1.4.1 Case Study: The Sundays Estuary

The South African National Parks (SANP) authority is currently establishing a MPA that borders the Addo Elephant National Park (AENP) in the Eastern Cape, South Africa, which includes objectives to contribute to the protection of vulnerable linefish species, like *A. japonicus*, *P. commersonnii* and *L. lithognathus* (Chalmers 2012, Childs 2013). The National Park Expansion Plan which proposed the MPA in Algoa Bay would be the first in South Africa to incorporate a bay environment, exposed rocky headlands and offshore islands. The proposed MPA would be zoned into control use and restricted zones as indicated in Figure 1.1. Whilst a large multiple-use MPA adjoining the terrestrial component was proposed in the mid-1990s, the lack of spatial data to quantify fishery costs and conservation benefits led to wide scale public opposition, and the process was halted until the study of Chalmers (2012).

Although the importance of estuaries and their protection was recognised in the systematic conservation plan conducted by Chalmers (2012), recommendations for the Sundays Estuary were only extended to the seaward side of the estuary mouth within the proposed footprint. The Sundays Estuary would only include a 2km buffer area around the estuary mouth (Chalmers 2012). SANP is currently working towards the inclusion of the Sundays Estuary in the proposed MPA following the proclamation of the Integrated Coastal Management Act (regulations promulgated in terms of the National Environmental Act No. 24 of 2008 [Government Gazette No. 31884]). Although SANP stated that they will restart the declaration process, expanding on the current draft plan (Bezuidenhout *et al.* 2011, IECM 2011, AENP 2015), to date, no protected area has been established and there is little information to guide the sustainable utilisation of estuarine fishery resources.

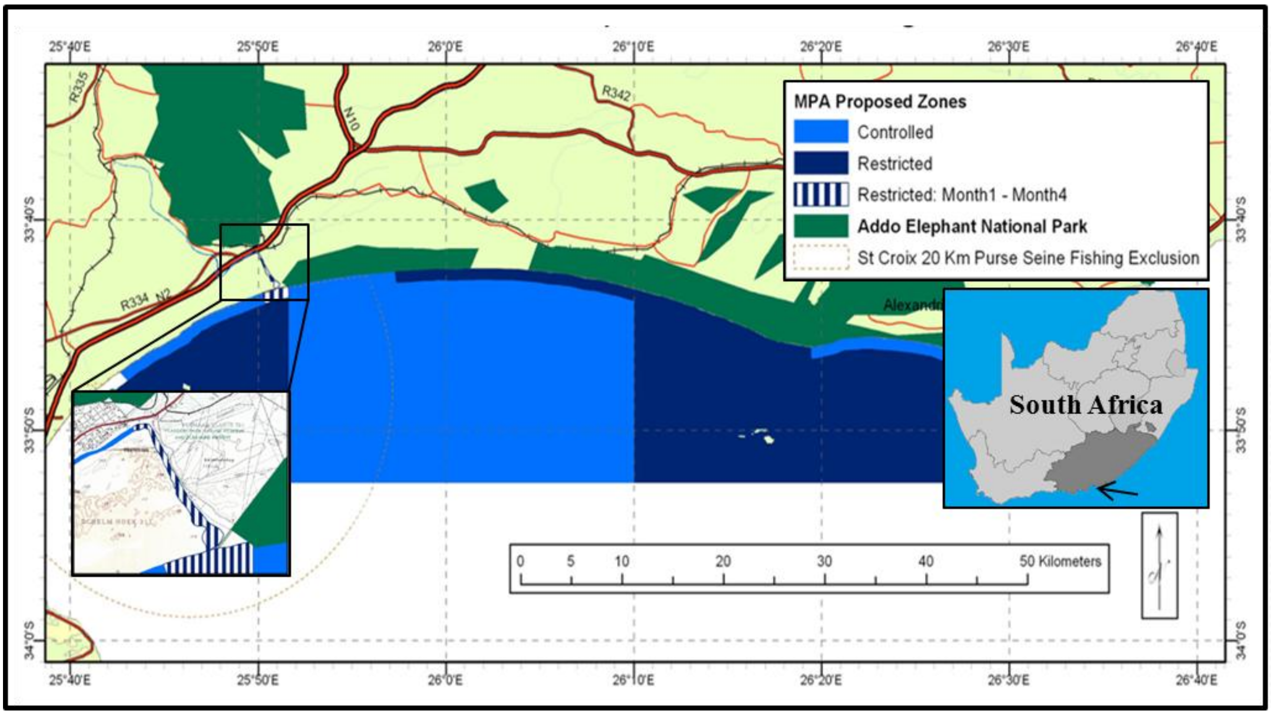

Figure 1.1: Proposed Addo Elephant National Park MPA, showing the proposed zoning into restricted (no-take) and controlled use zones (adapted from Oosthuizen *et al*. 2011).

The sustainability of fishery resources depends on the comprehensive understanding of the status of the fishery. Owing to this, Cowley *et al*. (2013) conducted an investigation on the fishery resource utilisation on the Sundays Estuary, which has confirmed the ineffectiveness of current management regulations and the need for alternative (spatial) measures.Effective spatial management strategies not only require socio-economic information about resource users, but also a better understanding of estuarine dependency, habitat use patterns and connectivity of fish species (Whitfield & Cowley 2010). An understanding of the spatial and temporal movements of exploited linefish species in estuaries is fundamental to the design of spatial management strategies (Tremain *et al*. 2004, Childs *et al*. 2008). Acoustic telemetry techniques have been widely used to investigate the movement behaviour of other estuary-dependent fish species (Childs *et al*. 2008, McCord and Lamberth 2009, Bennett *et al.* 2012).

Movement studies can reveal the role that estuaries play in the life-history of a species and identify ecologically important areas for conservation (Egli & Babcock 2004). The fine-scale fish movement data provided by acoustic telemetry is essential when identifying suitable management measures such as estuarine protected areas (Jones 2005, Bennett *et al.* 2011).

Successful marine spatial planning requires inputs from a variety of sources, including conservation biologists, stakeholders, planners and policy makers (Moilanen *et al.* 2009, Levin *et al*. 2015). These planning processes have to be flexible and able to react to changes in decision makers approaches, shifts in development, and to cope with the complexity and diversity of dynamic systems (Levin *et al*. 2015). Decision support tools such as Marxan conservation planning software, provides transparent and quantitative methods to evaluate different conservation plans and networks (Ball *et al.* 2009).

To date, no study has explored the integration of acoustic telemetry and spatial prioritisation software as an alternative spatial-based management approach for a single estuary-dependent fish species. This thesis will do so, using high resolution movement data of dusky kob (*A. japonicus*) and spatial distribution of resource users to improve the sustainability of the Sundays estuarine fishery.

1.4.2 Aims and objectives

This thesis has been divided into five chapters (Figure 1.2). The overall aim of this thesis is to investigate the appropriateness of a spatial-based management approach for estuarine fisheries, through Marxan conservation planning software, using the sustainability of the Sundays estuarine fishery as a case study.

This study is the first of its kind in South Africa to prioritise management of a single estuary-dependent fishery species using EBM principles. The aim of this study was to assess the current sustainability of the Sundays estuarine linefishery and evaluate the potential effectiveness of an alternative spatial-based management approach. A summary of the process involved in assessing the alternative management plan (spatial-based management) are shown in Figure 1.2

To achieve the objectives of this study, data obtained from a roving creel survey conducted by Cowley *et al.* (2013) on the resource use of the Sundays estuarine fishery was required (Figure 1.2 A). Key issues associated with South African estuarine fisheries were identified; detailed base-line information on the composition, relative abundance and size structure of targeted fish species were presented; and fine scale spatial and temporal trends in fishing activities in the Sundays Estuary were recognised in Chapter 2 (Figure 1.2 A). A sustainability assessment using rapid appraisal techniques, which highlighted fundamental issues within the Sundays estuarine fishery, was conducted in Chapter 3 (Figure 1.2 B & C), and a systematic conservation planning exercise was conducted in Chapter 4 to identify priority areas for conservation of juvenile *A. japonicus*, and evaluate the socio-economic costs of fishers thereof (Figure 1.2 E, F & G).

Figure 1.2: Flow diagram showing the steps taken to conduct a spatial-based management plan.

CHAPTER 2: REVIEW OF TARGETED ESTUARY-DEPENDENT FISHERY SPECIES, AND A CASE STUDY FROM THE SUNDAYS ESTUARINE FISHERY

2.1 ESTUARY-DEPENDENT FISHERY SPECIES

2.1.1 Status and ecology of targeted estuary-dependent fishery species:

There are approximately 160 fish species occupying estuaries in South Africa. Of those, 80 species are captured within South African fisheries, consisting predominantly of Sparidae and Muglidae families (Lamberth & Turpie 2001).

Whitfield (1994) divided estuarine fish into five categories according to their estuarine dependence.Estuary-dependent fish species utilise estuaries as nursery areas and all have a category rating of IIa. They are defined as those fish that have an obligatory estuary-dependent juvenile stage, and as those species whose populations would be adversely affected by the loss of estuarine habitats (Whitfield 1994). Many estuarine-dependent linefish species form significant proportions of the catches in a number of estuarine fisheries, including over-exploited and collapsed species such as; Dusky kob *A. japonicus,* spotted grunter *P. commersonnii,* white steenbras *L. lithognathus,* and leervis *L. amia* (Kyle 1988, van der Elst & Adkin 1991, Pradervand & Baird 2002, Cowley *et al*. 2013). Although many of these threats are environmental, overfishing is the single biggest threat (Whitfield & Cowley 2010). The status of a stock is based on its current size as a percentage of pristine stock size or spawner biomass (Lamberth & Turpie 2001).

Information to score stock status of the four species appropriately was collected by Lamberth & Joubert (2014) who developed certain criteria by which these and other species were scored. This information has been adapted to give detailed reasoning for each of their current stock statuses and is represented in Table 2.1.

Table 2.1: Stock status in terms of abundance trend (A), vulnerability (V), range (R), exploitation level (E) and knowledge (K) of the most targeted estuary-dependent fishery species in South Africa (adapted from Lamberth & Turpie 2001, and Lamberth & Joubert 2014).

Species	Category	Conservation Importance (%)				
		A	V*	R**	E***	K****
A. japonicus	IIa	~4 (collapsed)	100	40	100	86
P. commersonnii	IIa	40 (over-exploited)	100	0	100	57
L. lithognathus	IIa	6 (collapsed)	100	40	100	50
L. amia	IIa	50 (optimally exploited)	90	0	75	64

**Vulnerability*: Determined using eight life-history traits (estuary dependence, sex changes, spawning migrations, predictable aggregations, high age at maturity, longevity, residency and high catchability). Species displaying zero traits scored 0, those with one, two or three characteristics scored 70, 80 or 90 respectively, and those displaying four or more of these characteristics scored 100 (see Lamberth and Joubert 1999 for rationale).
***Range:* species' relative occurrence throughout its range was scored qualitatively by experts from the four different zones (i.e.: The number of zones it occurred in - zones are West Coast, South Coast, East Coast, KwaZulu-Natal) (Lamberth & Joubert 2014).
****Level of exploitation*. Species that are heavily exploited throughout its range scored 100, medium = 50, and low = 0 (Based on expert opinion, Lamberth & Joubert 2014).
*****Level of knowledge*: The 14 factors for scoring the level of knowledge for each species on a scale of 0-100 in van der Elst & Adkin (1999) and Mann (2000).

The results from long-term CPUE data show that the stocks of many estuary-dependent fishery species are either over-exploited or collapsed (Griffiths 2000). The abundance of estuary-dependent fish species have been scored and range between 0 (collapsed stock status) and 100 (underexploited stock status) (Lamberth & Joubert 2014). This was based on the percentage of pristine spawner biomass, breeding stock remaining and/or ratios of present versus historical CPUE and catch composition.

Recently, the stock abundance of linefishery species has been re-assessed (Winker *et al*. 2015). Fish exploitation status's were based on the following SB/R statistics; optimally exploited: if the SB/R statistic is above 40% of unexploited, overfished: if it is between 40 and 25%, collapsed: if it is below 25%, and critical: if it is below 5%. From this assessment, it was concluded that *P. commersonnii* has collapsed and the SB/R of *A. japonicus* is considered to be critical. Bennett (2012) confirmed that the current SB/R ratio of *L. lithognathus* was estimated at 6% of pristine (Bennett 1993), as early as 1993. Therefore, according to the LMP, the stock is considered collapsed (Griffiths *et al*. 1999, Bennett 2012). It has been suggested that the spawner-biomass per recruit of *L. amia* was 14% of pristine and thus considered to be collapsed according to the South African Marine Linefish Management Protocol (Griffiths *et al*. 1999, Smith 2008, Dunlop *et al*. 2014).

Whilst the primary threat to these fish species within estuaries is overexploitation (Griffiths 1997), aspects of a species life-history such as sex changes, spawning migrations, predictable aggregations, spawning migrations, high age at maturity, longevity, residency and high catchability (Table 2.2) are also major factors rendering a species vulnerable to overexploitation and threaten their future existence (Musick 1999, Fisk & Miller 2005, Lamberth & Turpie 2002, Griffiths 2010, Childs 2011).

The American Fisheries Society (AFS) proposed the use of a set of risk criteria that assessed the productivity of a species (Musick 1999). It was recognised that marine fish may be at risk of extinction (Hunts-man 1994, Musick 1999), following this, the IUCN held a workshop in 1996 that aimed to apply a quantitative risk criteria to marine fish to assess their risk of extinction and were based on a set of criteria. Species or populations can be classified into categories of high, medium, low and very low threat (Musick 1999). According to Musick (1999), a fish with high fecundity (>104), but late maturity (5-10 year), and long life span (>30 year) and a low estimated natural mortality (<0.1) would be classified under the 'Very Low Productivity' category. These risk criteria have recently been assessed for *A. japonicus* (Childs 2011). Like most sciaenids, *A. japonicus* have life-history traits that render them prone to overexploitation (Griffiths 1996), some of which include age and length at 50% maturity, longevity and concentrated spawning aggregations (Table 2.2). The large size at sexual maturity and therefore extended juvenile phase, along with estuarine residency contribute to the collapse of stock status and population decline. Following the set of risk criteria proposed by Musick (1999), Childs (2011) found that *A. japonicus* scored 'very low" in terms of resilience to fishing pressure, with a population decline threshold of 0.7 (refer to Musick 1999).

In this chapter, the same method was applied to other targeted estuary-dependent fish species. Since there is a lack of data on 'r' for *P. commersonnii, L. lithognathus* and *L. amia*, the age at maturity, *P. commersonnii*: 3 years (Wallace 1975); *L. lithognathus*: 5-6 years (van de Elst 1993) and *L. amia*: 4 years (Mann 2013) was used to calculate their population threshold decline (productivity). The results of which found that *P. commersonnii* and *L. amia* scored moderate (0.95) and *L. lithognathus* had a lower score (0.85).

Table 2.2: Vulnerable life history traits of the four most targeted estuary-dependent linefish species (Wallace 1975, Bennett 1993, Van der Elst *et al.* 1993, Griffiths 1996, Mann 2013, Childs 2013, Cowley *et al.* 2008, Childs *et al*. 2008, Cowley *et al.* 2013, Bennett 2012, SAIAB *unpubl.* data).

Life-history Traits	**Species**			
	A. japonicus	***P. commersonnii***	***L. lithognathus***	***L. amia***
Sex Changes	-	-	-	-
Spawning migrations	Oct-Jan : EC and WC Aug-Nov: KZN	Aug-Dec in KZN	Late winter, Jul-Aug	Sep-Nov in KZN
Predictable aggregations	✓	✓	✓	✓
Age at maturity	~6yrs	2.5-3yrs	~6yrs	~4yrs
Maximum size	2005 mm	910 mm	1376 mm	1800 mm
Longevity	42yrs	14-19yrs	25-30yrs	10yrs
Estimated natural mortality	$0.1yr^{-1}$	$0.28yr^{-1}$	$0.2yr^{-1}$	$0.33yr^{-1}$
Vulnerability	100	100	100	90
Catchability	High	High	High	Med-High
Residency	Juveniles: Resident to estuaries and coastal zones. Low levels of connectivity among estuaries. Adults: (> 1000 mm TL): Proportion migratory undertaking large scale coastal movements.	Juveniles: Resident to estuaries for the first 3 years. Low levels of connectivity among estuaries. Adults: (> 350 mm TL): Proportion migratory undertaking large scale coastal movements.	Juveniles: Highly resident to estuaries for the first 4 years. Negligible levels of connectivity among estuaries. Adults: (> 600 mm TL): Proportion migratory undertaking large scale coastal movements.	Juveniles: Resident to estuaries for first 2 years. High levels of connectivity among estuaries. Adults (> 800 mm TL): Largely migratory undertaking large scale coastal movements.

Following the set of criteria used by Musick (1999) and given the high risk of threat that these species face in estuaries, shown in the literature, there is a need to use this information to assess not only the sustainability of these resources within estuarine fisheries, but how this information can be used for future management decisions.

2.2 CASE STUDY: THE SUNDAYS ESTUARINE FISHERY

2.2.1 Study site

The Sundays Estuary, which enters Algoa Bay at 33°43′ S; 25°51′ E and is situated 30 km North-East of Port Elizabeth in the Eastern Cape, represents one of 44 permanently open estuaries in South Africa (Turpie & Clark 2007). It is a channel-like system, approximately 21 km in length, 50 m wide for the majority of its width and an average depth of 2.5 m (Marais 1981). It is characterised by steep banks with limited marginal vegetation. There is an absence of salt marshes or large mud flats, with the exception of a well-developed sandy flood tide delta at the mouth (Mackay & Schumann 1990). Submerged macrophytes occur at the head of the estuary and the upper reaches, whilst benthic algae dominate the middle reaches and a small bed of *Zostera capensis* sometimes establishes itself near the mouth (Harrison & Whitfield 1990).

The estuary has a large catchment (20 729km^2) with a mean annual rainfall of 323 mm. It has a mean annual runoff of 200 x 10^6 m^3 (Perry 1983) and is subject to periodic flooding and high levels of artificial freshwater inflow (Wooldridge & Bailey 1982; Cowley *et al*. 2013). The spring and neap tidal range is approximately 1.2-1.5 m and is 0.1-0.3 m, respectively (Harrison & Whitfield 1990). Water temperatures range from 13-26ºC in winter and summer, respectively (Jerling & Wooldridge 1991). There is a horizontal salinity gradient present in the estuary, increasing from the upper reaches to the mouth (Harrison & Whitfield 1990). A strong vertical salinity gradient is also present as a result of saltwater intrusion from the sea (Wooldridge & Bailey 1982). Salinity levels are highest near the mouth of the estuary due to the permanent connection with the ocean.

The Sundays Estuary has a high National Conservation Importance Rating of 41 (out of251 estuaries) which can be attributed to its size, high biodiversity and estuarine type (Turpie *et al*. 2012). Permanently open, phytoplankton-driven estuaries like the Sundays Estuary are uncommon in South Africa. In order to sustain their unique function and biodiversity, their conservation is essential (Bezuidenhout *et al*. 2011). The fish community in the Sundays Estuary constitutes 51 species (27 families), dominated by marine migrants, and followed by estuarine residents and marine stragglers (Bezuidenhout *et al*. 2011). According to Whitfield (2008) the estuary plays a crucial role as nursery and feeding habitats for many marine species, including the heavily targeted linefish species; *A. japonicus, P. commersonniiL. lithognathus* and *L. amia.*

The close proximity of Sundays Estuary to the Addo Elephant National Park and Coega Industrial Development Zone adds to its high economic, tourism and conservation potential (Turpie & Clark 2007). The landuse surrounding the Sundays Estuary includes residential, agricultural, conservation and recreational uses. Colchester and Cannonville settlement is the most develop settlement alongside the Sundays Estuary, with a combined population size between 1002 and 1279 for both in 2010 (Bezuidenhout *et al*. 2011). There are a number of restrictions in terms of public access, both natural, including steep banks, and man-made, such as private residences and land ownership adjacent to the banks (Figure 2.1). The estuary is a major tourist attraction and has a high recreational value, being utilised for a number of recreational activities that are both consumptive and non-consumptive (Cowley *et al.* 2013). It is most popular among recreational fishermen, with the effort of this activity far exceeding those of neighbouring estuaries (Pradervand & Baird 2002; Cowley *et al*. 2013).

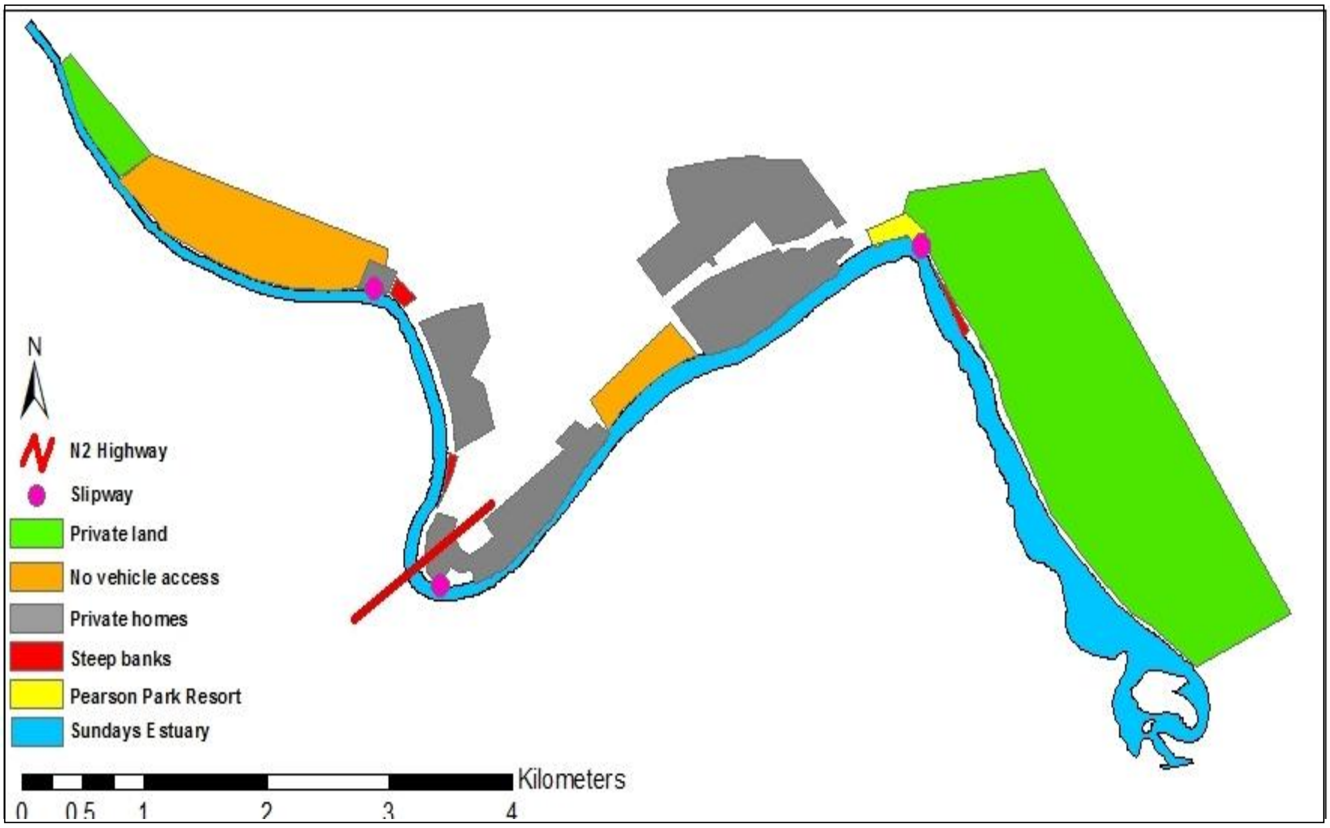

Figure 2.1: Map of Sundays Estuary showing main access routes and land use (adapted from Cowley *et al*. 2013).

2.2.2 Exploitation of living resources in the Sundays Estuary

Whilst recreational linefishing in South African estuaries has proved to be a very popular activity, research on the fishery has been limited in the past (Pradervand & Baird 2002). Reference to estuarine fisheries in Algoa Bay had been restricted to the work of Marais & Baird (1980) and Baird *et al.* (1996) who published work on the Swartkops and Sundays estuaries in the Eastern Cape. Since then a study was conducted between 1996 to 1997 by Pradervand & Baird (2002) who provided some base-line information on the Sundays estuarine fishery, by collecting catch and effort data on, and participation of the different user groups (subsistence and recreational fishers) of the shore- and boat-based line fisheries in South Africa. However, the socio-economic and institutional forces that act on the exploitation of estuarine resources were still fairly unknown. Cowley *et al.* (2013) then provided an assessment of the consumptive and non-consumptive resource use on the Sundays Estuary, including detailed information on spatial and temporal patterns of the different resource users and the demographics, effort, catch what was targeted by fishers compared to what was actually caught) and CPUE of the different fishery sectors. The findings from these studies highlight the need for improved estuarine management (Cowley *et al*. 2013).

The fishery survey conducted on the Sundays Estuary by Cowley *et al.* (2013) (including additional data utilised from Cowley *et al*. 2009) from September 2007 to August 2008 revealed the following:

- A total of 803 fishers were encountered during the 32 sample days.
- The fishery is male dominated (91.8%).
- 71.1% of the participants were white, 25.0% coloured and 2.3% black.
- 91.2% of participants resided in towns less than 50 km away, whilst 18.6% of the participants are local (living within 5 km of the estuary).
- 29.8% of participants held tertiary qualifications, 33.7% had incomplete secondary education, and 8.2% had no education.
- 49.3% were formally employed, while only 8.5% were currently unemployed.
- 63.5% claimed to have valid recreational angling permits, and 36.2% had no permits at all.

- 92.9% of fishers were from the recreational sector, and 7.1% from the subsistence sector.
- Effort was higher on weekends and public holidays and the holiday season (December to March) for recreational fishers, but no difference was noted for subsistence fishers. Angler counts peaked during the Christmas and Easter holidays. An estimated 64 367 angler-hours were fished during the survey period.
- Catch-per-unit-effort (CPUE) in terms of number was highest for boat-based fishers (0.30 fish.angler^{-1}.hour^{-1}) followed by recreational shore (0.23 fish.angler^{-1}.hour^{-1}) and subsistence fishers (0.21 fish.angler^{-1}.hour^{-1}). Overall average CPUE was 0.25 fish.angler^{-1}.hour^{-1}. In terms of mass, the subsistence sector had the highest CPUE estimate (0.21 kg.angler^{-1}-hour^{-1}), followed by the recreational boat sector (0.16 kg.angler^{-1}.hour^{-1}).
- *P. commersonnii* and *A. japonicus* were the most sought-after species, targeted by 51% and 44% of fishers interviewed.
- Nineteen species were recorded in the catches, of which *R. holubi* was numerically dominant (30.1%) followed by *P. commersonnii* (24.0%) and *A. japonicus* (21.8%). *P. commersonnii* and *A. japonicus,* the two most targeted species, collectively comprised 45.8% of the catch.
- 22.0% of the total catch (in numbers) was retained, 47.0% of which was below the minimum legal size limit, including 63.2% of *A. japonicus*, 30.2% of *P. commersonnii* and 100% of *L. lithognathus*.
- The majority of fish caught were under the minimum legal size (63.0% of *A. japonicus,* 30.0% of *P. commersonnii* and 100% of *L. lithognathus*).
- Catch rates were low, with fishers unable to attain daily bag limit of legal-sized fish for *P. commersonnii* and *A. japonicus* for >90% of outings.
- Knowledge of fish regulations was also poor, with only 13.0% of fishers providing correct minimum size limits for their target species. Minimum sizes also received the lowest score in terms of effective regulations.
- Compliance monitoring effort appears to be low, with 59.1% of fishers never having had their catches inspected, while 11.4% encountered law enforcement officers only once.
- 59.2% of fishers thought catches had decreased over time (catch rate and average size) with *A. japonicus*, *P. commersonnii* and *L. lithognathus* being the most

noteworthy. Only 4.8% of participants attributed stock decline to poor law enforcement.

- The high percentage of undersize fish kept by fishers was viewed as a major concern, and an increase in compliance monitoring efforts combined with an awareness raising campaign was strongly motivated.

Additional information

- The fishery is diverse, comprising of shore and boat-based fishers using a variety of gear types (rod and reel and handlines).
- The recreationally dominated fishery is a major economic asset of the Sundays Estuary.
- 36.0% of respondents said they fished at night, while only 7.0% responded positively when asked whether they would support a ban on night fishing in the estuary.
- 30.3% of participants indicated that a conservation authority (e.g. SANP) was responsible for managing living resources of the estuary, while 13.5% and 10.1%, respectively, believed it was National or Local Governments responsibility.
- 14.0% of the users suggested zoning of activities on the estuary was necessary. 25.8% of participants believed that closed areas would be effective, while 48.3% advocated closed seasons as a viable option.

The most targeted species in the Sundays estuarine fishery were *A. japonicus* and *P. commersonnii* and collectively dominated the catch (45.8%). This proportion is significantly lower compared to previous studies. In the survey conducted by Baird *et al*. (1996), 90.2% of fishers catches were comprised of *A. japonicus* and *P. commersonnii*, and observations of Marais & Baird (1980) found that *P. commersonnii* dominated fishers catch by 62.3%, followed by *A. japonicus* (27.9%) in the 1970's. These results are similar to other studies conducted in Eastern Cape estuaries who found that *A. japonicus* and *P. commersonnii* were the most targeted species and dominated the catch in the nearby Swartkops Estuary (87.4%) (Baird *et al.* 1996) and in the Great Fish Estuary 72.6% (Potts *et al.* 2005). In the Kowie Estuary however, the catch of these two species was relatively lower, comprising only 23.8% (Cowley *et al.* 2004, Cowley *et al*. 2013).

Cape stumpnose *Rhabdosargus holubi* and white sea-catfish *Galeichthys feliceps,* although not species targeted by fishers, were dominant in the catch in the most recent study (Cowley *et al.* 2013). The contribution of *P. commersonnii* to the total catch by number decreased significantly between the two studies. The percentage contribution of *R. holubi* significantly increased in the latter study (Cowley *et al.* 2013) (4.3 to 30.1% respectively). The change in catch contribution and the increased importance of smaller species like *R. holubi* may be a result of a decreased occurrence of the highly targeted *P. commersonnii* (Pradervand & Baird 2002). Although the percentage contribution, by numbers, of *A. japonicus* remains unchanged between the two survey periods, the contribution, in terms of mass, decreased which could be an artifact of increased targeting of juveniles, because of a decrease in the abundance of larger fish (Cowley *et al*. 2013).

The percentage contribution of the most targeted fish in terms of mass caught from the period 1996-1997 (Pradervand & Baird 2002) differed considerably to the catch found during the survey conducted by Cowley *et al.* (2013) (Table 2.3). Due to its large size, *A. japonicus* dominated the catch comprising of 69.0% of the total caught in the survey conducted by (Pradervand & Baird 2002). This was considerably higher than the contribution to the catch of *A. japonicus* (42.3%) in the most recent study (Cowley *et al*. 2013). The contribution of *P. commersonnii*, by mass remained unchanged for both surveys.

Table 2.3: Total catch composition (retained and released fish) for the Sundays Estuary in terms of the number and mass of the most dominant species caught (adapted from Baird *et al*. 1996, Pradervand & Baird 2002, Cowley *et al*. 2013).

Pradervand & Baird (2002) (for the period 1996-1997)			**Cowley *et al.* 2013 (for the period 2007-2008)**	
Species:	% contribution (Number caught)	% contribution (Mass (kg) caught)	% contribution (Number caught)	% contribution (Mass (kg) caught)
A. japonicus	20.3 (138)	69.2 (931.9)	21.8 (326)	42.3 (310.1)
P. commersonnii	43.1 (293)	23.4 (314.6)	24.0 (359)	24.2 (177.6)
L. lithognathus	0.7 (5)	0.1 (1.1)	7.4 (111)	3.4 (25.2)
L. amia	1.2 (8)	0.3 (4.0)	1.7 (25)	1.7 (12.6)
R. holubi	4.3 (29)	0.1 (0.7)	30.1 (450)	2.9 (21.4)
G. feliceps	24.6 (167)	1.6 (21.1)	9.9 (148)	4.9 (35.9)

The mean individual size of *A. japonicus* (359 mm TL), in the most recent study was similar to those observed in surveys conducted on the Kowie (337 mm TL); Cowley *et al.* 2004) and the Great Fish (418 mm TL; Potts *et al.* 2005), but substantially smaller than the recorded length of *A. japonicus* (632 mm TL) observed by Pradervand & Baird (2002). A similar trend was observed for *P. commersonnii* whose mean individual size (314 mm TL) caught in the Sundays Estuary was similar to that observed in the Kowie Estuary (326 mm TL; Cowley *et al.* 2004), but slightly lower than the mean size observed in the Great Fish Estuary (411 mm TL; Potts *et al.* 2005), and the earlier survey conducted on the Sundays Estuary (410 mm TL; Pradervand & Baird 2002) (Table 2.4). The dominant size class (<300 mm TL) of *L. lithognathus* in the most recent study (Cowley *et al*. 2013) was similar to that observed in other estuarine fishery surveys including (Pradervand & Baird *et al*. 2002, Cowley *et al*. 2004).

Table 2.4: Mean size (mm TL) of fish caught in the Sundays Estuary taken from Pradervand & Baird (2002) and Cowley *et al*. (2013).

Species	Pradervand & Baird 2002 (for the period 1996-1997)	Cowley *et al*. 2013 (for the period 2007-2008)
A. japonicus	632	359
P. commersonnii	410	314
L. lithognathus	228	250
R. holubi	Data not available	137
G. feliceps	Data not available	281
L. amia	320	381

Whilst decreases in size are indicative of increased exploitation levels, Cowley *et al.* (2013) suggested that the decrease in size between the Sundays Estuary studies may have been a result of the law enforcement officials used during the earlier study by Pradervand & Baird (2002). As a consequence of this, fishers may have not reported their undersized fish. However, since *A. japonicus* reach sexual maturity > 1000 mm TL (Griffiths 1996), growth over-fishing in estuaries is to be expected (Cowley *et al.* 2013). The decrease in *A. japonicus* catch, a competitor for prey, could explain the increase in *L. amia* catch. Unfortunately, information on the mean size of *R. holubi* and *G. feliceps* was not available for comparison.

The current minimum legal size limit for *A. japonicus*, *P. commersonnii* and *L. lithognathus* are 600 mm TL, 400 mm TL and 600 mm TL, whilst the mean individual size caught of these species in the Sundays Estuary during the most recent study was <400 mm TL, <300 mm TL and <400 mm TL respectively (Cowley *et al.* 2013). The large proportion of undersized fish is to be expected, owing to the nursery role that estuaries provide to these estuary-dependent species (Whitfield 1998).

In the most recent study (Cowley *et al.* 2013), the retention rate of fish was relatively low (22%), however, the retention rate of undersized fish was very high (47%). Another alarming result found in the study was the high proportion of retained undersized fish whose stocks are collapsed, namely *A. japonicus* (63%) and *L. lithognathus* (100%). This was similar to the results observed by Hutchings *et al.* (2008), in which 100% of undersized *L. lithognathus* (<600 mm TL) were retained in the Berg Estuary.

Cowley *et al*. (2013) also found an overall low angler success rate for the two most targeted species (*A. japonicus* and *P. commersonnii*). Fishers were unable to attain the daily bag limit for these two species in more than 90% of outings.

It is well understood that the South African linefishery, and estuarine fisheries in particular, are oversubscribed, yet effort and participation continue to increase (Baird *et al.*1996, Cowley *et al.* 2013). The information collated from recent and past fishery surveys highlight the ineffectiveness of current regulations such as daily bag and minimum size limits and the need for more appropriate management interventions, such as area closures, that aim to reduce total fishing effort (Whitfield & Cowley 2010). The high proportion and retention rates of juvenile fish in angler catches and high targeting effort towards vulnerable species, as well as the high levels of non-compliance and poor knowledge of regulations, question the sustainability of the Sundays estuarine fishery (Cowley *et al.* 2013).

The most recent survey conducted on the Sundays Estuary observed an increase in fishing effort during the summer months, which can be related to the increase of recreational fishers during the holiday periods (December-March). This trend was observed in numerous estuarine fishing surveys (Marais & Baird 1980, Mann *et al.* 2002, Pradervand & Baird 2002, Cowley *et al*. 2004). The recreational fishing effort was substantially higher (59 239 angler-hours) than the subsistence fishing effort (8 260 angle-hours). This highlights the dominance of recreational users on the estuary. The annual effort (per km of shoreline) was four times greater on the Sundays Estuary compared to the adjacent coastal zone (between Port Ngqura and Boknes) (Chalmers 2012, Cowley *et al*. 2013). This highlights the concentration of fishing effort placed on estuaries, and more importantly, juvenile nursery areas.

The sustainability of these resources will be determined by a comprehensive understanding of the status of the fishery and the adverse effects of exploitation within estuaries (Chapter 3), as well as a carefully planned management approach (Chapter 4) in order to prevent further declines in fish stocks and degradation of estuarine ecosystems (Cowley *et al.* 2013).

CHAPTER 3: SUSTAINABILITY ASSESSMENT OF THE SUNDAYS ESTUARINE FISHERY USING A SUITE OF INDICATORS

3.1 INTRODUCTION

Globally, many of the world's fish stocks are depleted as a direct result of overexploitation and are therefore in a non-sustainable state (Ye *et al*. 2013). There is a growing concern that intensive fishing in marine ecosystems, due to bad governance, a lack of compliance, a lack of knowledge and a demand that exceeds the resource is unsustainable (Hilborn 2007). In South Africa, many human activities are carried out on estuaries and their catchments, and this directly impacts estuarine resources (Mann 2013). The single biggest threat to estuarine fisheries is exploitation and unsustainable targeting of vulnerable, overexploited species (Whitfield & Cowley 2010).

As a result of the unsustainable fishing practices in South African estuaries, and the failures of traditional management regulations (e.g. bag and size limit restrictions), many scientists and managers have attempted to move towards fisheries management strategies that integrate the social, institutional and biological aspects of fisheries (Zhou *et al.* 2010). Globally, decisions relating to environmental issues have, in the past, been made on an *ad hoc* basis, with each particular problem being dealt with in isolation (Hák *et al*. 2012). Sustainable development is a concept that aims to integrate these aspects in decision making in order to meet the needs of present generations without compromising the ability for future generations to meet their own needs (Fletcher *et al*. 2005). The overall objective of sustainable development, which needs to be incorporated into practices and policy, is to achieve economic prosperity, social well-being, and environmental recovery and protection (Hák *et al*. 2012).

The idea of sustainable fisheries has been embraced by many countries including the USA, whose fisheries management is now based on the Sustainable Fisheries Act of 1996 and the American Fisheries Society (AFS) whose strategic plan has incorporated sustainability as a central element of its vision. In Australia, fisheries are now managed on the basis of Ecological Sustainable Development (ESD) where the framework is divided into components of ecological well-being and human well-being. The concept seeks to integrate economic, social and environmental effects and values into decision making (Fletcher *et al.* 2005). In South Africa, the Marine Living Resources Act (regulations promulgated in terms of the Marine Living Resource Act No. 18 of 1998 [Government Gazette No. 27453]) is based on sustainability principles, and its aim is to "*provide for the conservation of the marine ecosystem, the long-term sustainable utilisation of marine living resources and the orderly access to exploitation*". Following the Reykjavik conference held in Iceland in 2001, the Reykjavik Declaration on Responsible Fisheries in the Marine Ecosystem was signed; this recognised the ecosystem-based approach to fisheries as a form of fisheries management. In 2002, at the World Summit for Sustainable Development held in Johannesburg, South Africa, an EBM approach was reinforced, and it was agreed that management sectors from fishing nations would incorporate an EBM by the year 2010 (FAO 2003, Cochrane *et al.* 2004). South Africa joined the trend during this summit and agreed to rebuild fish stocks to levels that can produce MSY by no later than 2015.

There is increasing recognition that humans are a necessary part of ecosystems, and consequently dependent on the environment for their societal and economic development (Hughs *et al.* 2005). This concept has bridged the gap between marine ecology, fisheries biology and social science and management is now becoming a process-orientated activity that takes resilience of these sectors into account. That is, the ability of an ecosystem or species to prevent disasters, anticipate, absorb and recover from them in a timely and sustainable manner. This would include the protection, restoration and improvement of ecosystems in the face of threat (Hughs *et al.* 2005). Furthermore, social resilience would include the ability of a community to cope with external stresses and disturbances as a result of environmental change (Adger 2000). To identify the link between social and ecological resilience, there needs to be an understanding of the state and pressures placed on both the threatened species, and the society exploiting these species. It is clear that fisheries management cannot be solely focused on the resource, and an incorporation of human needs, which may change both spatially and temporally, is essential (Smith 2005).

Fisheries management therefore has to be adaptable as the needs of both the resource and society change. Increasingly, management systems are utilising indicators as tools to measure the changes, outcomes and impacts a fishery experiences at an ecological, societal and institutional level (Casto 2001, Smith 2005). Indicator-based approaches have been endorsed by many management and policy bodies related to marine systems (FAO 2002, Rice & Rochet 2005). Indicators represent an approach that is designed to meet environmental challenges and allow for expanding observations of social and environmental processes to be drawn into policy making (Hák *et al*. 2012).

In 1995, the FAO developed the code of conduct for responsible fisheries, in which, guidelines were provided for responsible fishing practices in order to ensure the "*effective conservation, management and development of living aquatic resources, with due respect for the ecosystem and biodiversity*". This code encouraged a strategy that incorporated ecosystem considerations into management practices. It provided a set of technical guidelines that consolidated the available knowledge related to fisheries and proposed a Sustainability Development Reference System (SDRS) in order to develop and organise indicators and reference points in such a way that resource managers and decision makers can adequately assess and monitor the sustainability of a fishery (Garcia *et al.* 2000, Pajak 2000, Dahl 2000, Rice & Rochet 2005, Hák *et al*. 2012). The SDRS uses indicators to provide a cost-effective way to track progress towards sustainability, predict warning about potential problems, compare performance between fisheries and inform policies aimed at advancing progress (FAO 1995).

Within the context of SDRS, the next step is to develop a framework that organises indictors in relation to sustainable development. A number of models have been developed arranging their indicators in different frameworks to illustrate sustainability. One of which is the Pressure-State-Response (PRS) and its alias the Driver-Pressure-State-Impact-Response (DPSR) framework (OECD 1993, European Environmental Agency 2003), which implies causal relationships between indicators, and considers the pressure imposed by human activities on aspects of a system, the state of the resource and the political response to adopt solutions response (OECD 1993, Fletcher *et al.* 2005, Hák *et al*. 2012). Others are derived from the FAO definition of sustainable development which results in dimensions including resources, institutions, people, technology and the environment. Some frameworks incorporate the FAO Code of Conduct for Responsible Fisheries and are divided into fishing operations, fisheries management, aquaculture development post-harvest practices and trade and fisheries research (Garcia & Staples 2000, Rudd 2003).
Rapid Appraisal Techniques have also been used in order to develop a qualitative understanding of a situation (Pitcher & Preikshot 2001). The RAPfish technique evaluates the sustainability of fisheries based on the quantitative scoring of sets of ecological, social, technological and institutional attributes (Pitcher *et al.* 2013). This technique has been

used in the sustainability assessments of several fisheries, including African lake systems (Preikshot *et al.* 1998), the Tagus Estuary, Portugal (Baeta *et al.* 2005), the Red Sea (Tesfamichael & Pitcher 2006) and for assessing the social sustainability of fishery cooperatives in the Guilan Province, Iran (Allahyari 2010).

Whist these models are different in their structure and have diverging principal components, their general themes of sustainability is the same. As long as the framework encompasses the purpose of the SDRS, in practice, it is not critical which framework is adopted (Garcia *et al.*2012).

Whilst South Africa's MRLA is based on sustainability principles and South Africa has pledged to incorporate an EBM approach into fisheries management, research on sustainable fisheries in estuaries is still in its infancy. In South Africa, estuaries are subject to certain regulations under the MLRA and, on paper, enjoy some level of protection (Van Niekerk & Turpie 2012). A National Estuarine Management Protocol under the Integrated Coastal Management Act (regulations promulgated in terms of the National Environmental Act No. 24 of 2008 [Government Gazette No. 31884]) has been published, in which guidelines for the development of estuary-specific management plans are given. Although these management plans call for the protection of estuaries in order to achieve sustainability, the current level of law enforcement is low, this leads to non-compliance and therefore impacts fish stocks (Bezuidenhout *et al.* 2011). Owing to the unsustainable targeting of vulnerable, overexploited species such as *A. japonicus*, *L. lithognathus* and *P. commersonnii* in the Sundays Estuary, and the failure of current management approaches, there is an urgent need for research to assess the sustainability of estuarine fisheries and resource utilisation in order to design suitable protocols by which to protect them.

3.1.1 A case study from the Sundays estuarine fishery

Following the proposed expansion of the Greater Addo Elephant National Park to include an MPA, which excluded the Sundays Estuary itself, Cowley *et al.* (2013) conducted an assessment of the consumptive and non-consumptive resource use on the Sundays Estuary. The survey designed by Cowley *et al.* (2013) used the parameters of the Driver-Pressure-State-Response framework in order to identify the pressure imposed on vulnerable fishery species (state) within the estuary, the current state of knowledge of regulations and level of non-compliance and the response in terms of law enforcement and management of the fishery (response).

The scope of a SDRS is to incorporate an ecosystem-based approach to fisheries, and it has been widely accepted that this approach strays from a single-species approach, in terms of stock assessments (FAO 2012). However, the state of targeted estuary-dependent fish species is dismal, and the lack of law enforcement and compliance in South African estuaries warrants the need to assess sustainability of a fishery at an individual species level using EBM principles. Assessing the state of sustainability at a species level can be incorporated into larger ecosystem assessments of emergent properties at a later stage (Sainsbuy & Sumalia 2003). Although the Sundays estuarine fishery is a multi-species fishery; 95% of the effort is targeted on only two species, namely *P. commersonnii* and *A. japonicus,* (Cowley *et al.* 2013, see Chapter 2). They were therefore chosen as the proposed indicator species because of their popularity amongst estuarine fishers and their dominance in the catches of Eastern Cape estuaries (see Chapter 2). Their susceptibility to overfishing due to their popularity and life-history traits, particularly *A. japonicus,* also makes them prime candidate species.

3.2 METHODS

3.2.1 Framework

The framework described below includes elements from the General Sustainability Framework, the ESD framework (Fletcher *et al.* 2005) and the Driver-Pressure-State-Response framework, together with a suite of appropriate indicators, adapted from principles used in RAPfish (Pitcher & Preikshot 2001), to assess the sustainability of the fishery at a local level. This assessment incorporates social, institutional and biological aspects of the fishery in order to identify priority areas of unsustainability.

A number of steps were involved in the selection of indicators (Figure 3.1). These steps form a framework based on the literature, and allow for an iterative process by which sustainability can be assessed. The first step was to collate information gathered by Cowley *et al.* (2013) on the consumptive and non-consumptive resource use on the Sundays Estuary, with details of temporal and spatial patterns of different resource use activities. This information gave insight into social and institutional aspects of the fishery. Information regarding the biological domain (e.g. targeted fish species) was collated into an extensive literature review (Chapter 2) from previous base-line assessments conducted on the Sundays Estuary (Baird *et al.* 1996, Pradervand & Baird 2002), as well as literature on the current status of the fish stock, and the biology and ecology of targeted species.

3.2.2 Identifying key issues and operational objectives

From the review of literature and current knowledge of social, institutional and biological domains, key sustainability issues were identified (Step 2, Figure 3.1). This highlighted the impacts fishing had on the environment, both biologically (at a population and community level), and socially (as a source of recreation or food). Following the FAO Code of Conduct for Responsible Fisheries (FAO 1995), and the key issues identified in Step 2 from the literature, objectives of the fishery were identified (Step 3, Figure 3.1). The objectives were divided into the three domains of social, institutional and biological sustainability (Table 3.1). From these three core objectives, major components of sustainable development were then identified that cover the social, biological and institutional areas to allow a provisional assessment of the sustainability of the fishery (Step 4, Figure 3.1 & Table 3.1). In order to assess the progress towards the objectives identified in step 3 and 4, appropriate indicators were developed (Step 5, Figure 3.1) (Smith & King 2010).

Table 3.1: Operational objectives of principles of sustainable development related to the FAO Code of Conduct for Responsible Fisheries (Garcia 2000), the South African Marine Living Resources Act and the Integrated Coastal Management Act used for this study.

Social	
S1: "The human needs (in terms of sustainable access to high quality and safe food and recreation), and societal / ethical values should be satisfied."	
S1.1	Protect the interest of subsistence fishers, and alleviate poverty
S1.2	Improved knowledge and compliance of regulations
S1.3	Facilitate effective participation in decision making
Institutional	
I1: "An effective management system should be in place, to orient the institutional and technological change required."	
I1.1	Provide suitable management of fish species through enforcement of regulations
I1.2	Promote awareness about conservation and management among fishers
I1.3	Monitor management performance and review management strategies
Biological	
B1: "The natural resource (fish) should be conserved: The target resource characteristic should be maintained at levels capable of ensuring its natural renewal and continuous exploitation under ecologically acceptable conditions."	
B1.1	Preserve the availability of resources of the Sundays Estuary
B1.2	Prevent over-exploitation of resources
B1.3	Protection of juveniles and spawners

3.2.3 Selection of indicators and performance criteria

Once the indicators had been developed, they were then categorised (Step 5, Figure 3.1) according to the chosen Driver-Pressure-State-Impact-Response model (Bowen & Riley 2003). A rationale was then given for why each indicator was chosen for each domain to achieve the overall objectives of sustainability (Step 6, Figure 3.1). The performance criterion specifies how to interpret the indicator by outlining one or more threshold reference points (target or limit reference point) or in terms of a trend that may be increasing which is desirable, or decreasing which is undesirable (Step 7, Figure 3.1). The operational objective, indicator and performance criteria are a package; all three are needed before any one of them is useful (Fletcher *et al*. 2005).

3.2.4 Visualisation of sustainability

Finally, once reference points had been set, and a quantitative value was established for each indicator, the performance criteria were scored on a scale from 0 to 4; representing a state from very poor to good (Garcia *et al.* 2000). They were then aggregated across the three domains to indicate sustainability. This was achieved by using a Rapid Assessment Matrix (RAM), which allowed for a simple and quick identification of the limiting components of each domain (Wood *et al*. 2003). The total scores were then converted into a percentage of the total possible score in order to compare values across domains. The Biological domain, consisting of separate scores for the two most targeted species (*P. commersonnii* and *A. japonicus*) were calculated separately to compare the sustainability of each species individually.

The results were then visually represented using a kite (or amoeba) diagram, where the axes illustrate each indicator (Garcia 2000). Such visual representation is simple and comprehensive, and easy to identify areas of unsustainability and which may need improvement (Pajak 2000). The diagram is divided into five percentages representing different performance criteria an indicator could score, with 0% (Very poor) at the centre of the diagram indicating non-sustainability, and 100% (Good) at the perimeter of the diagram indicating sustainability of the domain.

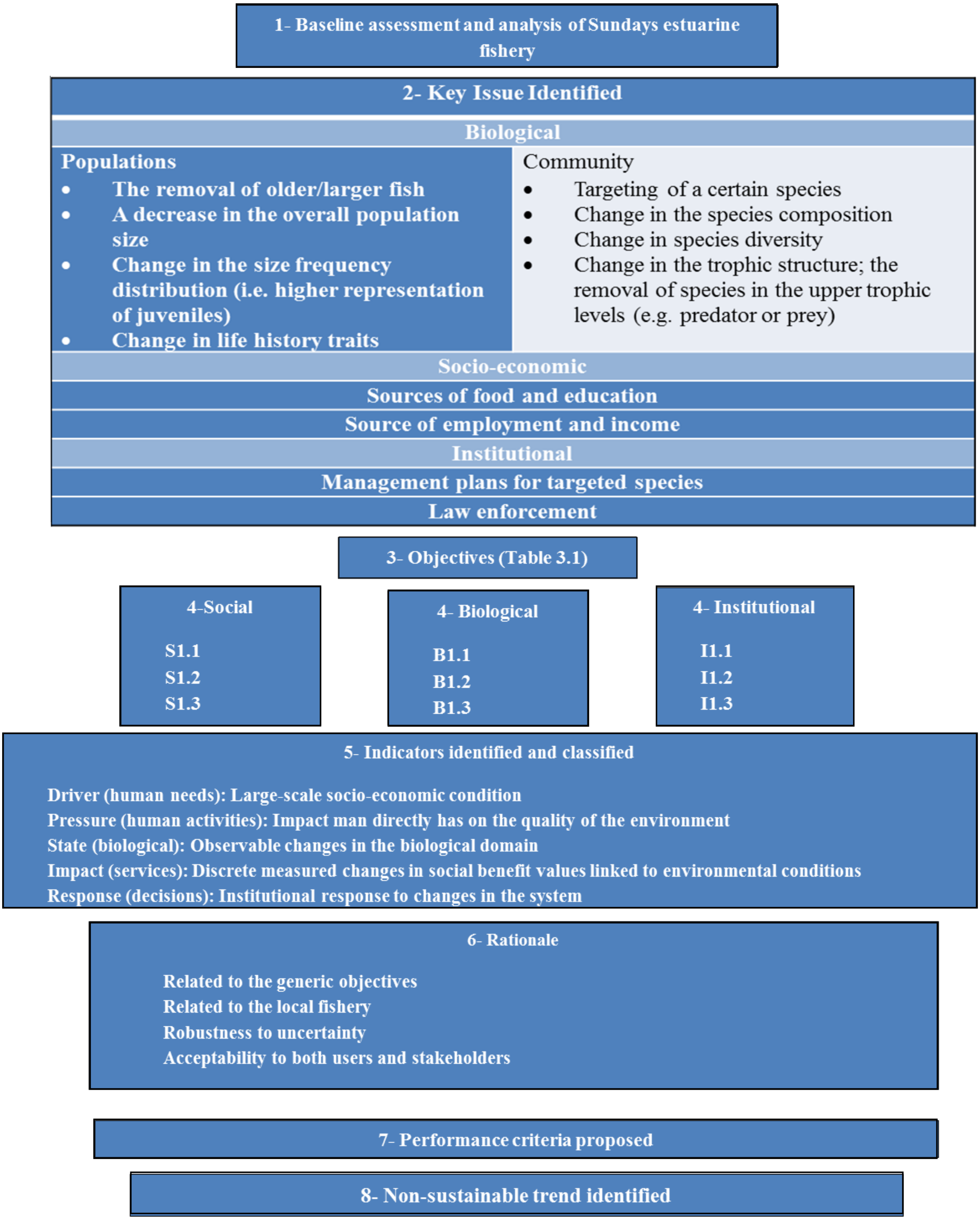

Figure 3.1: The eight steps involved in the selection of indicators. This diagram shows how the selection of indicators is dependent on the operation objectives and key issues relating to the fishery.

3.3. PROPOSED INDICATORS

3.3.1. Social domain

Human beings depend on nature and its resources for their well-being. However, our activities have made unprecedented changes to ecosystems and had detrimental effects on the biodiversity within these ecosystems (FAO 1995). The DPSR framework, considers the pressure imposed by human activities on aspects of an environment, as well the state of human needs, and the desired societal response (FAO 2012). Owing to the increase in fishing pressure in South African estuaries, there is a need to understand the adverse effects of consumptive exploitation within these systems (Cowley *et al*. 2013). The results from the survey conducted by Cowley *et al.* (2013) highlight a number of social issues that are applicable to the management of estuarine fisheries, and are discussed below (Table 3.2).

Table 3.2: Proposed indicators in relation to generic management objectives and specific management issues identified for the Social domain.

Domain	Principle[1]	Sub-principle[2]	Issue	Indicator	Indicator type (DPSIR)	Source[3]	Possible action
Social	S1	S1.2	Poor regulatory knowledge	% of fishers who knew the current linefish regulations	D	RCS	Educational: Awareness/signs
	S1	S1.2	Non-compliance	% admitted non-compliance	D	RCS	Increase awareness and enforcement
	S1	S1.2		% undersized catch kept	P	RCS	
	S1	S1.1	Dependence on fish as food	% of subsistence fishers	S	RCS	Alternative livelihood projects
	S1	S1.1		% of catch crucial to diet	S	RCS	
	S1	S1.3	Sense of belonging	% of local residents	S	RCS	Incorporation of residents in decision making

[1]: Major principles taken from the FAO Code of Conduct for Responsible Fisheries

[2]: Sub-principles taken from the FAO Code of Conduct for Responsible Fisheries

[3]: RCS: Roving Creel Surveys, IL: Independent Literature

Issue: **Poor regulatory knowledge and admitted non-compliance**

Rationale:

In the Sundays Estuary, one of the biggest issues found was the lack of knowledge and compliance of current regulations pertaining to targeted species. This suggests that well-educated, recreationally dominated fishers are ignorant of the law and they do not understand the needs of estuarine resources. Whilst fishers were found to be ignorant of current regulations, they believed that fish stocks had declined in the past 5 years, and attributed this to either illegal fishing, or general overexploitation which suggests that they are aware of the impacts of overexploitation of fish species (Cowley *et al*. 2013).

Knowledge of current regulations is the most primary step towards compliance with those regulations (Smith & King 2010). Compliance is paramount to the success of any estuarine management plan and should be regarded as the first step towards compliance of current regulations. By increasing knowledge and compliance of current regulations, the operational objectives and goals for sustainability of the Sundays Estuary can be achieved (S1.2, Table 3.2). The three indicators that were developed to address this issue are the percentage of fishers who knew current linefish regulations, and percentage of fishers who admit to breaking the current linefish regulations and the percentage of undersized fish retained.

Performance Criteria:

Indicator:	Very Poor (0)	Poor (1)	Moderate (2)	Fairly good (3)	Good (4)
Percentage of fishers who knew the current linefish regulations (%)	0-20	20-40	40-60	60-80	80-100
Proportion of fishers who admit to breaking the linefish regulations (%)	80-100	60-80	40-60	20-40	80-100
Percentage of fishers who kept undersized fish (%)	>50	40-50	30-40	20-30	<20

***Issue:* Dependence on fish as food**

Rationale:

According to Branch *et al.* (2002) subsistence fishers are classified as poor people who harvest marine resources as a source of food or income as a way to meet the basic needs for food security, live locally and use low technology gear such as handlines. There is a greater pressure on the fishery resource when there are more subsistence fishers in the fishery. In South Africa, the term subsistence fisher has been defined and distinguished from small-scale commercial fisheries, primarily on the grounds of the two groups using different resources and having different objectives (Branch *et al*. 2002, Sowman 2011). Whilst the differences have been acknowledged by the authors, the term 'subsistence' fishers will be used to describe fishers who depend on the fishery resource for their livelihood. A decrease in the number of subsistence fishers would alleviate the pressure of people with a dependency of the resource (S1.1, Table 3.1).

This indicator is split into two parts. The first being the proportion of fishers who said they fished for food (and not recreation), and the second identified the importance of catch in the fishers household diet.

Performance Criteria:

Indicator:	Very Poor (0)	Poor (1)	Moderate (2)	Fairly good (3)	Good (4)
Percentage (%) of subsistence fishers	80-100	60-80	40-60	20-40	0-20
Percentage (%) of fishers whose catch is crucial to their diet	80-100	60-80	40-60	20-40	0-20

***Issue:* Sense of belonging**

Rationale:

According to Pajak (2000) social sustainability will exist when basic human needs are met. Basic human needs are organised according to Maslow's hierarchy in which five basic human needs are identified, one of which is the sense of love and belonging (Schaffer 2016). If fishers have access to local fishery resources in the Sundays Estuary, they will have a higher likelihood of being invested in their conservation and management. Therefore it is assumed that the residents of the Sundays Estuary should form the majority of fishers in the Sundays estuarine fishery. Although the Sundays Estuary and its surrounding areas have much potential as an important tourist destination and the economic growth would be greatly beneficial to the area, the effects of increased tourism may render the ecosystem and its resources more vulnerable to (further) degradation and exploitation.

Whilst the number of fishers residing less than 50km from the estuary was high (91.2%), only 18.6% of these were considered to be 'local', residing less than 5km from the estuary. This indicator needs to be used with caution when identifying the distance at which fishers are considered to be local. Since Port Elizabeth, a large metropolitan city, falls within 50km of the estuary, 5km was chosen as the distance at which fishers were considered to be local.

Performance Criteria:

Indicator:	Very Poor (0)	Poor (1)	Moderate (2)	Fairly good (3)	Good (4)
Percentage of fishers who reside <5km from the Sundays Estuary	0-20	20-40	40-60	60-80	80-100

3.3.2. Institutional domain

Rationale:

As a result of the beach vehicle ban, improved infrastructure and technological advances, effort and participation in estuaries are increasing (Cowley *et al.* 2013). This has heavily impacted estuary-dependent fish species.

Pajak (2000) defined result-orientated institutions as organisations that are predominantly outcome based. They must have clearly stated and measurable objectives that are annually monitored. The technical guidelines on fisheries management (FAO 1997) describe a management plan as "a formal or informal arrangement between a fisheries management authority and interested parties which identifies the partners in the fishery and their respective roles, details the agreed objectives for the fishery and specifies the management rules and regulations which apply to it and provides other details about the fishery which are relevant to the task of the management authority". A management plan that is fully integrated with national fisheries objectives will therefore be beneficial in achieving the objectives of the fishery (FAO 1995).

The indicators developed in Table 3.1 will illuminate the effectiveness of the institutional domain. Although a draft Estuary Management Plan (Bezuidenhout *et al.* 2011), discussing the Institutional arrangements and responsibilities, exists, it has not yet been implemented.

Table 3.3: Proposed indicators in relation to generic management objectives and specific management issues identified for the institutional domain.

Principle[1]	Sub-Principle[2]	Issue	Indicator	Indicator type (DPSR)	Source[3]	Possible Action
I1	I1.1	Effective estuarine fisheries management plan	Existence of a fisheries management plan	R	Municipal policies	Development and implementation
I1	I1.3	Adaptable management plan	Monitoring plans in place	R		
I1	I1.1	Effective enforcement	Number of fishers inspected	D	RCS	Increase the frequency of inspections
I1	I1.1 I1.2		Number of times enforcement was witnessed during survey period	S	RCS	Increased signage, educational drives

[1]: Major principles taken from the FAO Code of Conduct for Responsible Fisheries

[2]: Sub-principles taken from the FAO Code of Conduct for Responsible Fisheries

[3]: RCS: Roving Creel Surveys, IL: Independent Literature

***Issue*: An effective management system should be in place**

Rationale:

Currently, the only management of fish species in the Sundays Estuary is through traditional bag and size limit restrictions. Whilst some estuaries in South Africa are afforded some temporal and spatial restrictions, such as the night fishing ban on the Breede Estuary (Wood *et al*. 2004) and the prohibition of fishing in parts of Goukou Estuary (DEAT 2005), very few restrictions are based on scientific data. To date, no management plans have been developed that successfully prohibit the exploitation of unsustainable fish species in the Sundays Estuary. The lack of compliance and enforcement and resulting decrease of fish stocks shown by Cowley *et al.* (2013) illustrates the need for Institutional implementation of the Draft Estuarine Management plan.

Performance Criteria:

Indicator:	Very Poor (0)	Poor (1)	Moderate (2)	Fairly good (3)	Good (4)
Nature of management plan	No management plan	Limited management (bag and size limits) but no specific plan per estuary	Integration of species-specific plan at an estuarine level, limited scientific evidence	Fully integrated species-specific and estuarine specific plan (spatial, temporal plans based on scientific evidence)	Fully integrated species-specific plan (no-take estuarine protected areas based on scientific evidence)

***Issue:* Management plans must be adaptable**

Rationale:

The establishment of an ongoing monitoring programme is essential for the success of a management plan. Integrated and flexible monitoring frameworks allow managers to measure and follow ecological quality and allows for continual defining and redefining of management issues (Tobey & Volk 2002, Knol 2010). Measuring ecological quality is done by identifying a reference framework (i.e. what is ‘normal’ and at what capacity the environment can sustain the ecosystem effects placed on it). An ecosystem-based management approach also needs to be adaptable, and transparently reflect changes in local political, socio-economic and ecological environments (Holness & Biggs 2011).

Although there have been previous assessments conducted on the Sundays Estuary (Marias & Baird 1980, Baird *et al.* 1996, Pradervand & Baird 2002) and the fishery (Cowley *et al.* 2013), sampling techniques are often incomparable and the objectives of the assessments differ

Performance Criteria (Adapted from Smith & King 2010)*:*

Indicator:	Very Poor (0)	Poor (1)	Moderate (2)	Fairly good (3)	Good (4)
Nature of monitoring programme	No monitoring programme	Occasional surveys	Annual surveys conducted, however data collected is limited	Annual long-term surveys in place with fishery and socio-economic data collected	Frequently collected Data, incorporated into a management plan which is analysed and assesses the success of management

***Issue*: Effective enforcement of current regulations**

Rationale:

It has now been understood that regulatory measures have little or no effect unless they are adequately implemented and enforced (Whitfield & Cowley 2010, Turpie & Goss 2014). Brouwer (1997) identified a direct correlation between the number of fishery inspections and angler compliance. If there is a larger presence of inspectors along the Sundays Estuary, an expected increase in voluntary compliance might occur. While the logistical problems associated with law enforcement are recognised, it is evident that the absence of adequate enforcement may increase non-compliance. This needs to be addressed in order to ensure the sustainability of estuarine fish stocks (Cowley *et al.* 2013). This indicator is based on the frequency of catch inspections on the Sundays Estuary.

This indicator is split into two parts, firstly indicating the number of times fishers had been inspected once out of 100 outings. This value was chosen to represent an adequate time-frame in which to judge the number of inspections. Secondly, indicating the number of times inspectors were present during the duration of the roving creel survey conducted by Cowley *et al.* (2013). Whilst very similar, these two indicators can be used together to give an accurate representation of enforcement on the Sundays Estuary.

Performance criteria:

Indicator:	Very Poor (0)	Poor (1)	Moderate (2)	Fairly good (3)	Good (4)
Percentage of fishers inspected (%)	0-20	20-40	40-60	60-80	80-100
Percentage of times law enforcement was witnessed during survey period (%)	0-20	20-40	40-60	60-80	80-100

3.3.3. Biological domain

The state of estuary-dependent fish stocks has reached a level of emergency (Chapter 1 and 2). Whilst there is a great need to assess the social and institutional forces impacting the state of estuarine fisheries, it is also important to assess the state of the fishery itself. The following indicators (Table 3.4) are based on the operational objectives (Table 3.1) defined by the FAO Code of Conduct for Responsible Fisheries, the MRLA and from the current unsustainable issues addressed in Chapter 2. Identifying the current state of fish stocks within estuarine systems can help management initiatives through comparisons under different management schemes (e.g. spatial and temporal closures).

A number of fish are regarded as important angling species found in South African estuaries. *P. commersonnii* and *A. japonicus* are especially important and form a significant proportion of the catches in a number of estuarine systems (Marais & Baird 1980, Kyle 1988, Baird *et al.* 1996). Studies conducted in Eastern Cape estuaries have found that *P. commersonnii* and *A. japonicus* are heavily targeted and comprised 87.4% of the catch in the Swartkops Estuary (Baird *et al*. 1996) and 72.6% in the Great Fish Estuary (Potts *et al*. 2005). Lamberth & Turpie (2001) found *P. commersonnii* and *A. japonicus* contributed 69.0% of the total biomass caught in all estuaries in South Africa. Based on interview data collected by Cowley *et al.* (2013) on the Sundays estuarine fishery; *P. commersonnii* and *A. japonicus* are the two most popular species in the fishery targeted by 51.0% and 44.0% of fishers interviewed, respectively. The two species collectively comprised 46.0% and 67.0% of the catch by number and mass, respectively. Owing to their popularity amongst fishers on the Sundays Estuary, and their stock status (Chapter 2), *P. commersonnii* and *A. japonicus* were chosen as indicator species for the Biological domain.

Table 3.4: Proposed indicators in relation to generic management objectives and specific management issues identified for the Biological domain.

Principle[1]	**Sub-principle**[2]	**Issue**	**Indicator**	**Indicator type (DPSIR)**	**Source**[3]	**Possible action**
B1 & B2	B1.1 B1.2	Stock vulnerability	Threshold population decline	S	IL	Decrease fishing efforts, monitor bag limits, increase size limits and promote closed spatial or temporal areas
B1 & B2	B1.1 B1.2 B1.3	Stock status	SB/R	S	IL	
B1	B1.3	Decline in mean size	Change in size frequency	S	RCS & IL	
B1	B1.3		% undersized fish caught	S	RCS	
B1	B1.1 B1.2 B1.3	Success rate	% successful outings	S	RCS	
B1	B1.1 B1.2 B1.3		% daily bag limits attained	S	RCS	
B1 & B2	B1.3 B1.2 B1.3	Change in catch composition	% target species	S	RCS	

[1]: Major principles taken from the FAO Code of Conduct for Responsible Fisheries

[2]: Sub-principles taken from the FAO Code of Conduct for Responsible Fisheries

[3]: RCS: Roving Creel Surveys, IL: Independent Literature (referenced in Rationale)

***Issue:* Stock resilience to fishing pressure**

Rationale:

It is widely acknowledged (Musick 1999, Lamberth & Turpie 2001, Whitfield & Cowley 2010) that life-history characteristics play an important role in the vulnerability of estuary-dependent species. In addition to estuarine dependence, life-history characteristics such as sex changes, spawning migrations, predictable aggregations, high age at maturity, longevity, residency and high catchability, all contribute to a species vulnerability to overexploitation (Musick 1999, Lamberth & Turpie 2001). Following the set of risk criteria proposed by Musick (1999), Childs (2011) found that *A. japonicus* scored 'very low" in terms of vulnerability to fishing pressure with a population decline threshold of 0.7, and *P. commersonnii* was scored as moderate (0.95) (see Chapter 2).

The performance criteria indicating the decline thresholds based on the four suggested categories of productivity used by Musick (1999) (see Chapter 2) have been adjusted to maintain the number of reference categories of each indicator at five.

Performance Criteria:

Indicator:	Very Poor (0)	Poor (1)	Moderate (2)	Fairly good (3)	Good (4)
Threshold population decline (over 10 years/3 generations	0.7	0.85	0.95	0.99	1

***Issue:* Stock status**

Rationale:

According to the FAO Code of Conduct for Responsible Fisheries adopted in 1995, fisheries management should adopt a precautionary approach in the conservation and management of exploited living resources. In doing so, precautionary reference points, which are derived from agreed upon scientific procedure giving information about the state of the resource and the fishery, can be used as a guide for fisheries management (Smith & King 2010). The Linefish Management Protocol (LMP) calls for all plans for linefish species to be developed with regulations based on clearly defined objectives and quantifiable reference points that are assessed through biologically based stock assessments and historical trends in catch and effort (Griffiths *et al.* 1999, Sauer *et al.* 2003, Smith 2005). Most precautionary reference points are based on time series of age dependent models such as various levels of fishing mortalities or biomass (Weyl 1999). In situations where data and age-dependent modelling is poor, catch-per-unit-effort (CPUE) can be used as an alternative estimator of biomass. CPUE data is often challenged by the lack of knowledge of the pre-impact state of the fishery which makes it impossible to set accurate target reference points. To address this, data collected from long-standing marine reserves have been used as proxy for estimates of pristine CPUE (Attwood 2003, Smith & King 2010). However, with regards to this fishery, there is no long-standing time-series of catch and effort data for *A. japonicus* and *P. commersonnii* in the Sundays Estuary and consequently no indication of pristine biomass. The only data available showing CPUE of either of these species under 'un-fished' conditions comes from quantitative assessments of fish communities conducted in Port Ngqura (20km from Sundays Estuary), where fishing is prohibited, and the Mbashe Estuary (approximately 330km from the Sundays Estuary) which falls in the boundaries of the Dwesa-Cweba MPA. The two areas provide protection to many fish species and may experience higher CPUE when compared to other openly fished estuaries. However, using data from these areas warrants serious caution when comparing to assessments conducted in the Sundays Estuary. As a result, it was decided that there are too many fundamental differences between these systems to compare CPUE. Alternatively, it has also been suggested (Weyl 1999) that in the absence of virgin biomass levels, a reference point may be set using the maximum CPUE observed during past years, or the mean CPUE experienced over a period of relatively high CPUE. Marais (1981) provides the earliest assessment of the abundance of fish in the Sundays Estuary, however, the sampling techniques (gill nets) are too different to compare to the most recent fishery assessment (rod-and-line with baited barbless hooks using mud prawn (*Upogebia africana*) or sand prawn (*Callianassa krausii* as bait.) (Cowley *et al*. 2013). It was also argued by Baird *et al.* (1996) that using angler catch data is fraught with difficulties. It was therefore concluded that the level of uncertainty when setting reference points for CPUE is too high, and using CPUE as an indicator for decline in the abundance of these species in inappropriate for this study.

With no long time-series of catch and effort data in the Sundays Estuary and consequently no indication of pristine biomass, spawner biomass-per-recruit (SB/R) models provide the most appropriate stock assessment methods available (Butterworth *et al.* 1989, Punt 1993 Griffiths 1997). Percentage of pristine spawner biomass (SB/R) has been used as a benchmark for estimating the abundance of species when data is limited (Pauly 1997, Griffiths 1997, Pitcher *et al.* 1998, Musick 1999, Lamberth & Joubert 2014). SB/R is a widely used biological reference point, and is expressed as a percentage of virgin biomass (Rochet 2000). The more heavily a stock is exploited, the lower the SB/R.

It has been shown that when the relative SB/R has been reduced to <20-30% of pristine levels, there is a high risk of stock collapse (Griffiths 1997). However, these threshold reference points are only to be used as initial objectives for management, and they should be used with caution considering the large risk associated with low spawner biomass levels (Griffiths 1997). The most current SB/R calculated for *A. japonicus* is between 1-4.5% (Griffiths 1996). Given the increasing exploitation pressures of *A. japonicus* in estuaries and a stock assessment that was calculated almost 20 years ago with no current indication to show any improvements in stocks, these values must be used with caution. The following criteria were set using the scoring intervals applied in Lamberth and Joubert (2014), with the addition of the limit reference point (Winker *et al.* 2015), by which fish stocks were considered critical if below 5%, maintaining the number of reference categories for each indictor at five.

Performance Criteria:

Indicator:	Very Poor (0)	Poor (1)	Moderate (2)	Fairly good (3)	Good (4)
SB/R	<5	5-25	25-40	40-50	50-100

***Issue:* Decline in mean size**

Rationale:

Decreases in the CPUE and size frequency distribution of fished species are known as the most detectable effects of fishing pressure (Jennings & Lock 1996). No clear trend has been observed from the shore angling competition catch data from 1982-1998 from the border region of the Eastern Cape for *A. japonicus* (Pradervand & Govender 2003, Mann 2000, Mann 2013). Similarly, there are also no trends from any long-term data sets observed for *P. commersonnii* (Mann 2000).

1. Dusky kob (*A. japonicus):*

The life history of *A. japonicus* has been well described (Griffiths 1996, see Chapter 2). Early juveniles (< 200 mm TL) have not yet recruited into the fishery. The limit reference point (250 mm TL) was however based on the minimum length distributions of *A. japonicus* caught by hook and line from estuarine environments in KwaZulu-Natal and the South-Eastern and Southern Cape (Griffiths 1996). A previous mean size found in the Sundays Estuary was 632mm TL (Pradervand & Baird 2002), anything below this value would be considered unsustainable (indicative of a performance score below 2). *Argyrosomus japonicus* only reaches sexual maturity at ~ 1000 mm TL. Juveniles (<1000 mm TL) do not generally migrate long distances and remain faithful to their nursery estuary until they reach sexual maturity (Griffiths 1996, Childs 2013). The threshold reference point was therefore set at 1000 mm TL. Other reference points were chosen according to the past and current minimum legal size (400 mm and 600 mm respectively) and formed the moderate interval criteria.

2. Spotted Grunter (*P. commersonnii*)

Selecting appropriate performance criteria for *P. commersonnii* is more difficult because of their dependence, as juveniles on estuaries as nursery habitats. As a result, evaluating a trend in the mean size of *P. commersonnii* caught in a fishery is challenging (Mann 2000). However, the large proportion of undersized fish caught must be addressed.

The limit reference point was set according to the minimum size caught in a telemetry study conducted by Naesje *et al.* (2007) who targeted *P. commersonnii* in the Great Fish Estuary, using gear similar to that used by recreational fishers on the Sundays Estuary (rod-and-line with baited barbless hooks using mud prawn (*Upogebia africana*) or sand prawn (*Callianassa krausii)* as bait.). The minimum size caught by Naesje *et al.* (2007) was 260 mm TL. Since data on the size at which *P. commersonnii* recruit into the Sundays estuarine fishery is limited, a minimum size of 250 mm TL was chosen as the limit reference point. *P. commersonnii* reach sexual maturity between 300 and 400 mm TL (Wallace 1975). For this reason, and considering the legal size limit for *P. commersonnii* is currently 400 mm TL, anything below 400 mm TL was considered 'poor'/unsustainable in the performance criteria, and anything above can be considered sustainable.

A study conducted by Childs *et al. (*2008) compared the relationship between the percentage of time *P. commersonnii* spent in the Great Fish Estuary and length of tagged fish. The results revealed that fish <400 mm TL spent significantly more time in the estuary and fish ≥500 mm TL spent significantly less (~50%), but equal proportions of time in the estuary. Therefore, 500 mm TL was used as the upper threshold reference point for this indicator, because once mature, *P. commersonnii* spend considerably more time at sea.

The second indicator used to address the issue of a decline in mean size of both *P. commersonnii* and *A. japonicus* was the percentage of undersized fish caught. This indicator was based on the findings of the most recent survey conducted on the Sundays Estuary (Cowley *et al.* 2013), and its rationale is based on the information above.

Performance Criteria:

Indicator:	Very Poor (0)	Poor (1)	Moderate (2)	Fairly good (3)	Good (4)
***A. japonicus*: Mean size (mm)**	<250	250-400	400-600	60-80	800-1000
***P. commersonnii*: Mean size (mm)**	<250	250-400	400-450	45-50	>500
Percentage (%) of undersized fish caught: *A. japonicus* & *P. commersonnii*	80-100	60-80	40-60	20-40	0-20

***Issue*: Success rate**

Rationale:

Fishing impacts fish communities in a number of ways but the most obvious is the removal of individuals (Heino & Godø 2002, Polunin 2002, Smith & King 2010). Indirect assumptions regarding the abundance of these two species can be made by monitoring the success rate of fishers. The indicators chosen for this issue included the percentage of times that fishers had been successful, which was taken as the percentage of fishers that had caught one or more individuals of each species. The second indicator was the percentage of outings where the angers daily bag limits were attained.

Performance Criteria:

Indicator:	Very Poor (0)	Poor (1)	Moderate (2)	Fairly good (3)	Good (4)
Percentage (%) of successful trips	0-20	20-40	40-60	60-80	80-100
Percentage (%) of angler outings reaching their daily bag limits	0-2	2-5	5-10	10-20	>20

Issue: **Change in catch composition**

Rationale:

Most estuarine fish are either fully or overexploited, owing to an increase in the number of fishers and the advancement in fishing gear and mechanisation of boats. The effects of increases in fishing pressure on target organisms include a decrease in their abundance and a change in the size and species composition (Cowley *et al.* 2013). Shifts in catch composition may be a strong indicator of an unsustainable fishery (Pradervand & Baird 2002). A decrease by number and mass of target species such as *P. commersonnii* and *A. japonicus*, may indicate a decline in their abundance in the estuary, resulting in a shift in pressure to a less-desired species (such as *R. holubi*) or size classes (Pradervand & Baird 2002).

Catch composition in estuaries have been recorded in numerous studies (Marais 1981, Clarke & Buxton 1989, Coetzee *et al.* 1989, Baird *et al.* 1996), however the method of sampling employed in these respective studies may lead to unsuitable comparisons. For this reason, the catch composition data recorded during the roving creel survey conducted by Pradervand & Baird (2002) was used as the benchmark to which the most recent survey by Cowley *et al.* (2013) can be compared. Furthermore, both data sets include retained and released fish, whereas previous studies utilised only retained fish.

It is difficult to set limit and threshold reference points for these indicators without appropriate historic data showing the catch contribution by *P. commersonnii* and *A.*

japonicus in the Sundays estuarine fishery. Therefore a change in the proportion of total catch by each species between the two surveys was used (i.e. no between surveys would indicate a sustainable trend, whilst a change of 50% or higher would indicate an unsustainable trend).

Performance Criteria:

Indicator:	Very Poor (0)	Poor (1)	Moderate (2)	Fairly good (3)	Good (4)
Change (%) in contribution to catch by number	>-25	-5-(-)25	0-5	5-25	>25
Change (%) in catch contribution by mass (kg)	>-25	-5-(-)25	0-5	5-25	>25

3.4 RESULTS

The Sundays Estuary linefishery indicate that the fishery is in a poor state and is considered unsustainable, with an overall sustainability index of 23.8% (Tables 3.5)

Table 3.5: Current scores obtained by each domain associated with the Sundays estuarine fishery, highlighting the present un-sustainability.

Social	Institutional	Biological	
		P. commersonnii	*A. japonicus*
		25.0%	12.5%
40.0%	**12.5%**	**18.8%**	
			Total: 23.8%

3.4.1. Social domain

The social domain scored the highest, with an overall sustainability index of 40.0% (Table 3.6, Figure 3.2). However, the low percentage (13.5%) of fishers who knew current management regulations and percentage of undersized fish retained (47.0%) needs to be addressed. The number of subsistence fishers (7.1%) and fishers whose catch is crucial to their diet (11.2%) is low. Only 20.4% of the total fishers were considered as locals of the fishery (resided <5km). This may have implications when including stakeholders and local communities in management plans.

Table 3.6: Sustainability matrix of the proposed Social indicators showing the current scores obtained for the Sundays estuarine fishery.

Issue	Indicator	Current Value	Score
Poor regulatory knowledge and admitted non-compliance	Percentage of fishers who knew the current linefish regulations	**13.5**	0
	Percentage of fishers who admit to breaking the current linefish regulations	**29.7**	3
	Percentage of undersized fish retained	**47.0**	1
Dependence on fish as food	Percentage of subsistence fishers in the Sundays Estuary	**7.1**	4
	The proportion of fishers whose catch was regarded as 'crucial' in their household's diet	**11.2**	4
Sense of belonging	Proportion of fishers that reside <5km from the Sundays Estuary	**18.6**	0
Total:			**12/20 (40%)**

The amoeba plot in Figure 3.2 indicates that where completely coloured in (in blue) there is sustainability of that indicator. The lower score reflects issues relating to unsustainability of that issue. This would indicate that the issues of local residence, poor regulatory knowledge and non-compliance add to the unsustainable social domain.

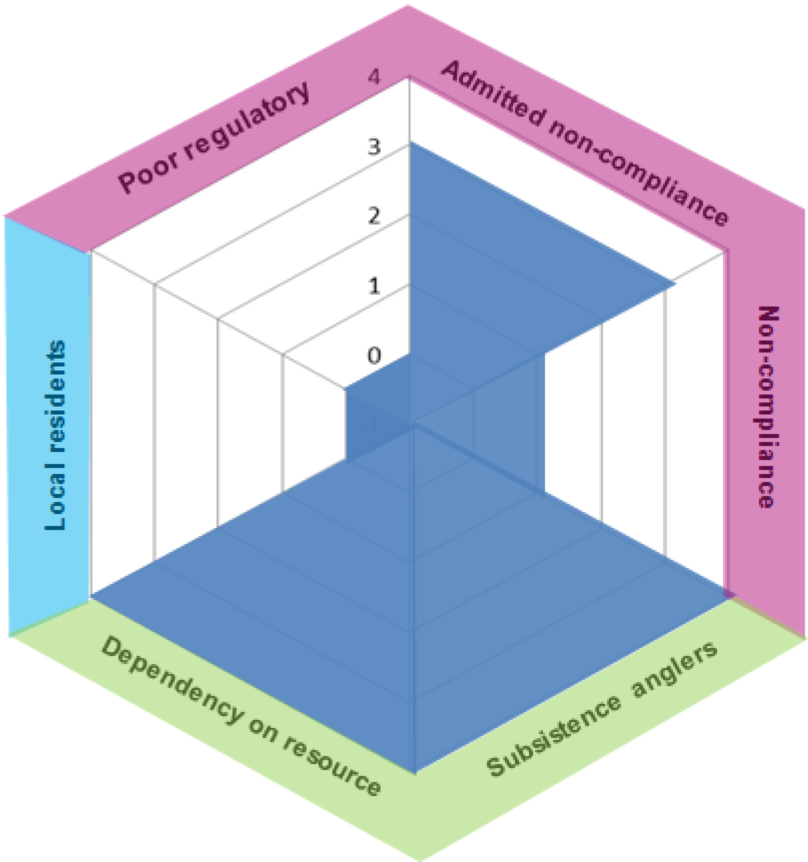

Figure 3.2: Depiction of the overall social sustainability in the Sundays estuarine fishery.

3.4.2. Institutional domain

Institutionally, the management of the estuarine fishery was low with a sustainability index of 12.5% (Table 3.7). The lack of a sound management plan for individual species, and the absence of law enforcement contributed to the low score. The issue relating to ongoing monitoring programs was the only indicator to score in this domain. It did, however, only score as ‘poor’ because very few surveys have been conducted in the past, and none have had any sort of follow up or monitoring since. Each of these indicators needs to be addressed in order for this domain to become sustainable.

Table 3.7: Sustainability matrix of the proposed Institutional indicators showing the current scores obtained for the Sundays estuarine fishery.

Indicator	Reference criteria	Current value	Score
Effective estuarine management plan	The existence of a management plan for the Sundays estuarine fishery with a scientific justification, with particular reference to vulnerable fish species	**Only national bag and size limit restrictions**	1
Adaptable management plan	A presence of ongoing monitoring programs that gather data used to assess and update management strategies	**Occasional surveys have been conducted (Marais 1980, Baird 1986, Pradervand & Baird 2002, Cowley *et al.* 2013)**	1
Effective enforcement	Proportion of fishers that had been inspected within the last 100 fishing outings	**6.8**	0
	Percentage of times law enforcement was witnessed during survey period	**4.4**	0
Total:		**1/16 (12.5%)**	

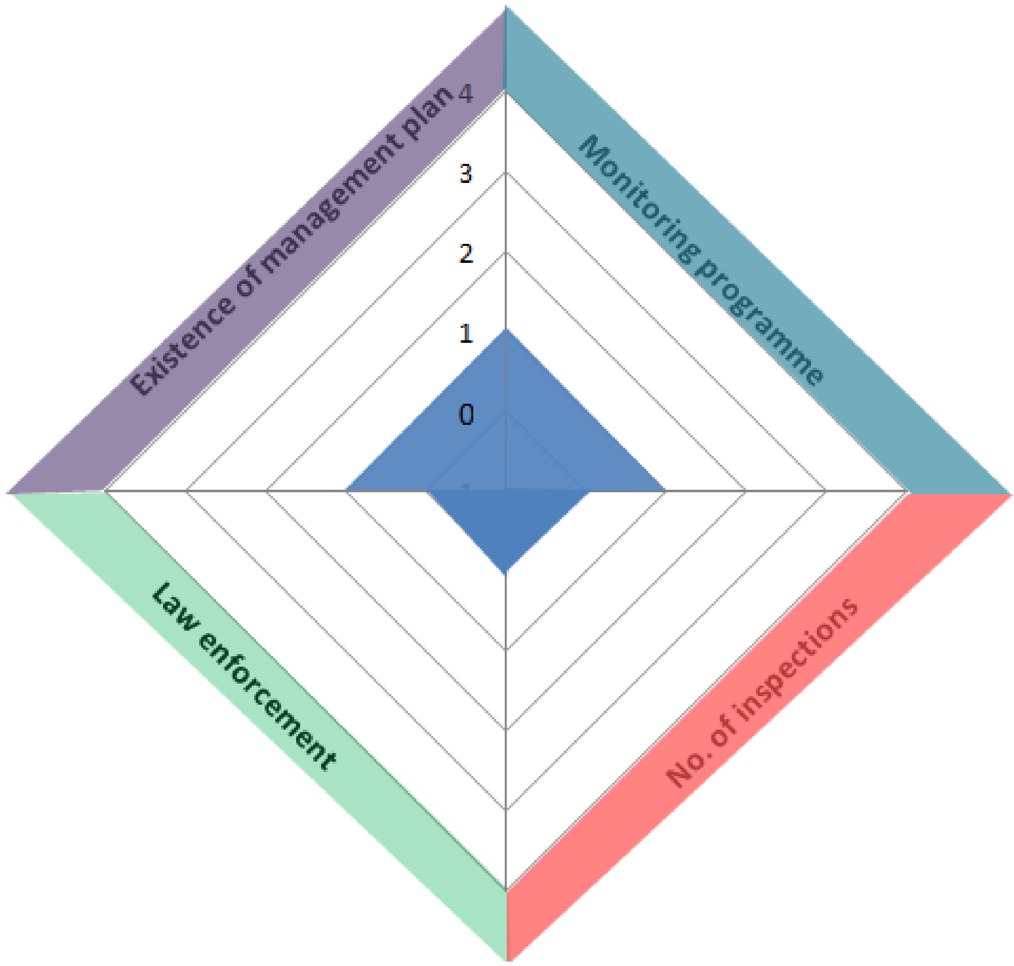

Figure 3.3: Depiction of the overall institutional sustainability in the Sundays estuarine fishery.

3.4.3. Biological domain

In terms of the biological state of the fishery, where a species-specific assessment was conducted, the overall score was low (18.8%) (Table 3.8). Although both target species are in a poor state, *A. japonicus* scored much lower (12.5%) than *P. commersonnii* (25%) (Table 3.8 and Figure 3.4, Figure 3.5). *A. japonicus* low score can be attributed to the low resilience of this species to risk (0.7), and its alarmingly low stock status (SB/R) (<5%). The life-history of *P. commersonnii* resulted in a moderate resilience score (0.95), however, it is still regarded as unsustainable due to the difficulty in attaining daily bag limits (0) and unsuccessful angler outings where zero fish were caught.

Table 3.8: Sustainability matrix of the proposed Biological indicators showing the current scores obtained for the Sundays estuarine fishery.

Issue	Indicator	Current value		Score	
		A. japonicus	***P. commersonnii***	***A. japonicus***	***P. commersonnii***
Stock resilience	Threshold population decline	**0.7**	**0.95**	0	2
Stock status	SB/PR	**1.0-4.5%**	**<25%**	0	1
Decline in mean size	Change in mean size (cm)	**35.9**	**-**	1	-
		-	**31.4**	-	1
	Percentage of undersized fish caught	**87.5**	**73.8**	0	1
Success Rate	Percentage of outings where fishers were unsuccessful	**3.4**	**7.5**	0	0
	Percentage of fishers attaining their daily bag limits (per trip) of legal size fish	**2.6**	**0.1**	1	0
Change in catch composition	Change in composition of target species by number	**1**	**-19**	2	1
	Change in composition of target species by mass	**-27**	**1**	0	2
Total:				**4/32 (12.5%)**	**8/32 (25%)**

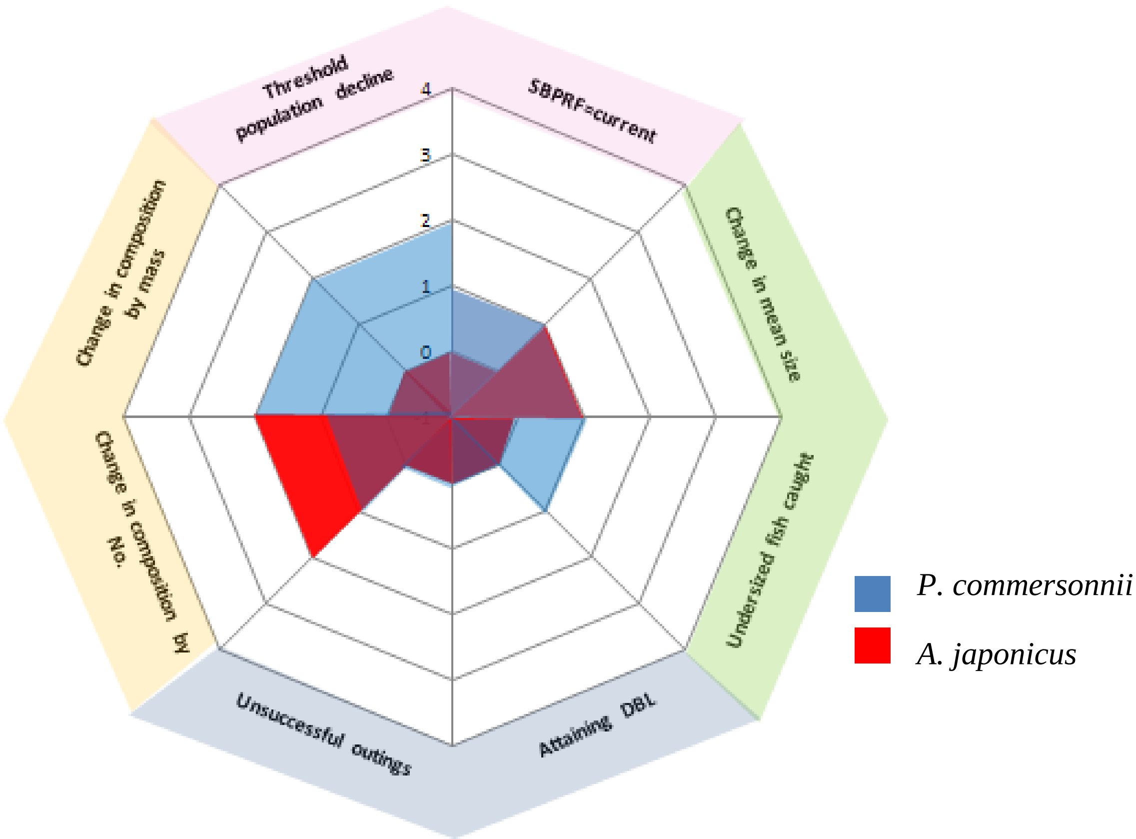

Figure 3.4: Depiction of both *P. commersonnii* and *A. japonicus* overall sustainability in the Sundays estuarine fishery.

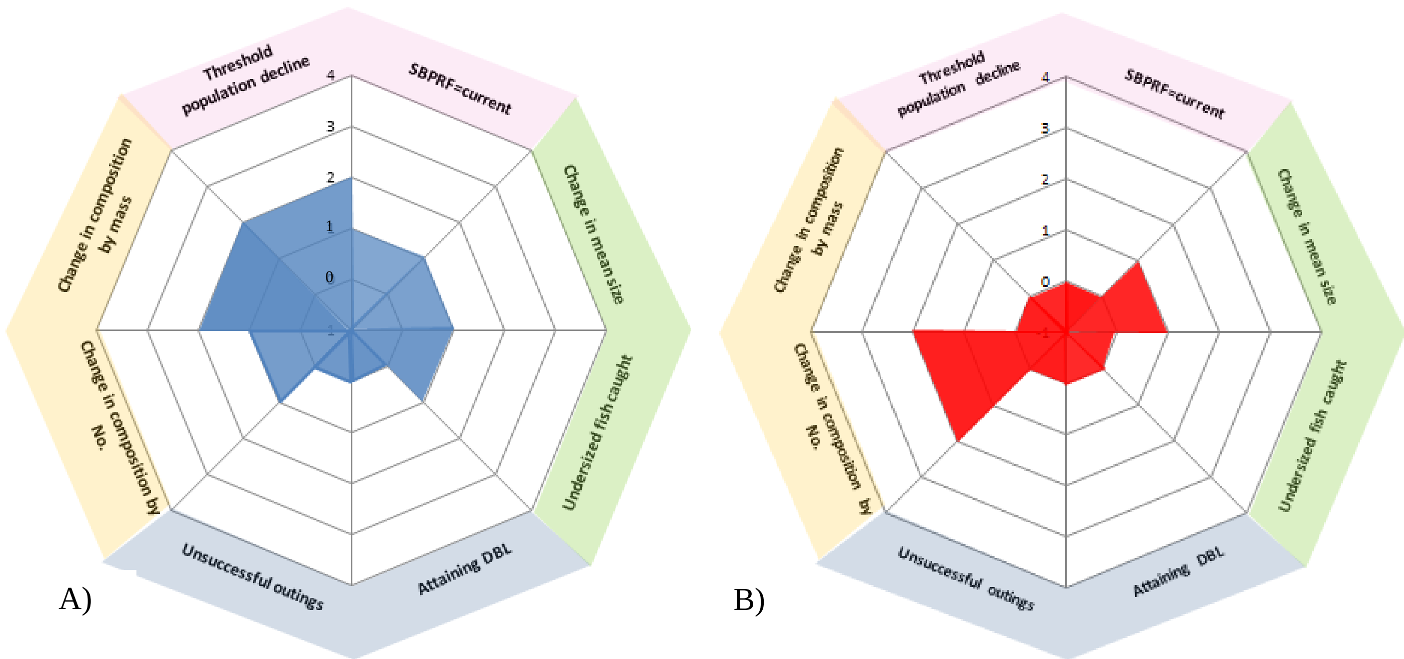

Figure 3.5: Depiction of A) *P. commersonnii* and B) *A. japonicus* overall sustainability in the Sundays Estuarine fish.

3.5 DISCUSSION

The sustainability of estuarine fisheries relies on a comprehensive understanding of the state of fishery resources and the adverse effects of over-exploitation within estuaries, as well as carefully planned management to prevent degradation of these ecosystems (Cowley *et al.* 2013). Whilst estuaries are well known as popular areas for fishing (for both recreational and subsistence purposes), and the consumptive and non-consumptive use of fishery resources have been documented, there is still a lack of assessment on the sustainability of fisheries in estuaries in South Africa. Using the proposed indicators, this study has shown that the linefishery in the Sundays Estuary is currently unsustainable (23.8%) and is in need of greater management effort.

The sustainability of a fishery is scored according to indictors which are grouped in three domains. Overall sustainability is reached through simultaneous achievement of all three domains. Within each domain, indicators should be prioritised according to their individual scores, with management efforts targeting indicators which have lower scores contributing to the unsustainability of a system. The development of indicators needs to be reinforced by data availability (FAO 1995). Data availability is a major issue in the selection of indicators. The data chosen to represent a system needs to be adaptable at a regional and local level and must have requirements that can be met by other fisheries managers.

Indicators are often used as surrogates to monitor changes in the condition of a certain parameter which is not easily measurable (Noss 1990, Chalmers 2012). Selecting appropriate indicators that assess the performance of management actions is fundamental to identify whether management objectives are being met, and allows for future improvement (Pomeroy *et al.* 2005). Indicators were selected for the ‘Pressure’ and ‘State’ categories based on the base-line assessments of fishery activities by Cowley *et al.* (2013). The Sundays estuarine fishery is dominated by recreational fishers. Recreational fisheries in South Africa are open access and there is no limitation on the number of participants, which places increased pressure on targeted resources (Chalmers 2012, Turpie & Goss 2014). By monitoring the trends in angler behaviour and effort, critical aspects of the pressures they place on marine resources can be evaluated. Monitoring the changes in species composition and mean size provide good indications of the pressures exerted by recreational fishers (Smale & Buxton 1985). Changes in the catch composition of fisheries in South Africa have been reported in the shore fishery (Brouwer *et al*. 1997, Maggs *et al.* 2015), and indicate serial overfishing with a shift in pressure to new species (Chalmers 2012). The relationship between the pressures from the social domain directly affects the state of the fishery, and the indicators chosen for this assessment can give direct information about where institutional efforts (response) should be concentrated.

The social domain (40%) scored the highest out of the three domains, which was to be expected as the fishery is dominated by recreational fishers (92.9%). Since these fishers fall into the upper educational and income brackets (see Chapter 2), it is thus assumed that their basic human needs (food, income and employment) are being met according to the generic principles of the FAO Code of Conduct for Responsible Fisheries (S1, Table 3.1) (Garcia 2000, Smith & King 2010). The low number of subsistence fishers in the Sundays estuarine fishery, suggests that there are few fishers below the poverty line, and therefore fishing activities are not based on the dependency of fish for sale or consumption, and sustainable practices such as catch and release can be maximised. It must be noted that the reason for the low level of subsistence fishers could be related to affordability, logistics and access to the fishery. If these obstacles were removed, there may be a different outcome. This needs to be addressed when management plans are designed.

The findings of the survey revealed that the recreational sector had a higher infringement of size and bag limits compared to the subsistence sector (Cowley *et al.* 2013), and necessitated the inclusion of non-compliance issues into this assessment. The number of fishers who knew the current linefish regulations was very poor (13.5%). This issue could be either educational, where the fishers are not informed about current regulations, or ethical, where they have not made an effort to learn the regulations. It is more likely to be an ethical issue because of the high education level of fishers in this fishery. Ethical issues that affect fisheries need to be dealt with in a holistic manner (FAO 2005). This implies that ethical issues such as non-compliance need to be seen in an interconnected manner with institutional aspects so that decision making requires a dialogue that includes the communities committing these ethically questionable acts in such a way that they can be made aware of the uncertain risks (FAO 2005). This issue could be addressed by implementing educational drives that not only inform the fishers of current linefish regulations, but increase their understanding of why the regulations are in place. Not knowing the current regulations would explain the high percentage of retained fish (47%), which could also be remedied by constant reminders of the current linefish regulations through signs, enforcement and educational initiatives. The number of fishers who admitted to not comply with the regulations was fairly low (29.7%). The sensitive nature of non-compliance, including fear of punishment, may reduce the likelihood that fishers will self-report their violations, and lead to bias results with information being withheld or misinterpreted (Solomon *et al.* 2015). Since this indicator relies on fishers truthfulness and may be an under representation of true non-compliance, it should therefore be used in conjunction with the percentage of undersized fish caught and retained. When looking at management initiatives of recreational estuarine fisheries, increased licence fees may be an attractive choice to manage recreational fishers. This would hopefully reduce fishing effort of some recreational fishers to an extent.

Within the framework proposed by Pajak (2000), the behaviour of individuals is central to achieving sustainability of a natural resource. The components of this framework relate to a sense of belonging and self-actualisation among fishers which increases their participation within the fishery and its management. If local residents access the local fishery resources, they will have a vested interest in its conservation. The indicator showing the number of 'local' residents (residing less than 5km from the estuary), participating in the fishery was very low (18.6%). Educating local residents and including them in decision making and management practices, might increase the voluntary compliance of fishers because of their sense of belonging for the estuary. Furthermore, there is a growing body of research that recognises the need to communicate and involve stakeholders in fisheries management decisions (Dedual *et al*. 2013). In order to improve communication, fishermen need to share their perspectives and attitudes towards the fishery and scientists and resource managers need to be aware of their diverging opinions (Dedual *et al.* 2013).

A South African context

One of the primary issues that affect the sustainability of many fisheries management projects, are inadequate administrative and legal frameworks (Griffiths & Lamberth 2002, King 2005). Consequently, the conservation of fish within estuaries needs appropriate legislation pertaining specifically to the activities taking place within the catchments and within the boundaries of the estuary that directly impact the fish species. The Sundays Estuary is managed by the Nelson Mandela Bay Metro, and is not a Marine Protected Area

(MPA) or National Park (Lee *et al.* 2013). The laws that relate to the Sundays Estuary are often breached and poorly enforced. This can be seen in the low scores in the social domain relating to non-compliance, where the number of inspections of fishers and presence of law enforcement scored very low (0). According to the results from the Sundays Estuary situation assessment compiled by Bezuidenhout *et al*. (2011), only one river control officer, who is responsible for ensuring policy of fishing and boating activities, is stationed at the estuary. The current study has revealed that the institutional forces governing the Sunday's estuarine fishery are currently unsustainable (12.5%). In terms of the presence of a management plan, under the Integrated Coastal Management Act, a National Estuarine Management Protocol has been drafted, in which guidelines are provided which aid in specific estuarine management plans (Cowley *et al.* 2013). In the NBSA (2012) overall estuarine population targets were set for overexploited (a target of 40%) and collapsed (50%) species. However, to date, no specific management plan has been established for the Sundays estuarine fishery. The lack of law enforcement promotes non-compliance, which negatively impacts local fish stocks (Cowley *et al.* 2013). Despite there being a lack of long-term biological and socio-economic data pertaining to the fishery, one indicator, relating to the presence of an on-going monitoring programme, managed to score (1/4) and this was only given a score because of the previous surveys that have been conducted, albeit sporadically, by different researchers (Marais & Baird 1980, Baird *et al.* 1996, Pradervand & Baird 2002, Cowley *et al.* 2013). However, the differences in sampling techniques between surveys did not allow for suitable comparisons. The existence of a monitoring programme for the fishery, in terms of resource users and the status of the fish species is essential in ensuring that management strategies are effective.

Ecological integrity relates to the degree to which the ecosystems elements (i.e. species, habitats and natural processes) are functioning in ways that ensure their sustainability to changing conditions and the impacts resulting from human activities (Pajak 2000). According to the National Biodiversity Assessment (2011), the current health of the Sundays Estuary is fairly healthy ('C' rating) and it was recommended that it be upgraded to an 'A' rating. However, this does not look at the current health of fish stocks that are being exploited in the fishery itself. Whilst ecosystem sustainability has been assessed, species-specific sustainability is still less known, and up to now has been largely ignored. The present study revealed that the biological components of the Sunday's recreational fishery are currently unsustainable (18.8%), with *A. japonicus* scoring lower (12.5%) than *P. commersonnii* (25%) (see Chapter 2). The low scores for *A. japonicus* can be ascribed to the species low population threshold decline (B1, Table 3.6) and collapsed stock status (B2, Table 3.6). The life-history of *A. japonicus* makes it extremely vulnerable to over-exploitation; more so than *P. commersonnii.* Since linefish species are managed as a single stock, according to the Linefish Management Protocol (Griffiths 1999), their stock status is a depiction of the entire stock, and as a result, the differences in coastal and estuarine sustainability need to be addressed.

The percentage of undersized fish caught (both retained and released) was high for both species; however, *A. japonicus* scored 0 for this indicator because 88% of the fish caught were below the legal size limit (600 mm). This demonstrates that the likelihood of growth over-fishing (when fish are caught at a size that is smaller than the size that would produce the maximum yield per recruit) in estuaries cannot be ignored. Griffiths (1997) recognised the collapse of the *A. japonicus* stock to the over-exploitation of juveniles, and considering that *A. japonicus* attains sexual maturity >1000 mm TL (Griffiths 1996**)**, growth over-fishing within estuarine habitats is to be expected (Cowley *et al*. 2009).

The paucity of information on unexploited estuarine fish stocks creates additional problems. Very little information was collected before the estuarine fisheries were developed, and as a result, it is difficult to compare the current status to how the stocks looked prior to exploitation. Often, there is a risk of using a situation already influenced by fishing as the base-line for evaluating further change (e.g. comparing the present study to Pradervand & Baird's survey in 2002) (Gislason 2001). Regardless of this, the local abundance of fish has been reduced, which is evident by the poor success rate of fishers (indicators B4a & B4b) and decline in mean size of target species (indicators B3a & B3b). The mean size of an over-exploited species (observed for both *A. japonicus* and *P. commersonnii*) can be expected to decline when there is an increase in fishing pressure (Baird *et al.* 1996). The observed decline in mean size can also be attributed to the selective nature of hook and line fisheries where the larger fish are targeted first (Baird *et al*. 1996). It must be noted, however, that although there is a decrease in the mean size of both species between the two studies, Pradervand and Baird (2002) used law enforcement officers to collect data during their survey. Consequently, it is likely that fishers did not reveal their undersized fish. The observed change in composition of the target species can also be an indicator of decline in stock. *A. japonicus* and *P. commersonnii* are heavily targeted species in many Eastern Cape estuaries (Cowley *et al.* 2013), and the Sundays Estuary is no exception. This indicator included both the change in catch by number and mass of *P. commersonnii* (-19 and 1) respectively and *A. japonicus* (1 and -27 respectively) between the chosen surveys. It is important to look at both criteria because of the differences in the life-histories of *P. commersonnii* and *A. japonicus* which may affect the number and mass of the catch composition differently. Like most sciaenids, *A. japonicus* exhibits a delayed maturity, with a length-at-50%-maturity of 1000 mm TL for females and 920 mm TL for males (Griffiths 1996). Owing to their prolonged juvenile phase, when growth overfishing occurs, even though the number of fish may remain constant, the contribution in terms of mass by *A. japonicus* may differ dramatically because of the number of juvenile fish caught with a larger variation in size of juveniles and adults. *P. commersonnii*, on the other hand, are fast growing, and the length-at50%-maturity occurs at 300 mm TL in males and 360 mm TL in females (Wallace 1975). As a result, the difference in mass of juveniles and adults caught in the estuarine environment is not as drastic as that exhibited by *A. japonicus*, and therefore the change in catch composition in terms of mass might not change as much as the contribution by number of *P. commersonnii.*

Whilst *P. commersonnii* and. *A. japonicus* are the two most targeted species in the fishery, *Rhabdosargus holubi*, which is considered as a by catch species, is becoming increasingly dominant (30.1%) in the catch. Pradervand & Baird (2002) found *R. holubi* to be more prevalent in their catches when compared to an earlier survey conducted by Marias and Baird (1980), and in a fishery assessment conducted on the Kowie Estuary, *R. holubi* was found to contribute 62% of the catch in numbers (Nsubuga 2004). The increase of smaller species such as *R. holubi* might be a result of a decrease in the larger and more targeted species (Pradervand & Baird 2002). Whilst this species stock status is considered to be in a healthy state, it might need to be included in future assessments because of its increasing importance in the catch. Another species that did not dominate the catch and is not targeted in the fishery, but is considered to be in a collapsed state is *Lithognathus lithognathus*. Juvenile *L. lithognathus* have an obligatory estuary-dependent nursery phase (Wallace *et al.* 1984). They form major components of shore catches in the costal fishery (Mann 2000), and have historically formed a large component in recreational and competition catches in certain estuaries (Marais & Baird 1980), however recreational boats in estuaries land low numbers of *L. lithognathus* annually (Mann 2000). This species had the highest proportion of retained undersized fish (100%) and is of major concern. Whilst *L. lithognathus* was not included in this sustainability assessment, its on-going exploitation and collapsed stock status warrants its inclusion in future management plans.

It is clear that human influences have significantly impacted ecosystems and species that were once regarded as pristine. Furthermore, changes in the structure and diversity of fisheries, such as the role of top predators have changed dramatically because of large-scale declines (Baum & Myers 2004). In many cases, long-term data of fisheries are lacking and historical perspectives are bias by the most recent dataset. Without prior knowledge of baseline information and pristine values, researchers are at risk of becoming content about the rarity of a species and might accept the present as natural (Pauly 1995, Baum & Myers 2004). Knowledge of previous data can provide a source of data for conservation of rare or threatened species, but researchers need to be aware that this may change over time (Turvey et al. 2010). This is known as a shifting baseline and needs to be considered into management decisions.

The continued decline of fisheries species in South African estuaries and the ineffectiveness of current management methods have been highlighted, together with an increase in effort and poor angler knowledge, the future of the Sundays estuarine fishery is questionable. An alternative management approach with clear management objectives and the involvement of local communities, will aid in the sustainability of future catches and stocks of over-exploited and collapsed targeted species in the Sundays Estuary. Estuaries in South Africa differ in their biological characteristics, resource-use activities, socio-economic situations and demographics of resource users, and as a result, management initiatives need to be unique to individual systems (Cowley *et al*. 2013). Sustainability frameworks are estuarine specific and need to be modified depending on certain requirements for each estuary. Managers need to adopt an estuary-specific management approach and conduct individual sustainability assessments like this one in order to highlight priority issues.

This study has shown Sundays estuarine fishery to be non-sustainable at present levels of exploitation due to the poor institutional capacity by the organisations responsible for fisheries management. An alternative management solution is imperative for future generations to continue benefiting from the estuarine environment. The low proportion of subsistence fishers suggests that management responses should include the adoption of spatial-based area management, with a catch-and-release option to ensure increased survival of estuary-dependent juveniles and recovery of adult breeding populations.

In order for effective management measures to be implemented for the Sundays estuarine fishery, an improved understanding of the ecology of vulnerable estuary-dependent species in the estuarine environment, as well as the fishery resource utilisation is required.
Since there is no management plan at present to guide the utilisation of the Sundays estuarine fishery, the following chapter proposes an approach for a spatial-based management plan using high resolution acoustic telemetry which can help identify ecologically important areas for protection of over-exploited and collapsed estuary-dependent species.

CHAPTER 4: ASSESSMENT OF A SPATIAL-BASED MANAGEMENT APPROACH FOR THE SUNDAYS ESTUARINE FISHERY

4.1 INTRODUCTION

Estuaries serve as critical nursery habitats for a number of species that form important components of recreational, subsistence and commercial fisheries in South Africa (Beck *et al*. 2011, Cowley *et al.* 2013). Although estuaries provide many economic and social benefits, management of these ecosystems and their associated fisheries has been largely inadequate and as a result, the stocks of many estuary-dependent fishery species are either over-exploited or collapsed (Griffiths 2000). The results from the sustainability assessment conducted on the Sundays estuarine fishery (Chapter 3) highlighted the ineffectiveness of current management regulations, the lack of compliance to the regulations and their consequent effects on the biological resources of the fishery. In South Africa there are limited resources dedicated to the enforcement of recreational fisheries regulations in estuaries (Cowley *et al.* 2013) and alternative management measures such as spatial and temporal regulations that require less enforcement could provide a viable option forward (Childs 2013).

Effective spatial-based management approaches have been developed through the successful implementation of Marine Protected Areas (MPA), and the concept has been well established in South Africa and elsewhere (Chapter 1). MPAs are holistic in their nature, and provide protection of spawner stock, the opportunity for a recruitment source for surrounding areas and the restocking of adjacent areas through adult migration (Whitfield & Cowley 2010). The goals of MPAs can range from conserving species to supporting sustainable fisheries (Lester *et al.* 2009). Many marine conservation efforts have focused on the designation of protected coastal and oceanic systems, whilst often largely ignoring their estuarine components. Since estuaries have been recognised as nursery areas for many important fish species, it is surprising that these systems have not yet been purposefully included in the selection of marine reserves (Whitfield 1998). The application of closed areas as a fisheries management tool is controversial. It is imperative in management that closed areas are designed to significantly reduce overall fishing mortality of vulnerable life-history stages of target species (Hunter *et al*. 2006).The implementation of Estuarine Protected Areas (EPAs) could benefit these species and reduce growth overfishing (the capture of fish before they have realised most of their growth potential) in their nursery habitats (Griffiths 1996). There is a need for an expansion of existing protection of estuaries, as well as the upgrading of selected estuaries where activities are zoned. In Australia, estuaries are included in the general category of marine reserves and 288 Marine and Estuarine Protected Areas (MEPA) have been established, in which activities are zoned (Rigney 1990). In the USA, estuarine protection is well established, with their National Estuarine Research Reserve System (NERSS), estuaries are afforded special management requirements (Attwood *et al.* 1997).

The well-documented success of marine protected areas (MPAs) in South Africa (Bennett & Attwood 1991; Cowley *et al.* 2002; Maggs *et al.* 2015; Kerwath *et al.* 2014; Mann *et al*. 2015) and the inclusion of estuarine protection in Australia and the USA should apply equally to estuaries in South Africa (Whitfield & Cowley 2010). In order for the implementation of EPAs in South Africa, a comprehensive spatial planning assessment of the habitat, species distribution (Turpie 2004), institutional arrangements and socio-economic consequences, are required (Childs 2013).

Currently, the need for estuarine protection in South Africa has been realised through the National Spatial Biodiversity Assessment Plan (NSBA) (Turpie 2012), which assessed estuaries on a broad scale, rather than a detailed estuary-level approach, but recognised the need for individual estuarine management plans, following the proclamation of the Integrated Coastal Management Act. Recent research on the movements, space use patterns and habitat connectivity of estuary-associated fishes (e.g. Cowley *et al.* 2008, Childs *et al*. 2008; Bennett *et al*. 2015) has provided a better understanding of the need for effective spatial management of estuarine fishery species, which needs to be considered at a local estuary-specific level (Gillanders *et al.* 2012, Childs 2013).

4.1.1 Review of existing marine and coastal/estuarine spatial planning methods

Marine and coastal management involves dealing with the controlling of human use activities in an environmental context, looking particularly at where activities overlap or impact the ecosystems in a negative way (Harris 2012). Conservation planning is defined as “the process of locating, configuring, implementing and maintaining areas that are managed to promote the persistence of biodiversity and other natural values (Pressey *et* *al.*2009). Conservation and management are not the same, but they are also not mutually exclusive, and true ecosystem-based management strategies need to incorporate conservation goals (Harris 2012). This process has become increasingly important in the attempt to amend the costs and biodiversity losses incurred by previous *ad hoc* allocations of protected areas (Pressey & Tully 1994, Carwardine *et al*. 2010). The necessity for conservation tools that aid in the decision making process has led to the development of many approaches (Carwardine *et al*. 2007) and there are many methods for mapping conservation priorities and management actions in marine systems (Allnutt *et al.* 2012).

Traditionally, conservation plans rely on setting conservation targets, which is a quantitative expression of potential protected areas conservation goals, and defines how big the planning area needs to be to meet these goals. Biodiversity algorithms are then used to find the most efficient solution to meet the defined goals and facilitate the identification of marine and estuarine sites (Ball & Possingham 2000). Various software systems are available to perform the calculations which aid in the decision making process, including Marxan, Marxan with zones, Zonation and C-Plan (Ball & Possingham 2000, Possingham *et al*. 2000). Most of these tools provide 'decisions' about where and what to protect in an ecosystem. The main difference between these software systems is their ability to make and work with planning areas and with numerous datasets at the same time. For example, Marxan utilises a minimum-set approach in which the objective is to achieve the target of each conservation feature, whilst minimising the cost (Ball *et al*. 2009), and Zonation utilises a maximum-coverage approach whereby the objective is to maximise the amount of conservation benefits, given in a fixed budget (Rojas-Nazar *et al*. 2012).

The quantitative use of systematic conservation planning (SCP) for evaluating existing MPAs or identifying priority areas for conservation has increased rapidly in recent years (Chalmers 2012). SCP is now widely accepted and provides transparent and comprehensive methods that support decision making and equip managers and stakeholders with the necessary tools to evaluate options for potential implementation and enforcement. Ensuring effective reserve systems entails the implementation of a CARE principle (i.e. Comprehensiveness, Adequacy, Representation and Efficiency) (Possingham *et al.* 2006). Marxan uses the CARE principle to achieve optimal conservation plans for coastal and marine ecosystems.

Marxan is the most popular conservation planning software and is used by over 2600 individuals in 110 countries worldwide (Watts *et al*. 2009, Levin *et al*. 2013). It, along with other software packages like C-plan, uses an algorithm called simulated annealing which aims to achieve conservation target goals in a spatially efficient manner (Geselbracht *et al.* 2009). Simulated annealing selects reserve locations that are superior to those chosen using *ad hoc* methods (Stewart *et al*. 2003). This is based on user-defined conservation features, targets and penalties that the simulated annealing algorithm uses to generate the ‘best’ solution for conservation area (Possingham *et al*. 2000). In this way, the software tries to satisfy all requirements in a spatial context, whilst identifying a reserve system with a minimum cost (Ball & Possingham 2000). Marxan finds good solutions to a problem by comparing alternative solutions. This is achieved through a mathematical objective function (Figure 4.1) that gives a value for a collection of planning units (PUs) based on various costs of the selected set and the consequent penalties for not meeting conservation targets. Planning units (PUs) are parts of the seascape that are analysed as the potential building blocks of a reserve system. They allow for the comparison of different candidate areas. Each PU has its own unique conservation feature amount and cost (Adron *et al*. 2010).

$$\overbrace{\sum_{PUs} Cost} + BLM \underbrace{\sum_{PUs} Boundary} + \overbrace{\sum_{Con\ value} SPF * Penalty} + \underbrace{CostThresholdPenalty(t)}$$

Figure 4.1: Description of Marxan Objective Function (*adapted from* Game & Grantham 2008).

a. The total cost of the reserve network (required)

b. The penalty for not adequately representing conservation features (required)

c. The total reserve boundary length, multiplied by the Boundary Length Modifier (BLM) (optional)

d. The penalty for exceeding a pre-set cost threshold (optional - not generally advised to use)

Letters *a.* and *c.* can be seen as 'costs', whilst letters b and d are 'penalties' for not meeting the specified criteria (Game & Grantham 2008).

Marxan has been used in a number of different marine ecosystems including the Gulf of Mexico (Beck & Odaya 2001), the Channel Islands of California (Sala *et al.* 2002), the Florida keys (Leslie *et al*. 2003) and the Great Barrier Reef in Australia (Day 2008). In South Africa, Marxan has been used in a number of studies, for example; to identify the potential for sandy beaches along the coastline to be included in coastal reserves (Harris 2012), to propose potential closed areas to reduce by catch in the South African inshore trawl fishery (Lombard *et al*. 2010) and to identify focus areas for offshore biodiversity protection in South Africa (Sink *et al*. 2011) to name a few. However, the most important study in the context of this thesis was conducted in 2012, when Marxan was used to conduct a SCP analysis and identify priority areas for conservation in Algoa Bay, Eastern Cape (Chalmers 2012).

In 2011, a National Biodiversity Assessment with an estuarine component was conducted. The analysis used Marxan to prioritise estuaries that should be assigned Estuarine Protected Area (EPA) status. Although social and economic costs were not taken into account, ecosystem health was used as a surrogate for these costs (Turpie 2012). Furthermore, this analysis failed to identify areas for biodiversity conservation within each estuary. To date, there is a lack of studies that look at individual estuarine fisheries protection based on the spatial distribution of both human use activities and the distribution of vulnerable species.

Conservation planning specifically for estuarine and freshwater ecosystems has fallen behind the rapidly growing approaches to marine and terrestrial systems. This is mainly because of the lack of methods to adequately address conservation planning in these systems (Linke, Turak & Nel 2011). Estuaries face unique challenges that are associated with their conservation plans, including their small size, linear structure and the connectivity between marine and terrestrial ecosystems in which key ecological processes such as species movement and migrations are maintained (Hermoso *et al.* 2015) as well as their socio-economic importance to surrounding communities (Blaber 2002). Conservation planning software systems are usually applied over large areas, which make their implementation in small estuaries difficult (Hermoso *et al*. 2015).

Research into spatial management of marine resources requires the ability to investigate the ecology of populations on a small spatial scale in order to examine issues that impact their conservation (Ehrenburg *et al*. 2014). Marine and estuarine protected areas require information about the movement, interactions and connectivity of species at risk. Because movement is species-specific, protected areas should be designed with targeted species in mind (Sale *et al.* 2005, Marshell *et al.* 2011). For example, some species, with large home ranges may spend more time outside the allocated protected area, which could increase their exposure to exploitation (Marshell *et al.* 2011). Quantifying the movement of fish species has benefited greatly from the evolution of acoustic telemetry (Heupel *et al.* 2006). Acoustic telemetry provides detailed information on the movement and behaviour, such as home range, connectivity and seasonal behaviour, over temporal and spatial scales (Marshell *et al*. 2011). It is a powerful tool that provides high resolution fine-scale temporal and spatial data, by continually tracking and monitoring tagged fish (Childs 2013). This fine-scale data provides comprehensive understanding of fish movement behaviour through the acquisition of ecological and biological information that can be used for management and conservation of the studies species (Cooke *et al.* 2004, Childs 2013). Whilst acoustic telemetry has been widely used to quantify fish use of marine protected areas (Parsons *et al.* 2003, Topping *et al.* 2005), its application for estuarine protected areas is less well known.

The life-history and level of estuarine-dependence of the most targeted estuarine fishery species in South Africa are well described (e.g. Griffiths 1996 Whitfield 1998, Mann 2013) (see Chapter 3).As a result of the collapsed stock status of species such as *A. japonicus* and *L. lithognathus*, and the increasing decline of over-exploited stocks of *P. commersonnii* and *L. amia*, recent research efforts, using acoustic telemetry, have focused primarily on estuarine movements, space use patterns and habitat connectivity of these species (Childs 2015, Cowley *et al.* 2008, Bennett *et al.* 2011, Dames 2014, South African Institute for Aquatic Biodiversity (SAIAB) *unpubl. data.*). This information is essential for the development of appropriate management measures for species of fishery importance and the identification of vulnerable habitats for conservation without having to use surrogate data (Bennett *et al*. 2011). Whilst acoustic telemetry is one of the most widely used methods to track fish movement, only a few studies have been able to translate telemetry data into a more relevant approach to design or evaluate MPAs in South Africa (e.g. da Silva *et al*. 2013).

4.1.2 Sundays Estuary case study

SAIAB conducted research on dusky kob *Argyrosomus japonicus* in the Sundays Estuary, using passive acoustic telemetry techniques over a period of three years (2008-2010). These research efforts focused on assessing the estuarine dependency, movements and habitat connectivity of *A. japonicus* during its prolonged juvenile phase (Childs 2013, Dames 2014). The vulnerability of juvenile *A. japonicus* was highlighted by the recapture of 41% of the fish acoustically tagged during the above mentioned study (Childs *et al.* 2015).

The results from the roving creel survey conducted in 2008 by Cowley *et al*. (2013), which formed the basis of the sustainability assessment in Chapter 3, included detailed spatial and temporal patterns of resource use along the estuary. This study provided information on the spatial and temporal distribution of fishing effort and trends in the participation by different sectors (i.e. recreational boat, recreational shore and subsistence).

Making use of fine-scale high resolution acoustic telemetry data of the movements an *A. japonicus* (after Childs 2013 and Dames 2014) together with information of the distribution of fishing effort (after Cowley *et al.* 2013) on the Sundays Estuary in Algoa Bay (Eastern Cape), this study aimed to evaluate the potential of Marxan to assist with the conservation planning and management of *A. japonicus* in the Sundays Estuary.

4.2 METHODS AND MATERIALS

4.2.1 Marxan software: planning domain and planning units

Planning units are the building blocks of a reserve system. They are the units that Marxan evaluates and selects during an analysis in order to form solutions (Pressey & Logan 1998).

Detailed information regarding the Sundays Estuary is given in Chapter 2. The planning domain for the Sundays Estuary was defined as the area approximately from the mouth to 21km up the estuary. For the purpose of the movement study conducted by Childs (2013) Sixteen automated data-logging acoustic receivers (VEMCO VR2 and VR2W receivers) (1-16) were placed along the length of the estuary and moored roughly 1km apart, with the exception of receiver 16 which was 3.6km above receiver 15 (Figure 4.3). The lowermost acoustic receiver (receiver 1) was situated 2km from the estuary mouth and the uppermost (receiver 16) was situated 21km from the estuary mouth (Childs 2013). The study area was divided into 17 planning units (PUs) according to each receiver using a square grid divided with the receiver in the middle of each PU in ArcGIS version 10.2 (Environmental Systems Research Institute) (Figure 4.2). Due to the placement of receivers, the planning unit '0' refers to anything below the first PU (receiver 1, situated 2km from the mouth) and is referred to as the 'sea' planning unit. This PU '0' was added in order to account for the proportion of time spent by *A. japonicus* either in the mouth region, where no receiver was present, or had gone out to sea.

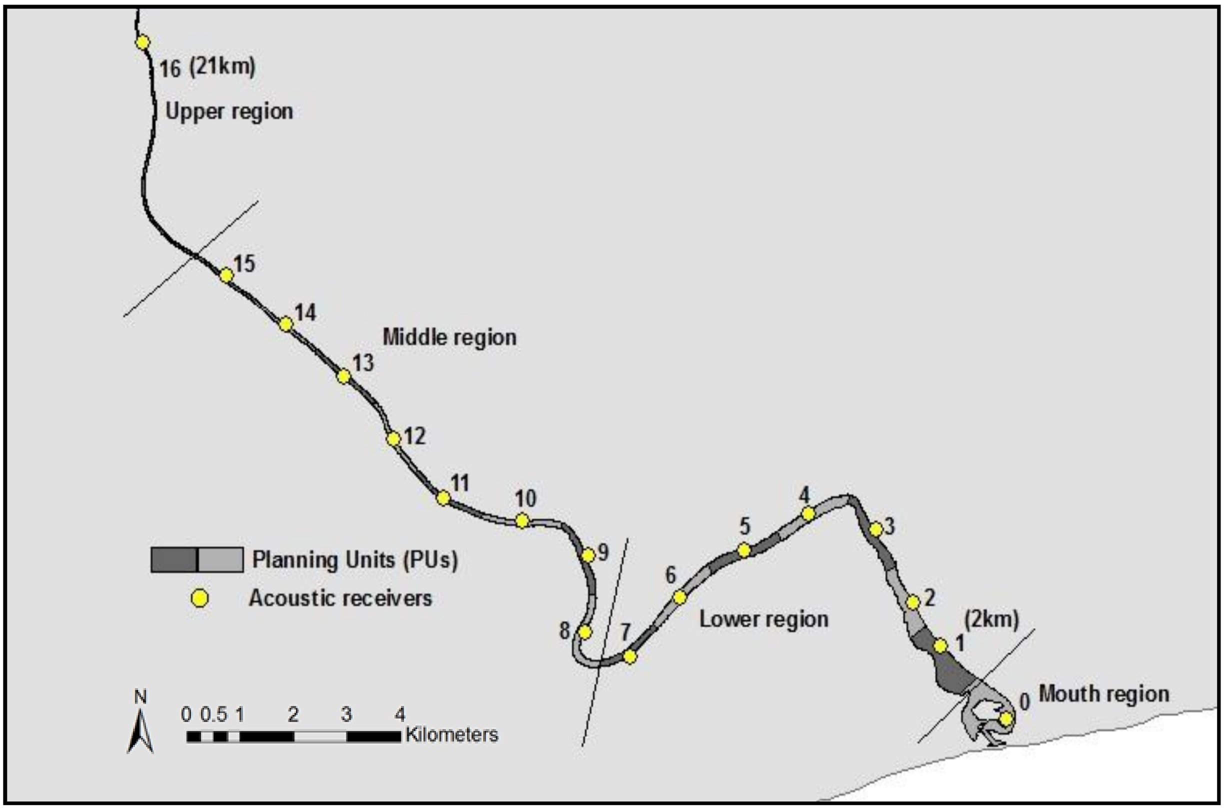

Figure 4.2: The Sundays Estuary showing the locations of the acoustic receiver station within each PU (0-16) and the boundaries of the different estuarine regions.

4.2.2 Conservation features and targets

A total of 66 juvenile, sub-adult and adult *A. japonicus* ranging from 237 to 1110 mm TL were surgically equipped with acoustic transmitters in the Sundays Estuary. Fish were tagged in three temporally segregated batches and their movement behaviour and area use was recorded over a period of three years (2008 – 2010) (Childs 2013, Dames 2014). In order to address the issue of growth-overfishing and non-compliance of regulations pertaining to juvenile *A. japonicus* (<600 mm) (see Chapter 3), legal-sized (>600 mm) larger *A. japonicus* were excluded from the analysis leaving 56 juvenile *A. japonicus*.

4.2.2.1 Conservation feature: Area use by tagged *A. japonicus*

Area use was measured for each fish as a proportion of time each individual spent in the vicinity of each of the 16 receivers (PUs 1-16), and at sea (PU 0 - when the fish left the estuary and returned at a later date), following the methods described by Cowley *et al.* (2008). Two factors that are often considered in MPA literature is the timing of fish movement on a daily and seasonal scale in relation to fishing activity (Kerwath *et al*. 2009). For the diel analysis, the average time spent by each fish within each PU for every hour of the day was calculated, and then hours were combined to give the proportion of time spent at each receiver for day (from 06:00am to 17:00pm) and night (from 18:00pm to 05:00am). Similarly, months were grouped to form two seasons, the warmer months (September-February) and the colder months (March-August) in order to see whether there was a significant different in the area use by the tagged *A. japonicus*. A two-way factorial ANOVA was conducted in STATISTICA (Version 12.0, Statsoft, Inc.) to determine the effect time of the day had on area use by all fish. The data was normally distributed and had equal variance, and therefore met the assumptions of normality and homogeneity required for conducting analysis of variance tests.

The overall, seasonal and diel area use was calculated for each PU and added as a shape file in ArcGIS v10.2. These temporal distributions (diel and seasonal) were used as conservation features in separate Marxan analysis to evaluate any potential differences.

4.2.2.2 Conservation target

Conservation targets indicate how much of a biodiversity feature needs to be conserved (Pressey *et al.* 2003). They are usually selected based on the conservation goals determined through policy, expert opinion, stakeholder interactions or a combination of these (Pressey *et al*. 2003, Chalmers 2012). Targets can change considerably according to the area and system being analysed, and there is on-going debate over how much of a feature is required in order to ensure long-term persistence (Svancara *et al.* 2005). Two targets for *A. japonicus* have been used in the literature in a systematic conservation plan for Algoa Bay, Chalmers (2012) chose *A. japonicus* as a surrogate for near-shore and estuarine species and used a conservation target of 15% due to the low spatial resolution of the available data. The National Biodiversity Assessment (2011) (Turpie *et al*. 2012) used population targets that were based on the number of individuals per species for estuarine dependent fish species. A conservation target of 50% was used for red list species, which included *A. japonicus.*

Biodiversity targets should ideally be set using ecological principles that achieve species persistence (Levin *et al.*2015) and targets are often set using minimum viable population sizes and species/area curves (Adron *et al.* 2010). However, because species vary widely in their spatial requirements, conservation managers may lack the information to set evidence-based targets (Levin *et al*. 2015). This kind of information is often unavailable in the marine realm, and often, the only substitute is expert opinion (Adron *et al.* 2010).

The target set for *A. japonicus* (50%) in the NBA (Turpie *et al*. 2012) was for the entire *A. japonicus* stock although there is currently no direct information on the population size of *A. japonicus* (Childs 2011)and so setting targets for juveniles in a particular estuary is difficult.

Often, when there is doubt about setting a blanket target that has been used for a particular study, a range of targets can be used for Marxan analysis (Loos 2006). For example, Levin *et al.* (2015) used uniform targets from 5-100% and variable targets based on species IUCN class and Lieberknecht *et al*. (2004) looked at targets between 10-40%. By analysing different targets, stakeholders and managers may be able to visualise different solution sizes and compare different scenarios and assess whether conservation plans will differ with changing targets.

Since a target of 50% was set for *A. japonicus* adult stock (SB/PR 1-4.5%) (Turpie *et al.* 2012, Winker *et al.* 2015), and given the high proportion of non-compliance to juvenile *A. japonicus* within the Sundays estuarine fishery (Chapter 3), four conservation targets (50%, 60%, 70% and 80%) were chosen for this analysis.

4.2.3 Costs: Distribution of human activities

Costs are assigned to individual planning units (PUs) and can be based on the planning unit area, economic or social cost, or a combination of these (Lieberknecht *et al*. 2004). A high cost assigned to a PU, will mean that the PU is less likely to be included in the final solution, because the objective is to minimise the overall solution value (Ball & Possingham 2000).

For this analysis, the cost assigned to each PU was a social cost, where the distribution of human activities was taken as the cost layer. Recent literature has highlighted the importance of incorporating social costs into conservation planning in order to minimise the impacts on resource users (Klein *et al.* 2008), and reduce the conflicts between resource users and conservationists (Carwardine *et al.* 2008, Klein *et al.* 2008, Ban & Klein 2009).

The information used to create this cost was taken from the results of the monthly on-site direct-contact roving creel survey and a series of instantaneous estuarine user-counts conducted in 2009 by Cowley *et al.* (2013). This provided detailed temporal and spatial distribution of resource use activities of different fishery sectors which is used as a proxy for fishing pressure.

The spatial distribution of different estuarine use activities was determined through daily instantaneous counts that recorded the resource use activity type, number and demographics of user and the GPS coordinate for each activity (Cowley *et al.* 2013). For the purpose of this analysis, three resource use activities were chosen to represent the cost layer. These included recreational boat fishing, recreational shore fishing and subsistence fishing. The spatial distribution of each activity was analysed in ArcGIS v10.2 to give three shapefiles (UTM-WGS84). Each shapefile was overlaid onto the PU polygon shapefile and the number of points which fell within each PU was counted using the spatial join tool. To determine the relative distribution of fishing effort in each PU, the number of resource users in each PU by the total number of resource users in the fishery (for the three different resource use sectors).

The instantaneous counts spanned a 12km stretch of the estuary, resulting in no data values being captured for PUs 13-16. Spatial analysis requires the cost data to cover the whole area, and results will be bias towards areas with no cost. In order to account for differences in the sampling design between the fish and fisher distribution, the data points for each estuarine activity were imported into ArcGIS v10.2, and the data was Kriged using the spatial analyst tool, to interpolate a cost for the remaining four PUs that were not included in the survey (13-16).

4.2.3.1 Weighting activities

One of the limitations of the current formulation used in decision-support tools such as Marxan is that costs have to be summarised into a single cost unit in one layer (Ban & Klein 2009). Combining multiple costs into one cost layer is one of the biggest challenges in incorporating social data into conservation planning.

Two different scenarios with different cost layers (with costs not weighted, and costs weighted accordingly) were created in order to account for misrepresentation of different fishery sectors. The first scenario's cost layer was created by averaging the distribution of recreational boat, recreational shore and subsistence fishers within each PU equally (Table 4.1.) The second cost scenario weighted the different activities according to their importance and impact on the fishery. A Kolmogorov-smirnov two sample test using STATISTICA (Version 12.0, Statsoft, Inc.) was conducted to identify any significant differences between weighted and not weighted costs. If not significant, only the weighted cost was used for Marxan analysis.

The results from the survey conducted by Cowley *et al.* (2013) found that 92.9% of fishing within the Sundays Estuary was recreational, which accounted for 89% of the total catch (boat sector = 46% and shore sector = 43%), and 76% of the total mass (boat sector =50% and shore sector = 26%). It was also found that the recreational sector, with higher effort and greater catch, had a higher non-compliance of size and bag limits restrictions than the subsistence sector (Cowley *et al*. 2013). Furthermore, recreational boating (with the highest CPUE of all angling activities, 0.30 fish angler^{-1}-hour^{-1}) poses a higher risk because boats are not limited to access and can reach areas of the estuary that shore angers cannot. The Sundays Estuary, not unlike other estuaries in South Africa, occasionally experiences overcrowding as far as recreational boats are concerned (Cowley *et al*. 2009, Lee 2014). Another issue associated with boat angling is the trolling of artificial lures, which is a widely used practiced technique by fishers that are targeting *A. japonicus* in the

Sundays Estuary. This technique is particularly effective because large areas can be covered, and when high use areas are located, fishers can cover that area repeatedly to ensure success (Dames 2014). Therefore, the cost of recreational boat angling was doubled in order to account for their high mobility and increased risk.

The objectives of the Small-scale Fisheries Policy (SSFP) (regulations promulgated in terms of the Small-scale Fisheries Policy 2012 [Government Gazette No. 35455]) are to ensure that subsistence fishers are not denied their right to fish and they are accommodated accordingly to address imbalances of the past. The results from the survey conducted by Cowley *et al*. (2013) found that only 7% of fishers were subsistence, and they had limited distribution which was most likely related to the distance they had to walk to reach fishing destinations and through private land areas where they could not afford the daily fee or were not allowed to walk. As a result, it was decided that the contribution of subsistence cost to the overall cost would be halved (Table 4.1) so that this sector would be displaced as little as possible.

Table 4.1: Different weighting given to the three fishery sectors (Recreational boat, recreational shore and subsistence sector) which form the Sundays estuarine fishery.

Fishing activity	Scenario 1 not weighted	Scenario 2 weighted
Recreational Boat	0.33	0.15
Recreational Shore	0.33	0.30
Subsistence	0.33	0.55

4.2.3.2 Temporal trends in fishing effort

The proportion of fishing effort for each sector was calculated from instantaneous counts in each PU for each month. The fishing effort was also weighted, and a Kolmogorov-smirnov two-sample test was conducted to check for significant differences in weighting fisher distribution for both seasons and if not significant, only weighted costs were chosen. Unfortunately, due to the design of the survey, day and night fishing effort was not collected, and a cost layer could therefore not be established in order to run a Marxan analysis.

4.2.4. Marxan testing

Marxan (version 2.43) (Ball & Possingham 2000) with the Zone Cogito Decision Support System (Segan *et al*. 2011) was used to evaluate which areas of the Sundays Estuary are most important for the strategic conservation of juvenile *A. japonicus*. Many user interfaces, like Zonae Cogito, have been developed to assist running the software. These have been found to be particularly helpful for generating appropriate input files and displaying Marxan outputs (Game & Grantham 2008). This approach was conducted in such a manner to evaluate whether high resolution acoustic telemetry data (for an important fishery species) and human distribution can be used effectively as input data for Marxan software.

4.2.4.1 Input files

The input files required for Marxan to run need to be in a specific technical format. All Marxan input files use ".dat" file extension. Marxan does not tolerate format mistakes and the generation of the files needs to be done in a thorough manner. The following four input files were generated in QGIS (version 1.8.0) which is an Open Source Geographic Information System (GIS). Qmarxan is a set of free software tools that allows for the creation of Marxan input files within QGIS.

Spec.dat – Conservation Feature file

- This file contains information about the conservation feature being considered, including its name (with a unique identifier or "id") and the target amount for the conservation feature. The "spf" column refers to the species penalty factor. For this analysis the conservation feature used was area use of juvenile *A. japonicus*, including temporal area use.

Puvspr.dat – Planning Unit versus Conservation Feature file

- This file contains information on the distribution of conservation features in each PU. Conservation features are assumed to occur only in PUs where an amount has been entered. For example, this would include the proportion of time spent by juvenile *A. japonicus* at each PU (receiver station).

Pu.dat – Planning unit

- This file contains information about the PUs themselves, such as PU id, cost, location and status (availability for selection).

Input.dat – Input Parameter file

- This file is used to set values for all the main parameters that control the way Marxan works, where to find the input data and where to place the output files. This file was created using the Qmarxan plugin tool 'configure scenario' in QGIS.

Bound.dat – Boundary length file

- This file contains information about the actual length of the shared boundaries between PU. This file is necessary if the Boundary Length Modifier (BLM) which improves the compactness of reserve solutions.

(Adron *et al.* 2008, Game & Grantham 2008).

When using decision support tools, like Marxan, sensitivity analysis and calibrations are used to better achieve biodiversity targets whilst minimising costs and threats (Levin *et al.* 2015). The last steps needed to run Marxan effectively were to determine the current protection status of PUs, considering the boundary length and the species penalty factor.

4.2.4.2 PU status

The status field, which is located in the pu.dat file, defines whether a PU is locked in or out of the initial reserve design. For example, a PU value of '0' means that the PU is not guaranteed to be in the initial reserve system. If a PU has a value of '1', it is included in the initial reserve system, but may not be included in the final solution. A PU value of '2 or 3' means that it is either 'locked in or out' of the initial reserve system and cannot be removed or added respectively (Game & Grantham 2008).

Whilst this variable is not necessary, it can help in instances where PUs are located in existing protected areas and could be assigned a status of '2' because it is unlikely that areas already being protected will be traded for others (Adron *et al*. 2008).
For example, given the limited access, yet importance, of subsistence fishers, consideration was given to 'lock in' the PUs that had high levels of subsistence fishers.

4.2.4.3 Boundary Length Modifier calibration

The Boundary Length Modifier (BLM) is a variable that controls the length of the boundary of the reserve system. It places emphasis on minimising the overall reserve system boundary length in order to produce a more compact reserve system which may be more desirable (Adron *et al*. 2008). When the BLM is set at zero, or a low value, the algorithm will drive the result in order to reduce the cost, in this case, fishing effort. When setting higher values of BLM, the algorithm will drive the results to minimise the boundary length, and therefore form a more compact system (Ball & Possingham 2000). The BLM can be thought of as a relative sliding scale, that ranges from cheap fragmented solutions (with a low BLM) to more compact expensive ones (high BLM) (Adron *et al*. 2008).

4.2.4.4 Species Penalty Factor

The Species Penalty Factor (SPF) is a multiplier that controls the level of penalty applied when the conservation target is not met. The higher the value, the greater the relative penalty and the more emphasis Marxan will place on ensuring that the features' target is met (Adron *et al*. 2008). If features of a high conservation value that are, for example, highly threatened or have significant social or economic importance, they will have a higher SPF value than less important features. Whilst a range of targets are being used for this analysis, there is a great need for them to be met because of the vulnerable state of estuary-dependent juvenile *A. japonicus.* In order to determine an appropriate SPF for the conservation feature, some experimentation was required.

4.2.4.5 Marxan output

Marxan was run with the simulated annealing algorithm and 100 repeat runs performed for each reserve design scenario. The number of repeat runs Marxan performs is effectively the number of solutions to the reserve problem that is generated. Whilst each run is independent from the previous one, the same parameters are used (Adron *et al.* 2008). Marxan outputs include a best solution which is a spatial output of the runs which achieved all the targets at the lowest cost, and the 'selection frequency', which is the number of times a PU is selected out of the number of runs in an analysis, which is a representation of the conservation importance of a PU in achieving defined targets (Game & Grantham 2008). The selection frequency is often regarded as giving a better indication of conservation importance of a PU in attaining targets because it is based on results of multiple runs rather than the single best solution (Grantham *et al*. 2011, Chalmers 2012).

PUs are selected above a certain threshold percentage of runs, and can be considered as high-priority areas, for example, 90% (Adron *et al*. 2008). After each run, Marxan generated summary data that included the best solution and summed solution (selection frequency), together with the number of planning units, the boundary reserve length and the score, which is calculated from the objective function equation (Figure 4.1).

It must be noted that Marxan is a decision support tool, which is designed to help guide the selection of efficient reserve systems, its output should not be regarded as the answer (Adron *et al*. 2008). Marxan will produce mathematically best solutions, however there is no single best solution to many conservation planning problems, and there is a likelihood of many good solutions depending on factors that are not necessarily incorporated in the analysis.

4.2.4.6 Planning scenarios

Four scenarios were used to investigate the influence of different conservation features (overall and seasonally), conservation targets (50%, 60%, 70% or 80%) and opportunity costs on reserve design (Table 4.2). Marxan takes into account costs data that are spatially variable in order to avoid areas of high costs where conservation features are present in alternative sites, in this way, the overall socio-economic impact of spatial restrictions are reduced (Chalmers 2012). This study aimed to identify potential no-take zones for fishing in the Sundays Estuary using count data as a proxy for the distribution of fishing effort. Non-consumptive recreational and tourism activities are unlikely to be affected through spatial zoning of consumptive use and were therefore not included in the development of the cost layers used.

Owing to the way the survey was designed, Cowley *et al.* (2013) did not gather information on the diel distribution pattern of fishers. Therefore, a cost layer cannot be developed for a Marxan analysis. However, there is data on the diel distribution of juvenile

A. japonicus at each PU. Whilst priority areas for conservation of juvenile *A. japonicus* could not be identified using Marxan software, because of a lack of cost data, the differences in day and night fish distribution were calculated.

4.2.4.7 Scenario 1 & 2: Overall area use (with different conservation targets)

Scenario 1 was used to identify a set of priority areas for the conservation of features with the calculated weighting costs applied to the three different fishery sectors (recreational boat, recreational shore and subsistence) (Table 4.1). Four different targets were used (Scenario 1a. = 50%, Scenario 1b. = 60%, Scenario 1c. = 70% and Scenario 1d. = 80%)

The conservation features (overall proportion of time spent at each PU for juvenile *A. japonicus*) was kept constant for Scenario 1. Scenario 1 was initially run with all parameters (BLM and SPF) set to 0. Following this preliminary analysis, a calibration SPF was done and the scenarios were run once more using the calibrated values for Scenario 2 (Figure 4.9). Because of the scale at which this analysis was run, as well as the linear distribution of PUs, BLM was kept at 0 for all Marxan runs.

4.2.4.8 Scenario 3 & 4: Seasonal area use (with different conservation features and costs)

Scenario 3 was used to assess whether there were seasonal (summer) differences in chosen priority areas for conservation of *A. japonicus*. The conservation feature used was the summer distribution (proportion of time spent at each PU) of juvenile *A. japonicus*. The cost used was the summer (September-February) distribution of fishers. This cost layer was calculated by totalling the proportion of fishers from each fishery sector within each PU for the summer months, and then weighted accordingly. SPF was set according to the calibration run in scenario 2, and BLM was kept at 0.

Scenario 4 was used to assess whether there were seasonal (winter) differences in chosen priority areas for conservation of *A. japonicus*. The conservation feature used was the winter distribution (proportion of time spent at each PU) of juvenile *A. japonicus*. The cost used was the winter (March-August) distribution of fishers. This cost layer was calculated by totalling the proportion of fishers from each fishery activity within each PU for the winter months, and depending on the results from Scenario 2, they were then weighted accordingly.

Scenario 3 and Scenario 4 were used to assess temporal changes in potential priority areas for conservation of juvenile *A. japonicus*. For this reason, the conservation target was kept the same for both scenarios, and was determined according to the best solution found in Scenario 2.

Table 4.2: Different scenarios analysed in Marxan, with varying input parameters; cost, feature, target, Boundary length Modifier (BLM) and Species Penalty Factor (SPF).

Scenario	Cost	Feature	Target	BLM	SPF
Scenario 1	Weighted cost	Overall proportion of time spent: juvenile *A. japonicus*	50,60,70 & 80%	0	0
Scenario 2	Weighted cost	Overall proportion of time spent: juvenile *A. japonicus*	50,60,70 & 80%	0	2.5
Scenario 3	Combined summer distribution of fishers	Summer distribution of juvenile *A. japonicus*	70%	0	2.5
Scenario 4	Combined winter distribution of fishers	Winter distribution of juvenile *A. japonicus*	70%	0	2.5

4.3 RESULTS

4.3.1 Conservation features

4.3.1.1 Area use by juvenile *A. japonicus*

The overall mean time spent at each receiver by all 56 juvenile *A. japonicus* was highest in the lower section of the middle reaches of the estuary (i.e. PUs 4-11), with a high proportion of time spent at receivers 6 and 10 (Figure 4.3). Mean time spent at receivers in the lower and upper reaches of the estuary were less than the mean time spent within the middle reach. The mean time spent below PU 4 and above PU 11 showed a general decrease in the proportion of time at those PUs (0-3). However, the mean time spent at PU 5 was less than time spent at adjacent PUs (i.e. receivers 4 and 6).

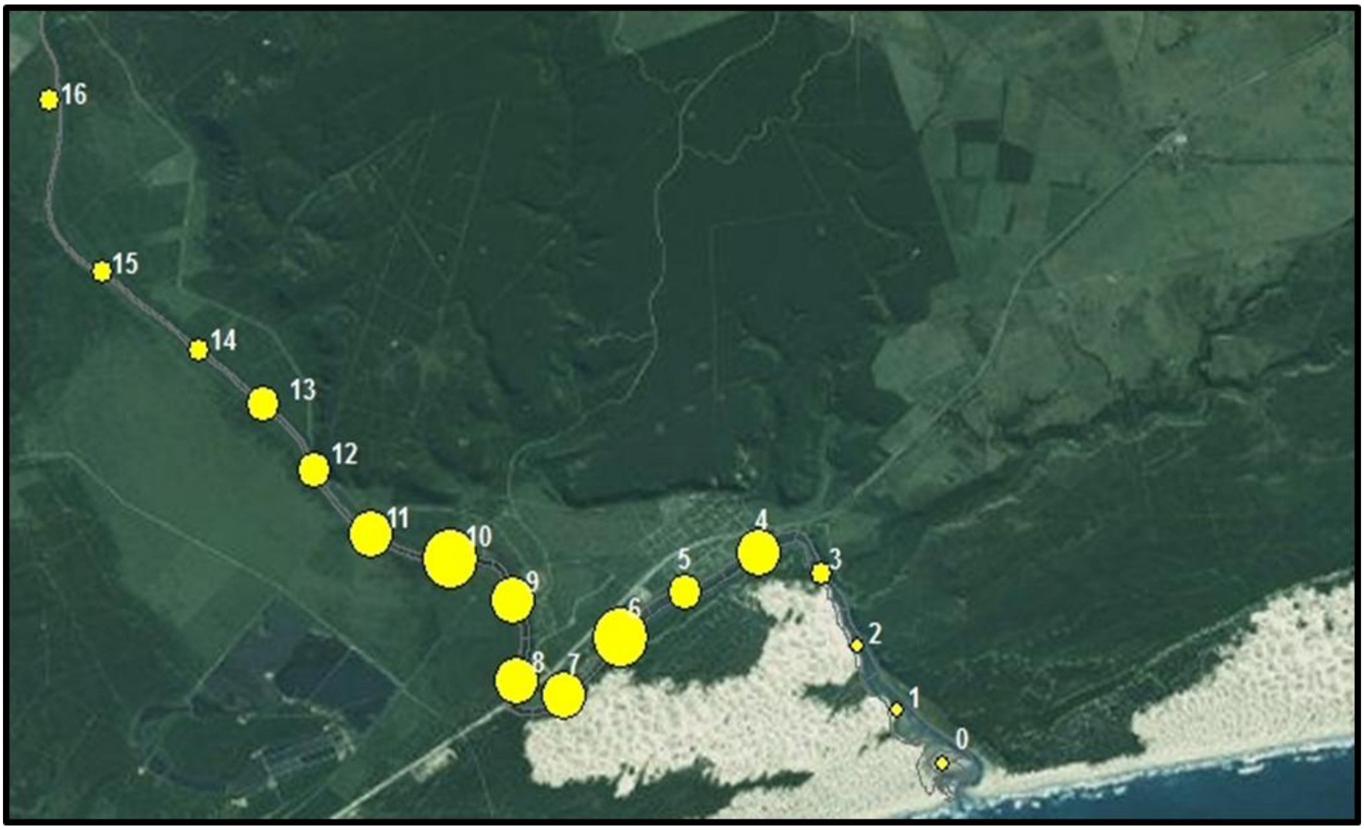

Figure 4.3: Map of the Sundays Estuary with a bubble plot representation of the mean proportion of time spent at each receiver (numbered 0-16) for all 56 acoustically tagged juvenile *A. japonicus*. PU 0 = Sea.

4.3.1.2 Seasonal area use

The distribution of time spent by juveniles *A. japonicus* within the 17 PUs (receiver stations) was significantly different between the two seasons; Summer (September-February) and Winter (March-August) ($F_{(16,\ 8789)}$= 4.67, p< .001) (Figure 4.4). Tagged *A. japonicus* spent a greater proportion of time in the upper section of the lower reaches and middle reaches (PUs 4-12) during winter compared to a greater proportion of time spent in the upper section of the middle reaches in summer (PUs 6-11), and the uppermost reaches in summer (PUs 15 and 16). Juvenile *A. japonicus* were more widely distributed throughout the estuary during the summer months. With the onset of winter the distribution of area use was concentrated in the middle and the lower sections of the upper reaches of the estuary, between receiver 4 and 11.

Figure 4.4: Map of the Sundays Estuary with a bar graph representation of the proportion of time spent at each receiver by all 56 juvenile *A. japonicus* over the summer (September-February; yellow bars) and winter (March-August; red bars) seasons.

4.3.1.3 Diel area use

Based on the proportion of time tagged juvenile *A. japonicus* spent at each PU, there was no significant diel difference ($F_{(16, 8789)} = 0.52$, $p = 0.94$) (Figure 4.5). Juvenile *A. japonicus* did spend more time in the lower reaches and at sea (PU 0) at night compared to the day, however, this was not significant ($p > 0.1$).

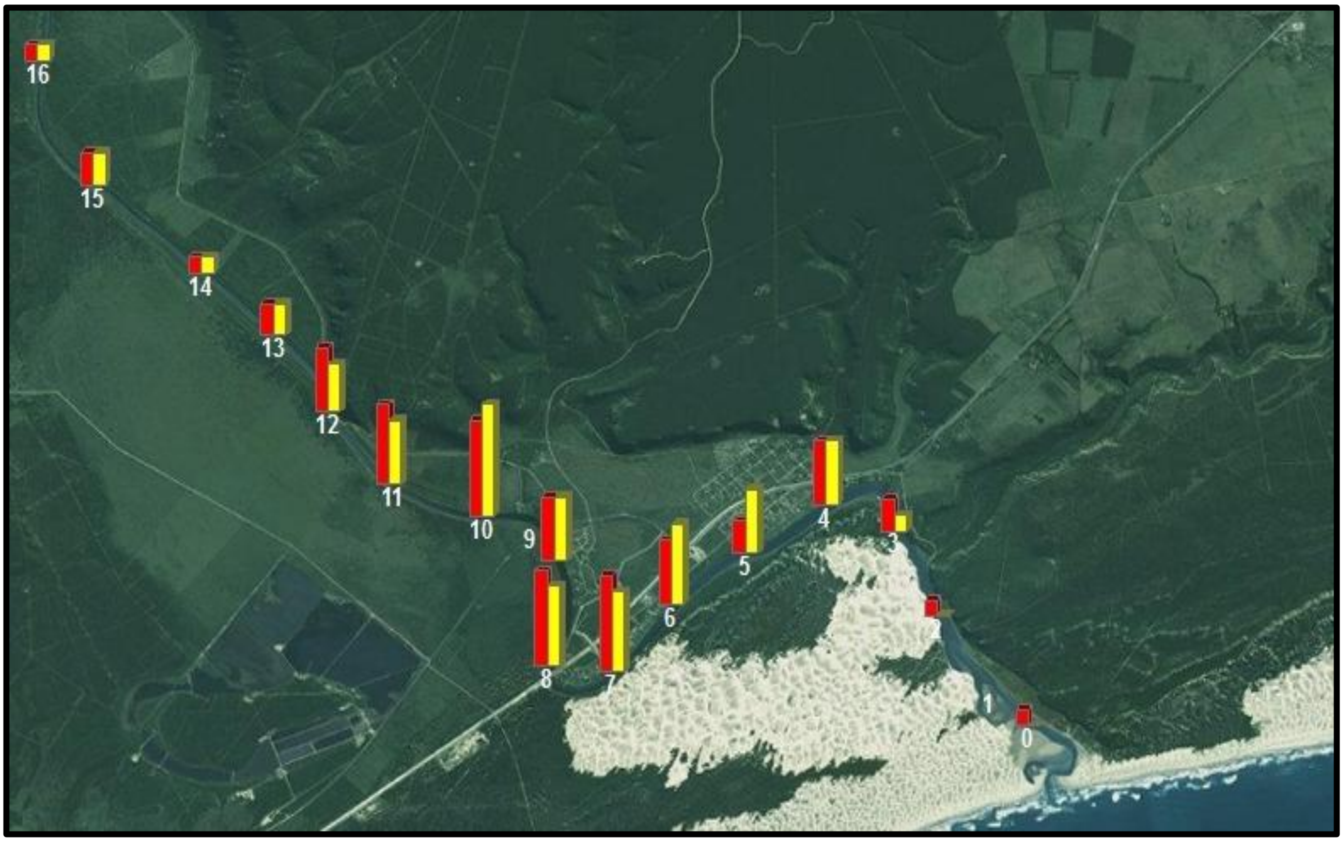

Figure 4.5: Map of the Sundays Estuary with a bar graph representation of the proportion of time spent at each receiver by all 56 juvenile *A. japonicus* during the day (yellow bars) and night (red bars).

4.3.2 Costs: Distribution of human activities

4.3.2.1 Overall fishing effort

The results from the instantaneous counts survey conducted by Cowley *et al*. (2013) revealed that recreational boat angling was distributed throughout the estuary (Figure 4.6), but mostly focussed in the lower reaches of the estuary, between PU 1 and 2. Recreational shore angling showed more of a patchy distribution (Figure 4.6), with a large amount of the effort at PU 4 and in areas adjacent to private homes (PUs 4, 5, 11 and 12). There was also a considerable amount of recreational shore fishers in the vicinity of PU 3 and 4, within the privately owned area. Subsistence angling was concentrated around developed areas (PU 3 to 6) easily accessible on foot. There was very little subsistence fishing above PU 7 and in the lower reaches of the estuary, possibly due to the daily fee charged to enter the Pearson Park facility (Cowley *et al*. 2013).

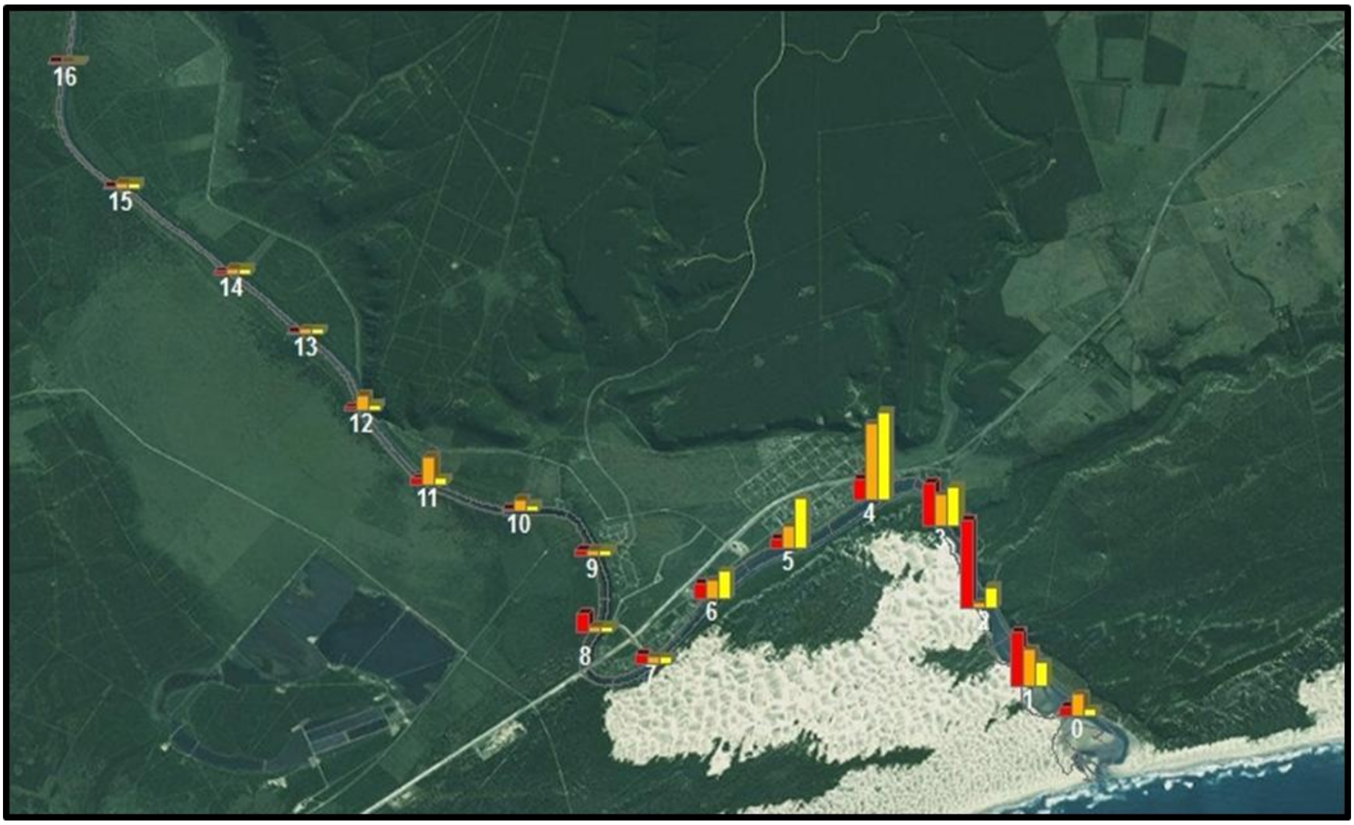

Figure 4.6: Map of the Sundays Estuary with a bar graph representation of the spatial distribution of recreational boat angling (red bars), recreational shore angling (orange bars) and subsistence angling (yellow bars).

4.3.2.2 Weighting of different fishery sectors

There was little difference between the overall distribution of fishing effort and the weighted distribution (Figure 4.7). However, as the weighting for subsistence fishing effort was doubled in order for their cost to be increased (to decrease the likelihood of this sector being displaced), there was a greater distribution of weighted effort in PUs 4, 5 and 6. There was also a change in the distribution of weighted fishing effort in the lower reaches, which is an area of high recreational boat angling activities, whose weighting was halved because of their higher risk (Table 4.1). There was no significant difference ($p>0.1$) found between weighted and not weighted costs, and as a result, the weighted fisher distribution cost layer was used for Marxan analysis.

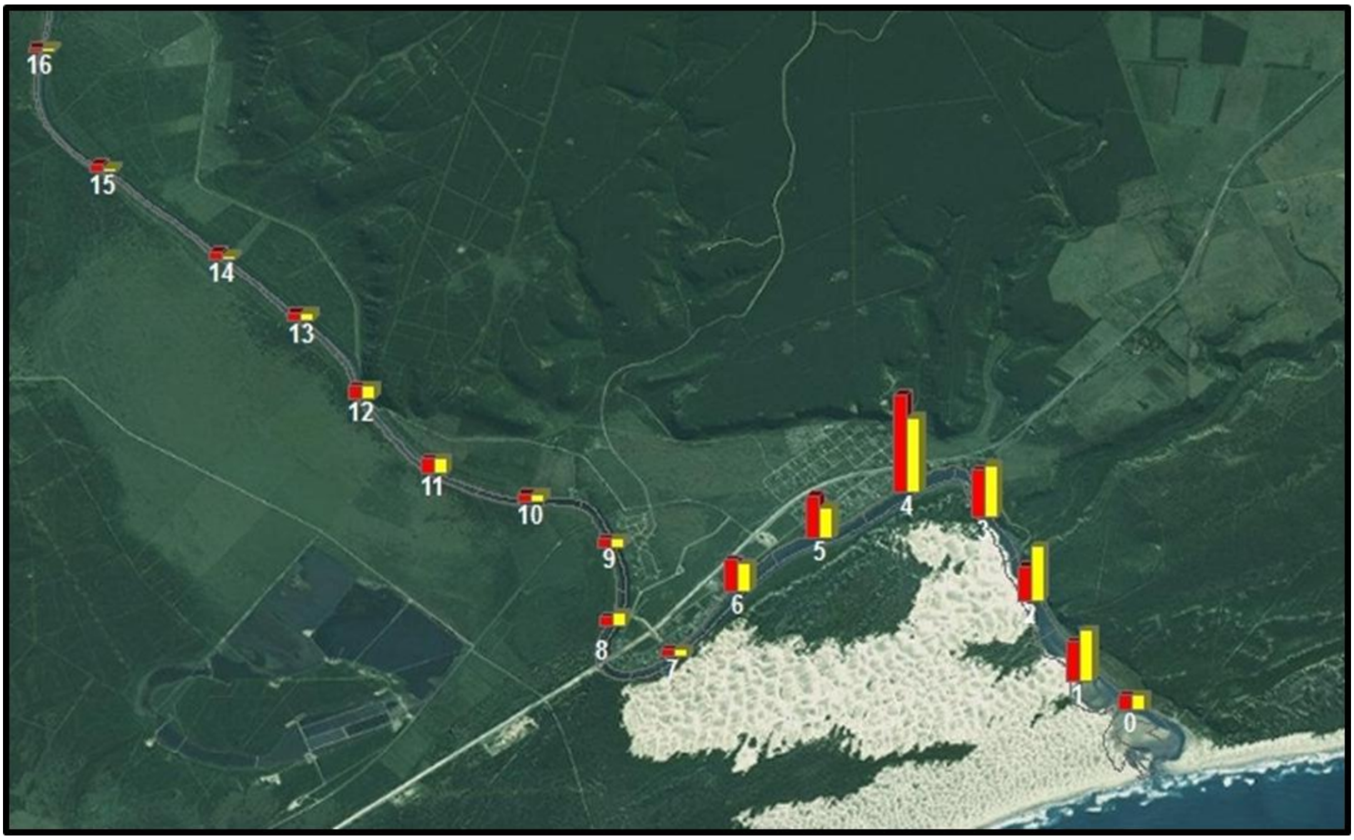

Figure 4.7: Map of the Sundays Estuary with a bar graph representation of the spatial distribution of combined weighted (red bars) and not weighted (yellow bars) fishing effort.

4.3.2.3 Seasonal fishing effort

No significant differences were found between weighted and not weighted fisher distribution for both seasons (K-S test: $p > 0.1$), hence a weighted cost layer was used for the Marxan analysis. Temporal trends in fishing effort in all PUs were relatively constant, with only slight differences between seasons (Figure 4.8).

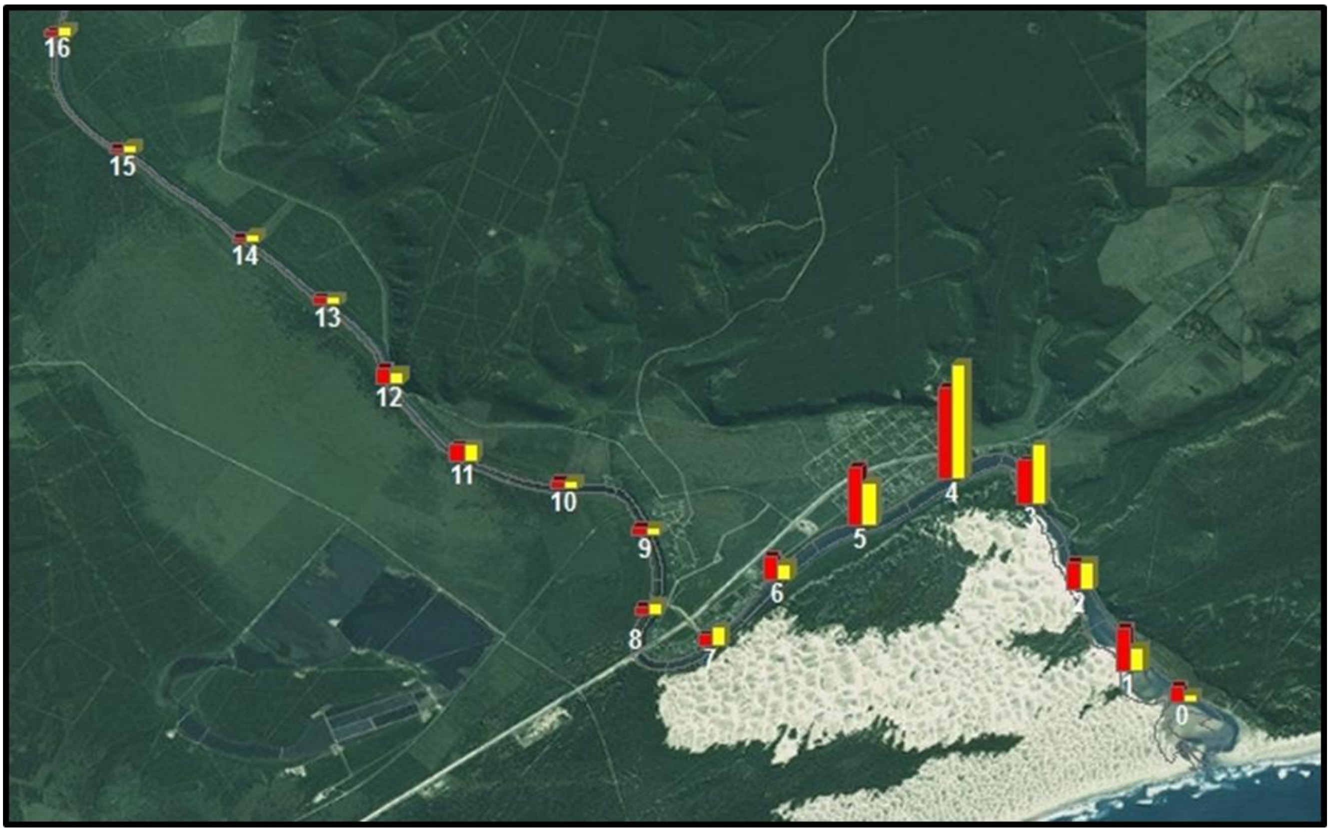

Figure 4.8: Map of the Sundays Estuary with a bar graph representation of the seasonal spatial distribution of combined weighted summer (red bars) and winter (yellow bars) fishing effort.

4.3.3 Marxan results

4.3.3.1 Scenario 1 & 2

Scenario 1 aimed to identify priority areas using weighted costs using a variety of targets (50, 60, 70 and 80%). Parameters such as BLM, SPF and the status of PU were kept at 0 for this scenario to test the importance of individual PU. Marxan failed to find near-optimal solutions for all three target runs. Because there was no penalty to meet specified targets, Marxan was unable to find a solution at a minimal cost. These preliminary results are unfeasible because whilst the objective function of Marxan is to find optimal areas to protect at a minimal cost, a solution where there is no penalty for not meeting targets does not result in a protected area would not be beneficial. In these situations, parameters need to be adjusted accordingly (Fischer & Church 2005).

In order for a Marxan analysis to be considered robust, sensitivity testing, in the form of adjusting parameters needs to be included to ensure good practice. Following the preliminary analysis (Scenario 1), the SPF was calibrated to set a higher penalty for not achieving targets (Figure 4.9). The point on the graph where the missing values (in this case, just one missing value) are approaching zero is the most efficient SPF, which for this analysis was 2.5. The missing values refer to the missing conservation features from a Marxan analysis.

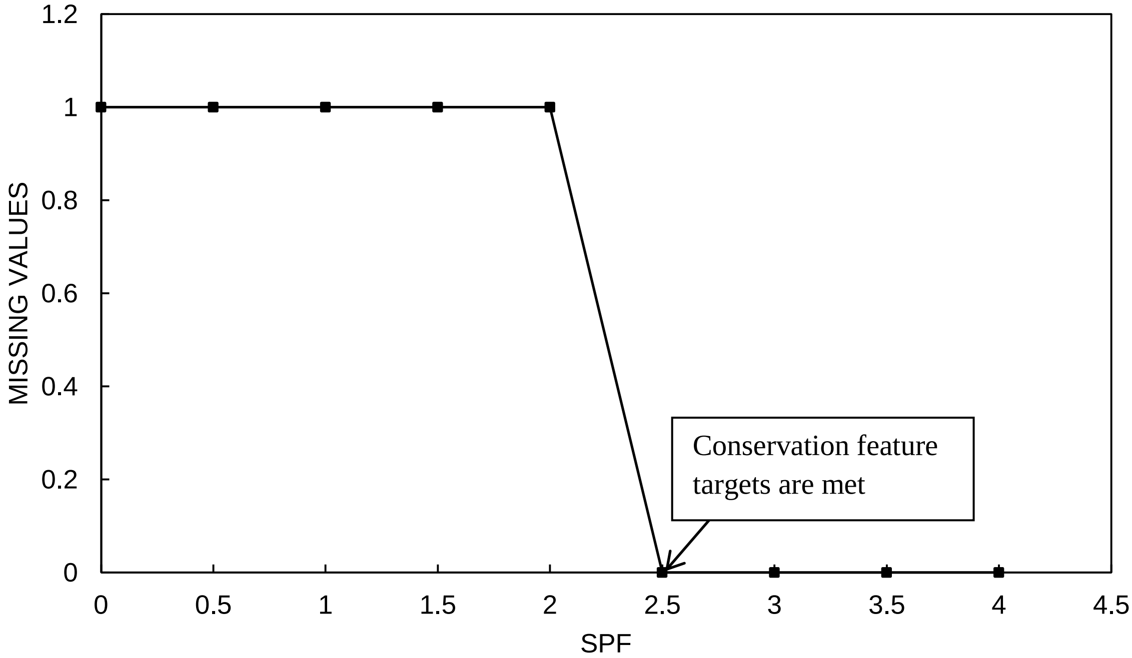

Figure 4.9: Species Penalty Factor calibration results run in Zonae Cogito, showing an SPF of approximately 2.5, the conservation feature is no longer missing.

Marxan was then re-run using a SPF of 2.5, for four different targets (50%, 60%, 70% and 80%). SPF had a considerable effect on the areas selected for protection, with more PUs chosen as the target increased (Figure 4.10). Marxan runs 100 different scenarios and PUs are chosen a specific amount of times within those 100 runs according to their irreplaceability or importance. However, one of the limitations of running Marxan at such a small scale, and with only one conservation feature resulted in PUs either being chosen zero or one hundred times, and there is no variability in solutions because the problem is too simplistic for Marxan to compute and there are not enough PUs to choose from to create a variety of solutions.

The incorporation of the SPF parameter into the analysis resulted in PUs being selected for potential reserves that conserved areas with highest proportions of fish time spent, whilst minimising the cost of displacing fishing effort (by the subsistence sector). The results, shown in Figure 4.10, are plotted for different targets whereby each of the four scenarios (50, 60, 70 and 80%) gave just one solution. The results show that as the target increases, so does the cost of the protected area, as a result of more PUs chosen to meet the higher target (Figure 4.10). Although none of the four scenarios reached their individual conservation target, 60, 70 and 80% targets would still, barring a target of 50% (45% conservation achieved), conserve more than 50% of time spent by juvenile *A. japonicus* in the Sundays Estuary. The objective function score also increased with increasing costs and more PUs, which was expected because it summed the costs, boundary length and SPF.

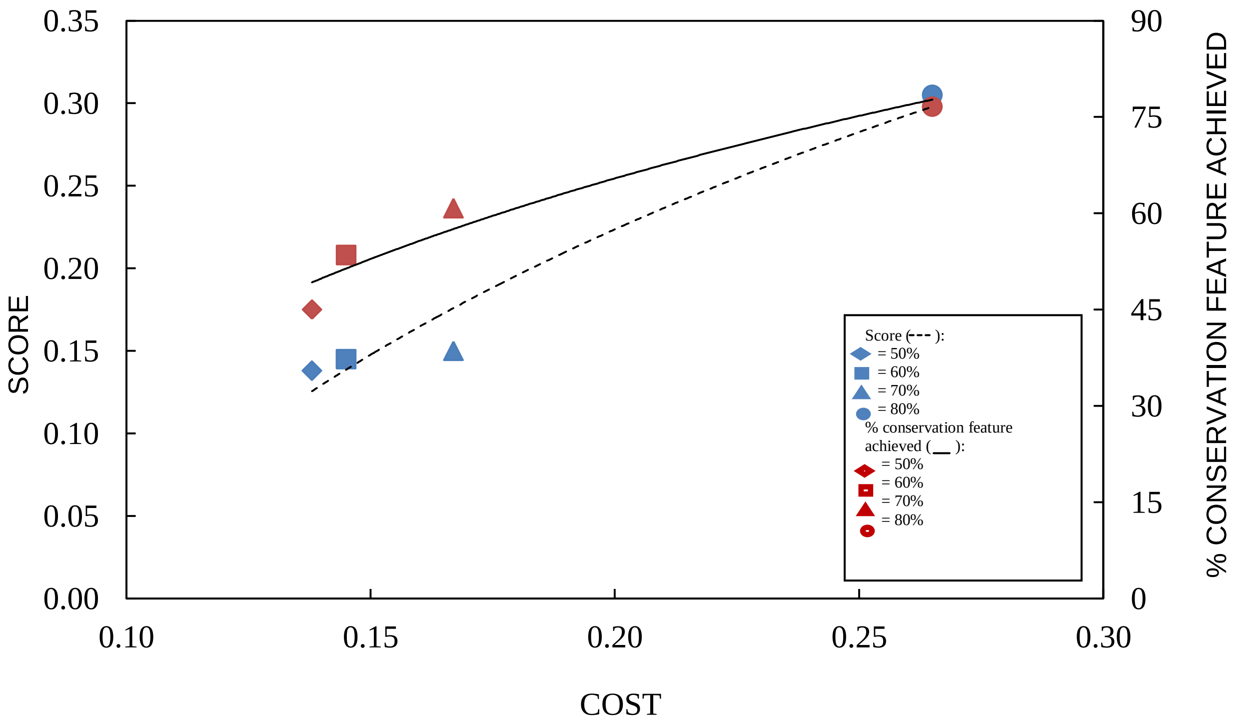

Figure 4.10: The relationship between cost and the objective function score (calculated from Figure 4.1) and percentage of the conservation feature that would be protected for four different scenarios (2a, 2b, 2c and 2d).

The best solution output from the four Scenarios using different conservation targets is depicted in Figure 4.11. As the conservation feature target increased, more PUs were included in the solution in order to meet the higher targets. More fragmented reserve solutions were chosen in Scenario 2a and 2b compared to 2c and 2d, but differences in compactness were relatively small. Whilst there was only a modest increase in the cost for the first three scenarios (2a = 0.14, 2b = 0.15 and 2c = 0.17), scenario 2d had a considerably higher cost (0.27) with its inclusion of PU 6, which had a higher fishing effort than the upper reaches of the estuary (Figure 4.6 and 4.7). Scenario 2d's increased protection of fish time spent can also be ascribed to the proportion of time spent by *A. japonicus* in PU 6 (Figure 4.3).

The Marxan output results suggests that the most appropriate scenario should include a protected area from PU 7 to 15 (Scenario 2c), with 0.61 of the proportion of time spent by juvenile *A. japonicus* being protected, and only 0.17 of the weighted fishing effort being displaced. Scenario 2c also had the least fragmented reserve system and excluded the high proportion of subsistence fishers in the middle and lower reaches of the estuary (PUs 0-6). This scenario would affect recreational shore fishers the most, with a displacement proportion of 0.3, followed by 0.2 for recreational boat fishers, and only a proportion of 0.16 for subsistence fishers (Figure 4.6). If PU 6 was to be included (i.e. Scenario 2d), the proportion of subsistence fishers that would be displaced increased to 0.25. Importantly, Scenario 2c also had the least fragmentation compared to the other scenarios, and the change in cost of displaced fishers is only slightly higher than the smaller area closures (Scenarios 2a and 2b). Based on these results, seasonal scenarios used a conservation target of 70%.

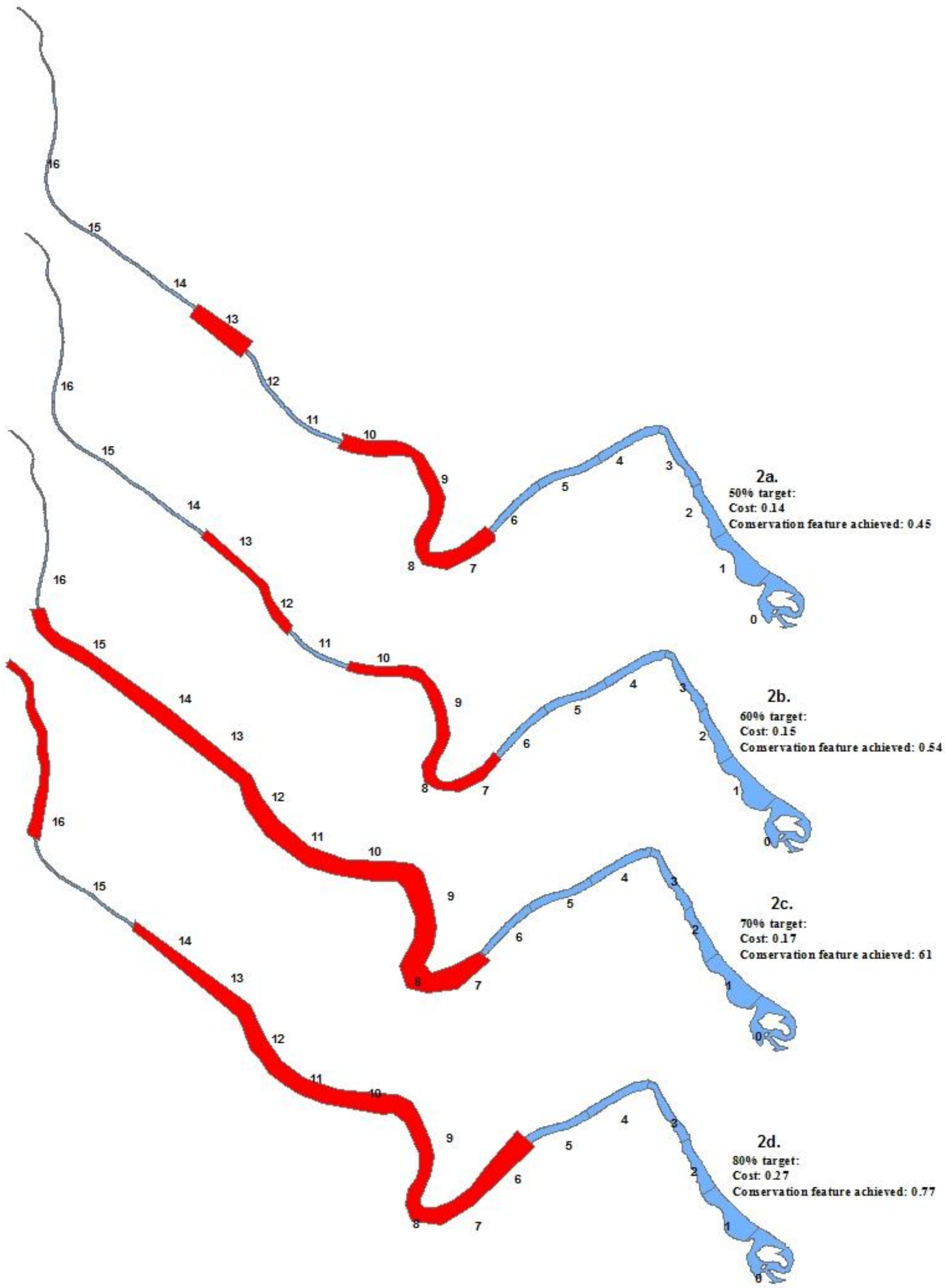

Figure 4.11: Marxan best solution outputs for differing conservation feature targets. The areas in red indicate the selected reserves. The changes in conservation feature targets result in a trade-off between conservation feature and cost. In a Marxan analysis, more compact reserves are often more expensive, but result in higher conservation features being met, and more logistical reserve planning.

4.3.3.2 Scenario 3 and 4

Despite a significant difference in the seasonal area use by juvenile *A. japonicus*, the Marxan outputs for Scenario 3 (summer) and Scenario 4 (winter) were similar. The best solution for the summer scenario included 9 PUs (7-11, and 13-16, Figure 4.12), protecting juvenile *A. japonicus* for 0.71 of the time, at a cost of 0.19 to fishers. The inclusion of the upper PUs (15 and 16) were ascribed to the increased use of the upper reaches during spring (Figure 4.5). For winter, the best solution included 7 PUs (6-12) which would protect juvenile *A. japonicus* for 0.69 of the time at a cost of 0.23 to fishers.

There is a common seasonal trend in proportion of time spent of *A. japonicus* between PU 6-12, with particularly high use in PU 8-10 for both seasons. The proportion of time spent by fish during winter was aggregated around the middle reaches (6 to 12) which explains the choice of the closed area in that section. The high use of the upper reaches in summer warrants that sections inclusion during that season. However, whilst differences in the time spent of juvenile *A. japonicus* exist for different seasons, there would only be a difference of 0.2 between summer (0.59) and winter (0.61) if a closed area was introduced for PUs 7-13 (Scenario 2). Furthermore, there would also only be a difference of 0.2 for seasonal fisher distribution (summer = 0.19, winter = 0.21) if the same closed area was introduced (Figure 4.8). The consideration of seasonal scenarios would therefore result in only very small differences.

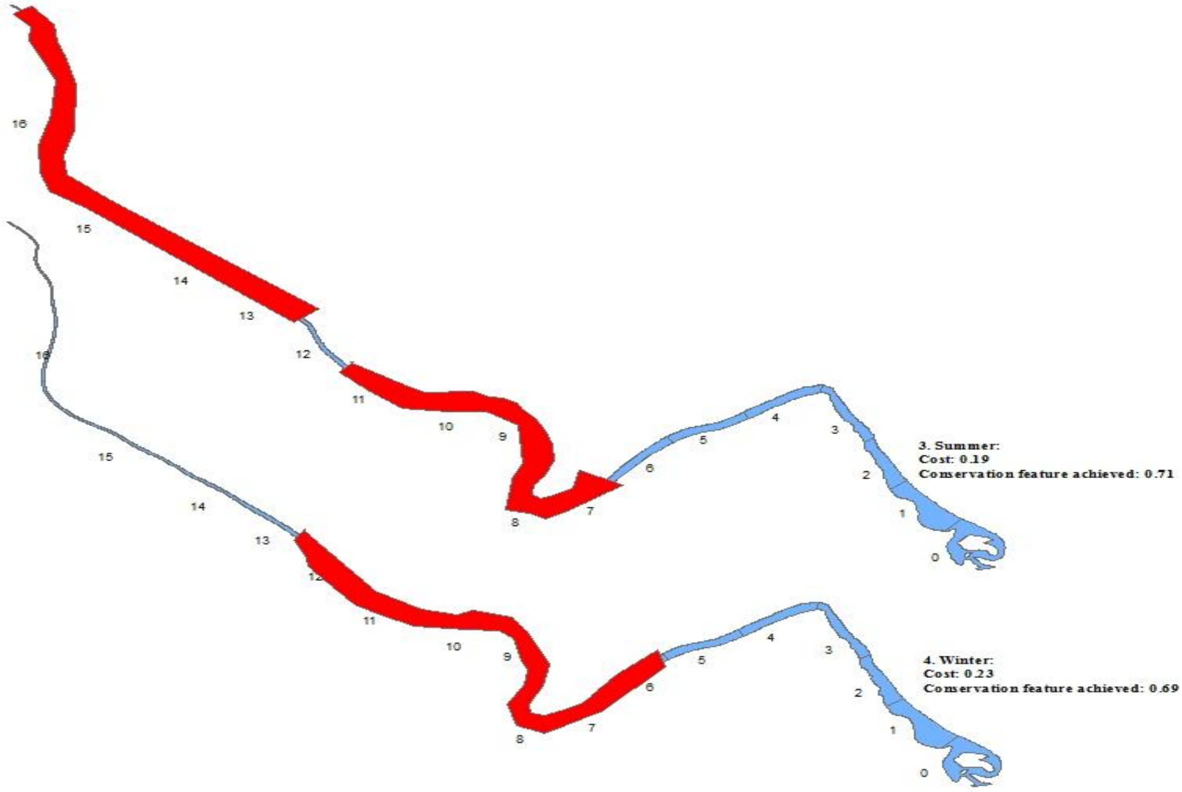

Figure 4.12: Marxan best solution outputs for seasonal scenarios. The conservation target for both scenarios was set at 70% based on the results from the second scenario, and an SPF of 2.5. The proportion of time spent for each season resulted in chosen PUs, however the bias associated with the simplistic Marxan analysis may have skewed the results for both season.

4.4 DISCUSSION

The conservation planning software programme Marxan was designed to aid in the selection of new conservation areas at a minimal cost, by exploring tradeoffs between meeting conservation and socio-economic objectives (Adron *et al*. 2010). Marxan was used by Chalmers (2012) to identify priority areas for fisheries management and conservation of vulnerable linefish in Algoa Bay, however, the Sundays Estuary which enters this coastal embayment was not consider in the planning process. This omission has potential consequences for the local coastal fishery as the Sundays Estuary serves as an important nursery area for several important, yet over-exploited fishery species, including *A. japonicus*. Therefore, this chapter aimed to assess the feasibility of Marxan as a conservation planning tool to identify areas of high use by juvenile *A. japonicus* in the Sundays Estuary at a minimal cost to fisheries. This was done using high resolution acoustic telemetry data and detailed information of fisher distribution

Based on the best solution of selected PUs, provided by Marxan in Scenario 2, key sites for the conservation of juvenile *A. japonicus* in the Sundays Estuary were identified. In particular, Scenario 2c with the inclusion of PUs 7-15 would be the most appropriate for a no-take estuarine protected area. According to Adron *et al*. (2010) compact reserve areas with low management costs may provide multiple benefits. Scenario 2c identified a single continuous stretch of the estuary, which if closed off to fishing would protect an area where juvenile *A. japonicus* spent a high percentage of time (61%). The identified area between PUs 7-15 would only displace fishing effort by 17%, of which most is the recreational sector who could afford the entrance fee charged by private land owners (e.g. Pearson Park) or have the means to move to the area above PU 16. Whilst this study focused on identifying high use areas for the protection of juvenile (<600 mm TL, legal size) *A. japonicus*, future analysis could include different size classes as individual conservation features.

Importantly, Scenario 2c excluded PU 6, which was identified as an area frequently used by subsistence fishers. Despite being the smallest fishery sector on the Sundays Estuary, the exclusion of PU 6 comes at a minimal cost to these fishers, with the additional benefits of avoiding potential conflicts of displacing them and in line with the recently Gazetted small-scale fishery policy (SSFP) of South Africa.

Since information on the spatial and temporal movements of estuary-dependent fish is essential to effective estuarine reserve design (Tremain *et al.* 2004). The timing of fish movement and area use in relation to seasonal fishing activity is an important factor to consider (Kerwath *et al.* 2009). Although *A. japonicus* were more active in the summer months, exhibiting a wider distribution of area use, with a significant difference between summer and winter, PUs 6-10 were important high use areas during both seasons. In terms of cost, there was a peak in fishing effort in the summer months between PU 2-4, however in the vicinity of the proposed close area (Scenario 2c), fishing effort was relatively similar.

Wood *et al.* (2004) provided a set of generic guidelines for the sustainable use of Eastern Cape estuaries which can be adapted or adjusted for particular systems. One of the guidelines was to restrict fishing to daylight hours. Recent legislation witnessed a ban on night fishing and the utilisation of trolling techniques in the Breede Estuary in the Western Cape Province (regulations promulgated in terms the Department of Environmental affairs and Tourism Act No. 18 of 1998 [Government Gazette No. 37047]), The Breede River Conservancy, with on-going catch monitoring data, suggested that a night fishing ban on the Breede Estuary would halve the total catch of *A. japonicus* (Lamberth 2007). There is currently no CPUE data to compare day and night fishing on the Sundays Estuary. The survey conducted by Cowley *et al.* (2013) found that 37% of respondents said they fish at night, and only 7% responded positively when asked if they would support a night fishing ban on the estuary. Without accurate knowledge of day and night effort and distribution of fishers, a temporal closed area scenario was not feasible.

Although the findings from this chapter do not adhere to the CARE principles in a traditional sense, there is still merit using this analysis at the scale at which it was done and it can be argued that the CARE principles have been adapted to this case study. For example, although this analysis used only one species within an area with a low number of planning units which would not necessarily address *Comprehensiveness* and connectivity with other estuarine systems. If connectivity is addressed at an estuarine level, *Comprehensiveness* within the estuary was achieved. There is also currently no direct information on the population size of *A. japonicus*, and so it is difficult to know how *Adequate* the identified closed area would be in terms of species persistence. However, given that there is low connectivity of *A. japonicus* among adjacent estuaries, and there is a high level of residency to the Sundays Estuary, it can be argued that the Sundays Estuary supports its own local fishery (Childs 2012). Additionally, the data used (time spent) was at such a high resolution, it provided empirical evidence that juvenile *A. japonicus* are extremely dependent on estuarine nursery areas. Recapture statistics from a study conducted in Sundays Estuary (Childs *et al.* 2015) showed that 33% of the tagged fish were recaptured in the fishery. In terms of species persistence, it would be critical to provide some form of protection to juvenile *A. japonicus* in the Sundays Estuary to maintain population persistence.

In terms of *Representativeness* it is questionable whether the adoption of a single-species approach compromised the ecosystem-based management philosophy of Marxan (Adron *et al*. 2010). However, it must be borne in mind that the aim of this research was to evaluate alternative means (i.e. using conservation planning software) to address the dire need for improved management of a heavily targeted and over-exploited species which is dependent on estuarine habitats as nursery areas. The analysis did achieve overall *Effectiveness* by choosing a reserve that affords the most protection of the conservation feature at a minimal cost.

Whilst several conservation solutions were obtained, it is possible that the results are misleading because the problem was too simplistic. There were not enough conservation features and planning units, and as a result, Marxan only chose one solution each time for 100 runs, giving no variation in selection frequency. For this reason, the best solution was used. With a small number of PUs, Marxan ultimately selected PUs associated with the lowest cost, which in this case were PUs 13-16.

Another limitation of this study was the need for an interpolation of the fisher distribution data because the sampling-design of the fishery survey did not cover the entire study area (0-16 PUs). The adopted Kriging approach to the limited dataset provided results that might have been driven by distance decay and ultimately introduced considerable bias. The Kriging approach that was used was based on data with a trend and the extrapolated data was calculated to best avoid bias. Marxan users have often omitted uncertain cost data during the prioritisation process, however it is argued that whilst planning scenarios with inaccurate data will not deliver the most desired conservation efficiency, uncertain cost data is more efficient than ignoring cost altogether (Ferraro 2003, Kremen *et al*. 2008, Carwardine *et al.* 2010). Consequently it is recommended that future studies conduct sampling along the entire length of the estuary to run spatial planning scenarios with more accurate cost data.

In terms of the conservation feature, juvenile *A. japonicus* (<600 mm TL) was chosen to address the issue of non-compliance of the current bag and size limit regulations. Childs *et al.* (2015) suggested that juvenile *A. japonicus* exist as a meta-population, with distinct estuarine and marine contingents. Their study showed that estuarine-dependent *A. japonicus* were more vulnerable to over-exploitation, and the continued removal of individuals has consequences at a meta-population level (Childs *et al*. 2015). If larger size classes' of *A. japonicus* were to be added to this type of planning scenario, they would need to be considered as a different conservation feature because of their significant differences in area use in the estuary compared to juveniles. Dames (2014) showed that larger *A. japonicus* (600-1000 mm TL) utilised the lower reaches of the estuary with limited utilisation of the upper parts of the estuary. Complimentary to this study, Chalmers (2012) proposed a 2km buffer area around the Sunday Estuary mouth in the local coastal fisheries conservation plan, and indicated that the inclusion of the Sundays Estuary mouth would increase regional no-take targets by 11% and make a significant contribution to the management of inshore estuarine-dependent linefish stocks in Algoa Bay. Consequently, it is recommended that future research efforts should include larger size classes of *A. japonicus* as separate conservation features, and assess the potential conservation benefits, for this species, and other vulnerable estuarine linefish species, if the mouth of the Sundays Estuary was included in no-take zones. Increasing the amount of conservation features to this analysis could add to the variability of solutions and Marxan could potentially better respond to a more complex problem.

Based on the results of this assessment it can be argued that the analysis was too simplistic for the Marxan heuristic algorithm. Integrating acoustic telemetry data and conservation planning tools is a relatively new field, and whilst the software has not yet been developed, there is much possibility for this type of work in the future (Dr R. Dwyer, University of Queensland, 2015 *pers comm*). In recent years, many tools have been developed to efficiently manage data collected using acoustic telemetry techniques, including the data management-based system called V-track (Campbell *et al.* 2012), which provides users with the flexibility to manage and plot their data. A new tool that uses passive acoustic tracking data managed through V-track software within a conservation decision framework has been developed to inform the optimal siting of fisheries closures (Dr R. Dwyer, University of Queensland, 2015 *pers comm*). To date, this tool has only been used once to find a management solution for the protection of spear tooth sharks (*Glyphis glyphis*) in a tropical river in Northern Australia. The method used fisheries cost layers as well as the above-mentioned acoustic telemetry data, and was funnelled through Marxan analysis. This software has not yet been released, but will help users optimise their own management actions and promote species conservation in the future (Dr R. Dwyer, University of Queensland, 2015 *pers comm*). There are few studies that incorporate acoustic telemetry data into spatial planning analysis, however there is great potential in the integration of the two (Campbell *et al*. 2012).

Although the use of Marxan provided spatial-based management options for the protection of juvenile *A. japonicus* in the Sundays Estuary, it is possible that the exercise conducted in this chapter could have been performed using simplistic approach in Microsoft Excel. For example, a potential reserve solution can be assessed by including each of the 17 planning units that are either in or out of the final solution (i.e. 2^{17} = all possible 131072 configurations) possibly making it more powerful than the 100 runs performed in Marxan in this assessment (J. McGowan, University of Queensland, 2015 *pers comm*).

Furthermore, decision science in the form of cost-benefit analysis also provides a means to look at different scenarios for closed areas. This process could involve weighing the total expected costs (in this case fisher displacement) against the total expected benefits (in this case proportion of fish time protected) in order to choose the best or most appropriate option. One can then produce trade-off curves where competing objectives (trade-offs) or complementary objectives (win-win) can be defined on the trade-off curve (Stewart & Possingham 2005). A basic trade-off curve was plotted (Figure 4.10) to reveal the potential application of this simplistic approach.

Nonetheless, in light of the growing demands placed on global fisheries resources, continued development of new methods and tools to analyse and plan spatial-based management options is expected and ultimately essential to address the challenges of effective management.

CHAPTER 5: DISCUSSION AND RECOMMENDATIONS

5.1 INTRODUCTION

Due to the global issue of overfishing, and the failures of traditional management regulations, alternative strategies such as spatially explicit management, based on ecosystem-based principles have been widely advocated (Sale *et al.* 2006). The search for improved management frameworks has led to a shift to a more holistic approach to fisheries management (Smith & King 2010) (see Chapter 3). Ecosystem-based management (EBM) is now a widely known paradigm underlying marine resource management worldwide (Möllmann *et al.* 2013). The development of a holistic and transparent ecosystem approach for fisheries management has received considerable attention and EBM has focused management on diverse human activities in a geographically defined area, particularly those such as fisheries that have a direct impact on resources (Pikitch *et al*. 2004). Furthermore, the designation and protection of essential fish habitats such as estuaries, which are important nursery areas, are becoming primary considerations in fisheries management (Beck *et al.* 2001, Childs 2013).

The development of an integrated ecosystem-based management approach requires extensive information on the social and institutional forces affecting the resource users and the status of the fishery itself (Castro 2001). Globally, there is a lack of scientific information on estuarine fisheries, and often a lack of technical expertise to assess their current state (Zann 1999). In most South African estuaries, there is a lack of data on the level of resource utilisation and socio-economic information of the resource users (Pradervand & Baird 2002, Cowley *et al*. 2004, Nsubuga 2004, Cowley *et al*. 2013). A comprehensive understanding of the status of the resource is required before changes in management regimes can be implemented (Cowley *et al.* 2013). Simple, but scientifically robust techniques have been developed for assessing resources and/ or activities, known as rapid appraisal techniques (Pido *et al.* 1997, Smith & King 2010). These rapid sustainability assessments prioritise areas for conservation and management and evaluate the sustainability status of fisheries according to EBM principles (Pitcher & Preikshot 2001).

Spatial trends in resource use and of the resource can aid in developing spatial management plans and is a central component of EBM. Marine spatial planning is one of the key tools which can be used to facilitate the implementation of EBM, and incorporates a full range of anthropogenic drivers on the marine environment (Halpern 2008).

The sustainability assessment conducted in Chapter 3 provided empirical evidence that the Sundays estuarine fishery is currently unsustainable. The estuary is an important recreational fishing destination for local and nearby residing people and its resources are heavily exploited. The sustainability of this fishery hinges on improved law enforcement and compliance to fishery regulations, particularly highlighting the capture and retention of undersized fish of many vulnerable species (Cowley *et al*. 2013). Current institutional inadequacies need to be prioritised and more appropriate management interventions need to be proposed.

In Chapter 4, the CARE principles associated with the design of Marine Protected Areas were discussed within the scope of the Sundays Estuary spatial management plan. The individual issues associated with this spatial-based approach were discussed, in particular, the challenges of this approach for a single estuary. However, when looking at the broader goal of the inclusion of the Sundays Estuary in the proposed Addo Elephant National Park MPA, those CARE principles can be effectively applied. *Comprehensiveness* requires reserves systems to sample a full range of biodiversity (including different life stages) at an appropriate scale within and across each bioregion. In terms of the proposed MPA, the inclusion of the estuary in the protected area will not only protect juvenile *A.japonicus* and other vulnerable species from growth overfishing, but also benefit the coastal inshore fishery in Algoa Bay achieving *Adequacy* in the reserve design. In Australia, estuarine systems are not separated from marine reserves and are included in the general category, Marine and Estuarine Protected Area (MEPA) (Rigney 1990, Whitfield & Cowley 2010). This type of protected area could be established through the zoning of certain activities, such as fishing. Furthermore, species such as *A. japonicus* exist as a meta-population with several subpopulations, each consisting of both marine and estuarine juvenile contingents (Childs *et al*. 2015); therefore protection of both contingents is required for effective conservation of the species.

The inclusion of the Sundays Estuary in the MPA would promote biodiversity persistence through the spatial relationship amongst habitats and offset the effects of overfishing to one area. It would also form a *Representative* system, in which a range of species could be protected (Anon *et al*. 2008). To make the problem more complex, more species could be added to the conservation plan, which would result in more variability in planning units (PUs), and a larger representation of the area's biodiversity. The results from Chapter 4 identified the highest conservation feature at a minimal cost therefore achieving *Effectiveness* of the system. Although this study identified the need to use multiple features to effectively use the tool; Marxan (see Chapter 4), it has highlighted how appropriate a spatial-based management plan will be for the Sundays estuarine fishery and identified the importance of the estuary in the expansion of the Addo Elephant National Park MPA.

5.2 A PRECAUTIONARY APPROACH TO ESTUARINE FISHERIES MANAGEMENT

A precautionary approach to fisheries management has been advocated, globally and locally (see Chapter 3), which takes into account the uncertainties in fisheries systems (King *et al.* 2015). Making decisions about the management of a fishery is difficult because scientists and decision makers, in most cases, do not have empirical knowledge about the resource prior to overexploitation, the resource users and the environment (Punt 2006). Management, according to a precautionary approach, exercises careful foresight to avoid undesirable situations whilst taking into account that changes in fisheries can be slowly reversible, difficult to control, not well understood and depend on human values (FAO 1995). Therefore, it is imperative that scientists provide decision makers with an analysis of the expected consequences of different management actions, the sensitivity of these consequences under the various assumptions of input data and to collect sufficient data that supports these analyses (FAO 1996, Kiker *et al*. 2005).

In South Africa, fisheries management relies on control measures that restrict the activities of people (i.e. gear restrictions, fishing seasons, restricted areas and bag and size limit restrictions). When these regulations are implemented, certain assumptions are made about how the fishers will respond, and the simplest assumption is that the regulations will not be violated (Punt 2005). However, despite the choice of management activities, a consistent outcome is that the fishers behave in a manner that is often unintended by the decision makers of the management system (Fulton *et al.* 2011).

Scientists and decision makers involved in fisheries management will always be faced with uncertainties and risks, however, decisions still need to be made. The failures of South African linefisheries management has been the result of a number of uncertainties when implementing management regulations. Firstly, South Africa's enforcement framework is in a dismal state because of the institutional inability to enforce management regulations (Whitfield & Cowley 2010, Sjöstedt & Sundström 2015) (see Chapter 2). This is compounded by the fact that South African estuarine fisheries are over-subscribed; yet effort and participation continue to increase (Baird *et al.* 1996, Cowley *et al*. 2013). Secondly, uncertainty and error in stock assessment approaches has resulted in inappropriate management. For example, South Africa's Linefish Management Protocol provides a variety of corrective catch and effort limiting restrictions, such as size and bag limits (Maggs *et al.* 2015). Error in life-history parameters and hence management due to the misidentification of two sympatric South African species, *A. japonicus* and *Argyrosomus inodorus* as one species, namely *Argyrosomus hololepidotus*, had a significant negative impact on the *A. japonicus* population. This is because the bag and size limits placed on *A. japonicus* (size limit of 400mm TL, and a bag limit of 10 fish.angler^{-1}.day^{-1}, first introduced in 1992) were based on incorrect life history characteristics, those of *A. inodorus*, which attains 50% sexual maturity at a considerably smaller size (males: 310 mm total length (TL); females: 340 mm TL) than *A. japonicus* (males: 920 mm TL; females: 1070 mm TL), and hence did not provide juveniles with sufficient protection (Griffiths 1997, Childs 2011) (see Chapter 3).

One of the ways to reduce uncertainty in fisheries management is to conduct decision analysis to identify research priorities (FAO 1996, Fulton *et al*. 2011, Cooke *et al*. 2014). This method which incorporates uncertainty explicitly into making choices about management actions (FAO 1996). For this chapter, a decision tree was developed under which different options for estuarine fisheries management were given for certain conditions. This is a comprehensive method that incorporates uncertainties explicitly into making appropriate choices about management actions (Keeney 1982, FAO 1996, King *et al.* 2015). The purpose of decision analysis in the form of a decision tree is to help rank alternative management actions, with considerable thought put into which options are reasonable and feasible under certain conditions (Frederick & Peterman 1995, Punt 2005).

The decision tree (Figure 5.1) provides a variety of scenarios that researchers and decision makers could follow to determine the most appropriate management application, especially estuaries with unique biological and social characteristics. The first step would be to identify the threat to the exploited fish species. Although the conservation of fish in estuaries in South Africa are threatened by various factors, including habitat degradation, disruption of ecological processes, hydrological manipulation, environmental pollution and climate change, Whitfield & Cowley (2010) identified overexploitation as the single biggest threat. This thesis focused on the threat of overfishing, hence the management appraisal provided in the form of the decision tree focused on that threat alone. Before conservation actions can be implemented, sound knowledge of the biology and ecology of the threatened species needs to be collected. According to Whitfield & Cowley (2010), Prof P. Skelton (*unpubl*. data) stated that "Research is an essential component of any conservation exercise. Conservation authorities need to know what species are threatened, why they are threatened and what the priority requirements are for the effective conservation of those species". To address the issue of uncertainty and the need for empirical data, scientists also need to incorporate information like fishing behaviour and distribution of resource users into decision making (Hilborn 2007, Futon *et al.* 2011). This can be done through robust observational roving creel surveys conducted on the fishery, like the survey conducted on the Sundays estuarine fishery by Cowley *et al*. (2013) (Chapter 2 and 3).

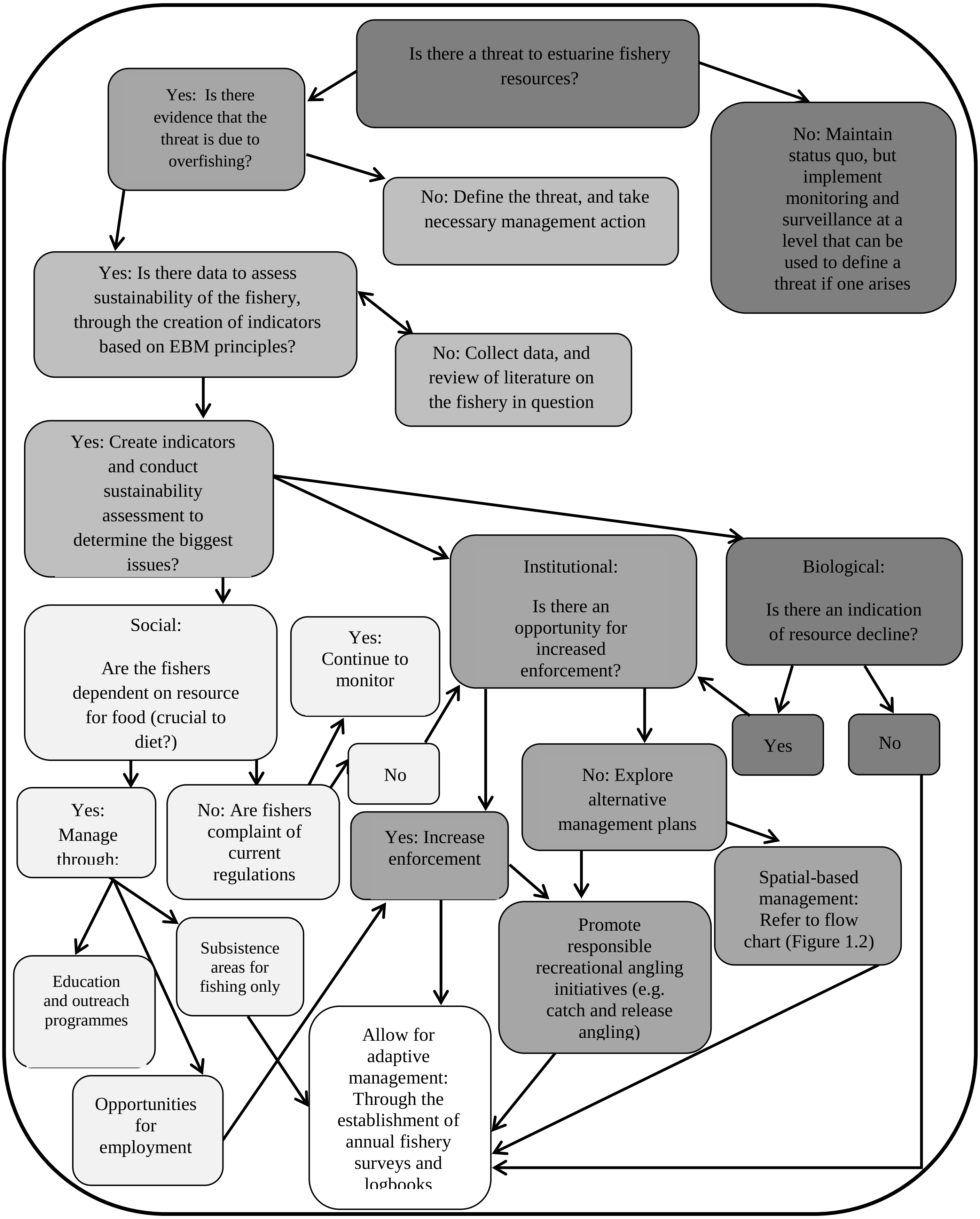

Figure 5.1: Decision tree used to identify appropriate management actions for vulnerable estuary-dependent fishery species.

Once the evidence has been collected, a precautionary approach to fisheries management can be used through the creation of a suite of appropriate indicators that can identify the threat and its causes according to biological, social or institutional domains which help guide management objectives and allow for continuous assessment through the comparison of sustainability assessment results. Given the focus of the interactions between fisher and vulnerable fishery species in this thesis, and the importance of understanding both fish and fisher behaviour in fisheries management (Arlinghaus *et al.* 2013), management interventions such as stock enhancement or habitat restoration are not discussed, but rather management of effort and catch.

In terms of social sustainability, consideration must be given to the demographics of the fishery (i.e. proportion of subsistence and recreational fishers that compose the fishery). If a fishery has a large proportion of subsistence fishers, as is the case for the Great Fish estuarine fishery (Potts *et al.* 2005), managers should consider the Small-Scale Fisheries Policy (SSFP), which promotes the continuity and growth of this sector (Branch *et al.* 2002). In order to achieve the objectives of the SSFP, subsistence fishers should have management initiatives created specifically for this sector. Harris *et al*. (2002) suggested that subsistence fishing rights should be allocated to specific areas and be exclusive to individual subsistence communities. In this way, areas of estuaries could be zoned for subsistence use, and permits issued should be valid for specific zones only. Another possible solution is to educate fishers about why the regulations are in place and why compliance to management regulations are there to safe-guard the future of their own fishing (King 2005). Alternative livelihood options could also be considered. For example, non-consumptive recreational catch-and-release fisheries, authorised via government concessions and managed by local subsistence communities might be appropriate. There is a growing trend towards developing informal regulations like education programmes led by stakeholders themselves (Cooke *et al.* 2013, Maggs *et al*. 2015). This may be a necessary approach for both recreational and subsistence fishers and can lead to higher level of compliance and knowledge of the current regulations (Maggs *et al*. 2015).

One other suggestion is to establish a training programme to involve subsistence fishers in management in order to reduce dependence on livelihood on marine resources (Harris *et al.* 2002). These training programmes could include information about vulnerable, unsustainable species, and help inform subsistence fishers about which species are sustainable. Furthermore, it has been suggested that a formalised list be created where species are classified according to their suitability for subsistence fishing (see Harris *et al.* 2002).

Recreational fishing pressure in estuaries has increased in recent decades, and this fishery often target critical life stages of species and that are vulnerable to exploitation , as is the case for the Sundays estuarine fishery (Jackson *et al.* 2001, Cowley *et al*. 2013). Recreational fish stocks targeted by recreational fishers are subject to open access and the impacts of cumulative harvesting have been a major contributor to the depletion of a variety of estuary-dependent fish species (Mann 2000, Turpie & Goss 2014). Unlike subsistence fishing, the primary goal of recreational fishing is not based on food security; therefore, management of recreational fisheries can be addressed differently.

If recreational fishers are knowledgeable of current regulations and the fishery is monitored effectively, fishers are likely to be compliant of regulations and support management initiatives. A recent study looking at alternative regulatory interventions on recreational fishing in the Breede Estuary found that a high regulatory compliance by fishers is indicative of a high level of support for regulations (Turpie & Goss 2014). Furthermore, recreational estuarine fishers have an inherent interest in the conservation and management of estuarine fisheries and show support for the implementation of additional regulations (Turpie & Goss 2014). Unfortunately, in South Africa, there are still high levels of non-compliance in most South African estuaries, which, in part, results from a lack of enforcement (Cowley *et al*. 2013). Where there is opportunity for increased enforcement with correct control and surveillance, this should be implemented. This could also have a positive feedback in terms of creating employment opportunities for subsistence fishers who rely on fishing as a source of income. For example, a mandatory requirement of hiring trained fishing guides when targeting vulnerable linefish species could be used as a management mechanism to ensure compliance of regulations (improve institutional sustainability), provide alternative income for subsistence fishers (improve socio-economic sustainability) and reduce the threat of overexploitation (improve the biological sustainability). The implementation of such measures can also be used regulate spatial and temporal trends in fishing effort and facilitate the collection data for monitoring and research purposes (Cooke *et al*. 2014). Furthermore, considering the vulnerability of estuarine associated fishery species, catch-and-release angling should be promoted (Cooke *et al.* 2014).

Whilst increased enforcement is recommended, there are limited resources dedicated to increasing enforcement of recreational estuarine fisheries in South Africa (Childs 2011). For this reason, an alternative management approach in the form of spatial and temporal regulations were explored, which would require less human capacity to enforce. Turpie & Goss (2014) suggested that no-take areas in estuaries would have a positive impact given their predicted and proven effectiveness at protecting fish stocks in the marine environment (Attwood & Bennett 1995, Johnson *et al*. 1999, Kerwath *et al*. 2013, Maggs *et al.* 2015).

Different regulations have different impacts on fishers and their catch and site choice. Certain regulations, such as spatial or temporal closures could result in loss of fisher welfare, through lack of access to fishing areas (Scrogin *et al*. 2004). Therefore, the loss of welfare needs to be balanced with the benefits of enhancement of the resource (Turpie & Goss 2014). At the Sundays Estuary, 48.3% and 25.8% of interviewed fishers said they would support seasonal and area closures respectively, which suggest that they would be willing to accept an increased degree of regulation in order to achieve an increase in fish stocks and to prevent fish stocks from declining. It has been suggested that allowing fishers to participate in the development of regulations can result in increased management success and should be kept in mind when making management decisions (Dedual *et al.* 2013, Turpie & Goss 2014). Without effective communication, fishery science may remain alienated in fisheries management and could lead to poor management actions and weak scientific insights (Dedual *et al.* 2013).

Using high resolution acoustic telemetry data on the movements of tagged juvenile *A. japonicus* in the Sundays Estuary and the spatio-temporal distribution of fishing activities on the estuary, Chapter 4 explored the use of systematic conservation planning software to identify priority areas for conservation of this vulnerable fish species. The results from this study revealed that spatial planning (identification of spatial closures/no-take areas) has merit and can be appropriate for estuarine fisheries. Although the software used wasn’t the most suitable for this exercise, there is definitely merit in integrating spatial-based tools like Marxan and acoustic telemetry for reserve designing purposes. Whilst this study focused on a single life-history stage of one species, it can be expanded to include multiple species and size classes to effectively protect estuary-dependent fish species. For example, adult *A. japonicus* and *P. commersonnii,* which make extensive use of the lower estuarine areas and show high site fidelity would benefit greatly from closed areas (Childs *et al*. 2008, Cowley *et al*. 2008, Turpie & Goss 2014). Closed seasons for the protection of adult estuarine associated fishes has been advocated, for example, Maggs *et al.* (2015) suggested that adult *L. amia* may benefit from closed seasons during their annual aggregation in Kwa-Zulu Natal (KZN). Similarly, temporal restrictions may also be a suitable method for protecting *A. japonicu*s during the spawning season (August to October) (Childs 2011). However, temporal restrictions for estuary-dependent juvenile *A. japonicus* may not be necessary as they are spatially restricted (resident) to their nursery estuary for first few years of their lives (Chapter 4).

The design of appropriate spatial-based management requires empirical data. Due to the frequent lack of consistent data, surrogate data such as habitat use, public opinion and catch data can be used as input features, and have been used in the past (Klein *et al*. 2008, Ball & Possingham 2009, Lombard *et al*. 2010, Sutcliffe *et al.* 2015). The use of surrogate data relies on the assumption that priority areas identified will adequately represent biodiversity; however there remains no clear understanding of what different factors might affect surrogate data (Hermoso *et al.* 2013). Ideally, the distribution of species across a study region would be known (Grantham *et al.* 2010). Although, acoustic telemetry data is expensive and time consuming, the empirical data gathered from it has important applications to management of fish species (Bennett *et al.* 2011). Therefore, it is recommended, if funding allows, that acoustic telemetry be integrated into spatial-based management appraisals for estuarine fish in the future.

Monitoring and evaluation is one of six original systematic conservation planning steps developed by Margules & Pressey (2000). Long-term monitoring is common in commercial fisheries; however there are limited number of long-term marine recreational and subsistence estuarine fishery data sets (Cooke & Cowx 2004, Cowley *et al*. 2013). As a result, there is no clear understanding of patterns of decline, factors influencing trends in CPUE and the effectiveness of implemented fishery regulations. Setting indictors for sustainability assessments with little data is complicated due to the lack of appropriate reference points. Continuous monitoring and research allows scientists to create appropriate indicators that can monitor and change management plans in the future.

Guidelines for marine and estuarine spatial planning have stressed the need for iterative and adaptive management (Margules & Pressery 2000, Mills *et al.* 2015). Adaptive management is advocated because it allows for decisions to be improved on, new data to be added and to accommodate for constant change in socio-economic and institutional systems (Mills *et al*. 2015). South African National Parks (SANP) has actively developed a strategic adaptive management (SAM) programme, which is the conceptual basis that supports biodiversity management within reserves (South African National Parks 2008, Holness & Biggs 2011). It is argued that an adaptive approach to systematic conservation planning improves its effectiveness in guiding the implementation of conservation initiatives (Holness & Biggs 2011) and can be continuously improved on as social and biological characteristics of the system change.

Spatial planning is often conducted as a once-off project, which results in plans quickly becoming out dated and fail to be implemented (Mills *et al.* 2015). In order to fully achieve the objectives of this thesis, this study must not be a static product, but rather the starting point for on-going adaptations. In order to achieve protection of vulnerable estuary-dependent fishery species, management approaches must be continuously revised and allow for feedback to improve understanding of effective management initiatives (Grantham *et al.* 2009). In keeping with the precautionary approach to fisheries management, it is the author's assertion that alternative spatial-based management of vulnerable estuarine-specific fish species be limited to plans for which scientific evidence is available and demonstrates that such initiatives are sustainable and the associated conservation has a net positive effect at a population level.

REFERENCES

Abecasis DMA.2013. Multispecies spatial dynamics under different protection levels: an evaluation of the effects and optimal design of the Luiz Saldanha Marine Park. PhD thesis. University of Algarve, Faro, Portugal.

Adger WN. 2000. Social and ecological resilience: are they related?. Progress in Human Geography 24: 347-364.

Agostini VA, Margles SW, Schill SR, Knowles JE, Blyther RJ.2010. Marine zoning in Saint Kitts and Nevis. *A path towards sustainable management of marine resource*s. The Nature Conservancy, Miami.

Allahyari MS.2010. Fisheries sustainability assessment in Guilan province, Iran. *Journal of Food, Agriculture and Environment* 8:1300-1304.

Allnutt TF, McClanahan TR, Andréfouët S, Baker M, Lagabrielle E, McClennen C.2012.Comparison of Marine Spatial Planning Methods in Madagascar Demonstrates Value of Alternative Approaches. *PLoS ONE* 7:28969. doi:10.1371/journal.pone.0028969.

Ardron JA, Possingham HP, Klein CJ.2010. Marxan Good Practices Handbook, Version 2. *Pacific Marine Analysis and Research Association*, Victoria, BC, Canada.

Arlinghaus R, Cooke SJ, Potts W. 2013. Towards resilient recreational fiseries on a global scale through improved understanding of fish and fisher behaviour. *Fisheries Management and Ecology* 20:91-98.

Attwood CG, Bennett BA.1995. A procedure for setting daily bag limits on the recreational shore-fishery of the south-western Cape, South Africa. *South African Journal of Marine Science* 15:241-251.

Attwood CG, Harris JM, Williams AJ.1997. International experience of marine protected areas and their relevance to South Africa. *South AfricanJournal of Marine Science* 18:311-332.

Attwood CG, Mann BQ, Beaumont J, Harris JM.1997. Review of the state of marine protected areas in South Africa. *South African Journal of Marine Science* 18:341-367.

Attwood CG.2003. Dynamics of the fishery for galjoen Dichistius capensis, with an assessment of monitoring methods. *African Journal of Marine Science* 25:311-330.

Attwood CG, Harris JM, Williams AJ.1997. International experience of marine protected areas and their relevance to South Africa. *South African Journal of Marine Science* 18:311-332.

Bennett BA, Attwood CG.1993. Shore-angling catches in the De Hoop Nature Reserve, South Africa, and further evidence for the protected value of marine reserves. *South African Journal of Marine Science* 13:213-222.

Baeta F, Pinheiro A, Corte-Real M, Costa JL, de Almeida PR, Cabral H, Costa MJ.2005. Are the fisheries in the Tagus estuary sustainable? *Fisheries Research* 76:243-251.

Baird D, Marais JFK, Daniel C.1996. Exploitation and conservation of angling fish species in selected South African estuaries. Aquatic Conservation: *Marine and Freshwater Ecosystem* 6:319-330.

Ball I, Possingham H, Watts M.2009. Marxan and relatives: software for spatial conservation prioritisation. In: Moilanen A. Wilson, Possingham HP (eds). *Spatial Conservation Prioritisation*. Oxford University Press, Oxford.

Ball Ian, Possingham Hugh.2000. MARXAN (v1.8.2): Marine Reserve Design Using Spatially Explicit Annealing. *A Manual Prepared for the Great Barrier Reef Marine Park Authority*.

Ball IR, Possingham HP, Watts M.2009. Marxan and relatives: Software for spatial conservation prioritisation. In: Moilanen A, Wilson KA, Possingham HP (eds). Spatial conservation prioritisation: *Quantitative methods and computational tools*. Oxford University Press, Oxford, UK.

Ban NC, Klein CJ.2009. Spatial socioeconomic data as a cost in systematic marine conservation planning. *Conservation Letters* 2:206-215.

Beck MW, Heck Jr. KL, Able KW, Childers D, Eggleston D, Gillanders BM, Halpern B, Hays C, Hoshino K, Minello T, Orth R, Sheridan P, Weinstein M.2001. The identification, conservation, and management of estuarine and marine nurseries for fish and invertebrates. *Biological science* 51:33-641.

Beck MW, Odaya M.2001. Ecoregional planning in marine environments—identifying priority sites for conservation in the northern Gulf of Mexico. *Aquatic Conservation* 11:235-242.

Bennett BA, Attwood CG.1991. Evidence for the recovery of a surf zone fish assemblage following the establishment of a marine reserve on the southern coast of South Africa. *Marine Ecology Progress Series* 75:173-181.

Bennett BA, Attwood CG.1991. Evidence for recovery of a surf-zone fish assemblage following the establishment of a marine reserve on the southern coast of South Africa. *Marine Ecology Progress Series* 75:173-181.

Bennett RH, Childs AR, Cowley PD, Naesje TF, Thorstad EB, Okland F.2011. First assessment of estuarine space use and home range of juvenile white steenbras, Lithognathus lithognathus. *African Zoology* 46:32-38.

Bennett RH, Cowley PD, Childs A-R, Næsje, TF.2015. Movements and residency of juvenile white steenbras Lithognathus lithognathus in a range of contrasting estuaries. *Estuarine, Coastal and Shelf Science* 152:100-108.

Bennett RH, Cowley PD, Childs A-R, Whitfield AK.2012. Area-use Patterns and Diel Movements of White Steenbras Lithognathus Lithognathus in a Temporarily Open/Closed South African Estuary, Inferred from Acoustic Telemetry and Long-term Seine-netting Data. *African Journal of Marine Science* 34:81-91.

Bennett, RH.2012. Movements patterns, stock delineation and conservation of an overexploited fisher species, Lithognathus lithognathus (Pisces: Sparidae). PhD Thesis. Rhodes University, Grahamstown, South Africa.

Blaber SJM.2002. ‘Fish in hot water’: the challenges facing fish and fisheries research in tropical estuaries. *Journal of Fish Biology* 61:1-20.

Bowen RE, Riley C.2003. Socio-economic indicators and integrated coastal management. *Ocean and Coastal Management* 46:229-312.

Branch GM, Hauck M, Siqwana-Ndulo N, Dye AH.2002. Defining in the South African context: subsistence, artisanal and small-scale commercial sectors. *South African Journal of Marine Science* 24:475-48.

Branch TA, Watson R, Fulton EA, Jennings S, McGilliard CR, Pablico GT, Ricard D,Tracey SR.2012. The trophic fingerprint of marine fisheries. *Nature* 468:431-435.

Branch GM, Hauck M, Siqwana-Ndlovu N, Dye A.H.2002. Defining fishers in the South African context. *South African Journal of Marine Science* 24:475-487.

Brouwer .SL.1997.An Assessment of the South African East Coast Linefishery from Kei Mouth to Stil Bay. MSc. Thesis, Rhodes University.

Brouwer SL, Mann BQ, Lamberth SJ, Sauer WHH, Erasmus C.1997. A survey of the South African shore-angling fishery. *South African Journal of Marine Science* 18:165-177.

Butterworth DS, Punt AE, Borchers DL, Pugh JB, Hughs GS.1989. A manual ofmathematical techniques for linefish assessment. *South African natural scientific Programmes* 160:1-83.

Campbell HA, Dwyer RG, Fitzgibbons S, Klein CJ, Lauridsen G, McKeown A, OLSSON a, Sullivan S, Watts ME, Westcott DA. 2012. Prioritising the protection of habitat utilised by southern cassowaries Casuarius casuarius johnsonii. *Endangered Species Research* 17:53-61.

Carwardine J, Rochester WA, Richardson KS, Williams KJ, Pressey RL, Possingham HP. 2007. Conservation planning with irreplaceability: does the method matter*? Biodiversity and Conservation* 16: 245–258.

Carwardine J, Wilson KA, Hajkowicz SA, Smith RJ, Klein CJ, Watts M, Possingham HP. 2010. Conservation planning when costs are uncertain. *Conservation Biology* 24: 1529-1537.

Carwardine J, Wilson KA, Ceballos G.2008.Cost-effective priorities for global mammal conservation. *Proceedings of the National Academy of Science USA* 105:11446-11450.

Castro NG.2001. Monitoring ecological and socio-economic indicators for coral reef management in Colombia. *Bulletin of Marine Science* 69: 847-859.

Chalmers R.2012. Systematic marine spatial planning and monitoring in a data poor environment: a case study of Algoa Bay, South Africa. PhD Thesis. Rhodes University, Grahamstown, South Africa.

Childs A-R, Cowley PD, Naesje TF, Booth AJ, Potts WM, Thorstad EB, Okland F.2008. Estuarine use by spotted grunter Pomadasys commersonnii in a South African estuary, as determined by acoustic telemetry. *African Journal of Marine* Science 30:123-132.

Childs A-R.2011. IUCN Red List Assessment for dusky kob, *Argyrosomus japonicus*, compiled for the National Biodiversity Assessment 2011: *Marine and Coastal component.*

Childs A-R.2013. Estuarine-dependency and multiple habitat use by dusky kob *Argyrosomus japonicus* (pisces: sciaenidae). PhD Thesis. Rhodes University, Grahamstown, South Africa.

Childs A-R, Cowley PD, Naesje TF. Bennett RH.2015. Habitat connectivity and intra-population structure of an estuary-dependent fishery species. *Marine Ecology Progress Series* 537:233-245.

Clarke JR, Buxton CD.1989. A survey of the recreational rock-angling fishery at Port Elizabeth, on the south-east coast of South Africa. *South African Journal of marine Science* 8:183–194.

Cochrane KL, Augustyn CJ, Cockcroft AC, David JHM, Griffiths MH, Groeneveld JC, Lipinski MR, Smale MJ, Smith CD, Tarr JQ.2004. An ecosystem approach to fisheries in the southern Benguela context.*African Journal of Marine Science* 26:9-35.

Coetzee PS, Baird D, Tregononing C.1989. Catch statistics and trends in the shore angling fishery of the east coast, South Africa, for the period 1959–1982. *South African Journal of marine Science* 8:155-171.

Cooke SJ, Cowx IG.2004. The role of recreational fishing in global fish crises. *Biological Science* 54:857-859.

Costanza R, D'Arge R, De Groot R, Farber S, Grasso M, Hannon B, Limburg K, Naeem S, O'Neill R, Paruelo J, Raskin RG, Sutton P, van den Belt M.1997. The value of the world's ecosystem services and natural capital. *Nature* 387:253-259.

Cowley PD, Brouwer SL Tilney RL.2002. The role of the Tsitsikamma National Park in the management of four shore-angling fishes along the south-eastern Cape coast of South Africa. *South African Journal of Marine Science* 24:27-35.

Cowley PD, Childs A-R, Bennett RH.2013. The trouble with estuarine fisheries in temperate South Africa, illustrated by a case study on the Sundays Estuary. *African Journal of Marine Science* 35:117-128.

Cowley PD, Kerwath SE, Childs A-R, Thorstad, Okland F, Naesje TF.2008. Estuarine habitat use y juvenile dusky kob Argyrosomus japonicus (Scienidae), with implications for managent. *African Journal of Marine Science* 30:247-253.

Cowley PD, Næsje TF, Childs A-R, Chittenden CM, Bennett RH.2011. The influence of extreme weather events on the behaviour of estuarine associated fishes. *First International Conference on Fish Telemetry. Book of Abstracts.* Hokkaido University, Sapporo, Japan.

Cowley PD, Naesje TF, Childs A-R, Bennett RH, Chittenden CM, Hedger R. 2013. Does the restricted movement paradigm apply to the estuarine-dependent spotted grunter *Pomadasys commersonnii*? In: A Decade after the Emergency: The Proceedings of the 4th Linefish Symposium. Attwood C, Booth T. Kerwath S, Mann B, Marr S, Duncan J, Bonthuys J, Potts WM (eds). WWF South Africa Report Series – 2013/Marine/001

Dahl AL.2000. Using indicators to measure sustainability: recent methodological and conceptual developments. *Marine and Freshwater Research* 51:427-433.

da Silva C, Kerwath SE, Attwood CG, Thorstad EB, Cowley PD, Økland F, Wilke CG, Næsje TF.2013. Quantifying the degree of protection afforded by a no-take marine reserve on an exploited shark. *African Journal of Marine Science* 35:57-66.

DAFF.2012.Policy for the small scale fisheries sector in South Africa. Government Gazette Vol. 474, No. 35455. Notice. 474. Department of Agriculture, Forestry and Fisheries, Republic of South Africa.

DAFF.2013. Draft revised traditional linefish policy on the allocation and management of fishing rights. Government Gazette vol. 575, No. 36460, Notice 473. Department of Agriculture, Forestry and Fisheries: Branch Fisheries Management, Republic of South Africa.

Dames M. 2014. Area use and movement behaviour of *Argyrosomus japonicus* (Pisces: Sciaenidae) in the Sundays Estuary, Eastern Cape, South Africa. Honours thesis, Rhodes University, Grahamstown, South Africa.

Day J.2008. The need and practice of monitoring, evaluating and adapting marine planning and management—lessons from the Great Barrier Reef. Marine Policy 32:823-31.

DEA. 2014. National Environmental Management: Protected Areas Act. Government Gazette. Vol. 589, No. 37802, Notice. 528. Department of Environmental Affairs, Republic of South Africa.

DEAT. 2000. General Notice in terms of section 16 of the Marine Living Resources Act, 1998 (Act No. 18 of 1998). Government Gazette Vol. 426, No. 21949, Notice 4727. Department of Environmental Affairs and Tourism, Republic of South Africa.

DEAT.2004. Amendments to regulations published in terms of section 44 of the national environmental management act, 1998: control of use of vehicles in the coastal zone (gn regulation 1399 of 21 December 2001). Government Gazette Vol. 474, No. 27066, Notice 8113. Department of Environmental Affairs and Tourism, Republic of South Africa.

DEAT. 2005. Marine Living Resources Act 1998 (Act No. 18 of 1998). Government Gazette Vol. 484, No. 1012, Notice 28129. Department of Environmental Affairs and Tourism, Republic of South Africa.

DEAT. 2010. National protected areas expansion strategy for South Africa 2008. Department of Environmental Affairs, Government of South Africa, Pretoria.

Dedual M, Sague Pla O, Arlinghaus R, Clarke A, Ferter K, Hansen Geertz P, Gerdeaux D, Hames F, Kennelly SJ.2013. Communication between scientists, fishery managers and recreational fishers: lessons learned from a comparative analysis of international case studies. *Fisheries Management and Ecology* 20:234-246.

Douvere F.2010. Marine spatial planning: Concepts, current practice and linkages to other management approaches. PhD Thesis, Ghent University, Belgium.

Dunlop SW, Mann BQ, Cowley PD, Murray TS, Maggs JQ.2014. Movement patterns of Lichia amia (Teleostei: Carangidae): results from a long-term cooperative tagging project in South Africa. *African Zoology* 50:249-257.

Egli DP, Babcock RC. 2004. Ultrasonic tracking reveals multiple behavioural modes of snapper (*Pagrus auratus*) in a temperate no-take marine reserve. *ICES Journal of Marine Science* 61:1137-1143.

Ehler C, Douvere F.2009. Marine spatial planning: A step-by-step approach toward ecosystem-based management. *Intergovernmental Oceanographic Commission and Man and the Biosphere Programme*, No. 53, IOCAM Dosier No. 6, Paris, UNESCO.

Ehrenberg JE, Steig TW, Greene CH, Brosnan I.2014. Using Acoustic Telemetry to Measure Fine-Scale Movement & Interactions of Marine Animals: Implications of Marine Protected Areas. *ICES Annual Science Conference*. A Coruña, Spain.

Everett BI, Fennessy ST.2007. Assessment of recreational boat-angling in a large estuarine embayment in KwaZulu-Natal, South Africa. *African Journal of Marine Science* 29:411-422.

FAO (Code of Conduct for Responsible Fisheries).1995. FAO Fisheries Technical Paper 350. http://www.fao.org/WAICENT/FAOINFO/FISHERY/agreem/codecond/codecon.asp

FAO (Food and Agricultural Organisation).1996. Precautionary approach to fisheries. FAO Fisheries Technical Paper No. 350, Part 2. Rome, Italy.

FAO (Food and Agricultural Organisation).1997. Fisheries Management. FAO Technical Guidelines for Responsible Fisheries. FAO fisheries department, Rome, Italy.

FAO (Food and Agricultural Organisation).2001. Reykjavik Conference on Responsible Fisheries in the Marine Ecosystem. Iceland, 1–4 October 2001. http://www.refisheries2001.org/.

FAO (Food and Agricultural Organisation). 2002. Indicators for sustainable development of fisheries. FAO fisheries department, Rome, Italy.

FAO (Food and Agricultural Organisation).2003. Fisheries Management. 2. The ecosystem approach to fisheries. FAO Technical guidelines for responsible fisheries. FAO, Rome, Italy.

FAO (Food and Agricultural Organisation). 2005. Putting into practice the ecosystem approach to fisheries. FAO fisheries department, Rome, Italy.

FAO (Food and Agricultural Organisation).2010. The state of world fisheries and aquaculture. FAO, Rome, Italy.

Fernandes L, Day J, Lewis A, Slegers S, Kerrigan B, Breen D, Cameron D, Jago B, Hall J, Lowe D, Innes J, Tanzer J, Chadwick V, Thompson L, Gorman K, Simmons M, Barnett B, Sampson K, De'ath G, Mapstone B.2005. Establishing representative no-take areas in the Great Barrier Reef: Large-scale implementation of theory on marine protected areas. *Conservation Biology* 19: 1733–1744.

Ferraro PJ.2003. Assigning priority to environmental policy interventions in a heterogeneous world. *Journal of Policy Analysis and Management* 22:27-43.

Fischer DT, Church RL.2005.The SITES reserve selection system: A critical review. *Environmental Modeling and Assessment* 10:215-228.

Fletcher WJ, Chesson J, Sainsbury KJ, Hundloe TJ, Fisher M. 2005. A flexible and practical framework for reporting on ecologically sustainable development for wild capture fisheries. *Fisheries Research* 71:175-183.

Frederick SW Peterman RM.1995. Choosing fisheries harvest policies: when does uncertainty matter? Can. *Journal of Fish Aquatic Science* 52:291-306.

Frisk, M.G. & Miller, T.J. 2005. Life Histories and Vulnerability to Exploitation of Elasmobranchs: Inferences from Elasticity, Perturbation and Phylogenetic Analyses. *Journal of Northwest Atlantic Fishing Science*. 35:27-45.

Fulton EA, Smith ADM, Smith DC, van Putten AE. 2011. Human behaviour: the key source of uncertainty in fisheries management. *Fish and Fisheries* 12: 2-17.

Gislason H. 2001. The effects of fishing on non-target species and ecosystem structure and function. *Reykjavik Conference on Responsible Fisheries in the Marine Ecosystem.* Reykjavik, Iceland.

Game ET, Grantham HS.2008. Marxan User Manual: For Marxan version 1.8.10. University of Queensland, Australia, and Pacific Marine Analysis and Research Association, British Columbia, Canada.

Garcia SM, Kolding J, Rice J, Rochet MJ, Zhou S, Arimoto T, Beyer JE, Borges L, Bundy A, Dunn D, Fulton EA, Hall M, Heino M, Law R, Makino M. Rijinsdorp AD, Simard F, Smith ADM. 2012. Reconsidering the consequences of selective fisheries. *Science* 335:

1045-1047.

Garcia SM, Staples D. 2000. Sustainability indicators in marine capture fisheries: introduction to the special issue. *Marine and Freshwater Research* 51:381-384.

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

Garcia SM.2000. The FAO definition of sustainable development and the Code of Conduct for Responsible Fisheries: an analysis of the related principles, criteria and indicators. *Marine and Freshwater Research* 51:535-541.

Geselbracht L, Torres R, Cumming GS, Dorfman D, Beck M, Shaw D.2009. Identification of a spatially efficient portfolio of priority conservation sites in marine and estuarine areas in Florida. *Marine and Freshwater Ecosystems* 19:408-420.

Gillanders BM, Elsdon TS, Roughan M. 2012. Connectivity of Estuaries. In: Wolanski E, McLusky DS (eds), *Treatise on Estuaries and Coasts*. Amsterdam: Elsevier.

Gilliland PM, Laffoley D.2008. Key elements and steps in the process of developing ecosystem-based marine spatial planning. *Marine Policy* 32:787-796.

Gislason, H. 2001. The effects of fishing on non-target species and ecosystem structure and function. *Reykjavik conference on Responsible fisheries in the Marine Ecosystem* Reykjavic, Iceland.

Grantham HS, Bode M, McDonald-Madden E, Game ET, Knight AT, Possingham HP.2009. Effective conservation planning requires learning and adaptation. *Frontiers in Ecology and the Environment* 8:431–437.

Grantham HS, Game ET, Lombard A, Hobday AJ, Richardson AJ, Beckley LE, Pressey RL, Huggett JA, Coetzee JC, van der Lingen CD, Petersen SL, Merkle D, Possingham HP.2011. Accommodating dynamic oceanographic processes and pelagic biodiversity in marine conservation planning. *Plos ONE* 6:1-16.

Griffiths MH.1996. Life-history of the dusky kob Argyrosomus japonicus (Sciaenidae) off the east coast of South Africa. *South African Journal of Marine Science* 17:135-154.

Griffiths MH.1997. Management of the South African dusky kob Argyrosomus japonicus (Sciaenidae) based on per-recruit models. *South African Journal of Marine Science* 18:213-228.

Griffiths MH. 1999. Dusky kob management plan. In Proceedings of Linefish Workshop to Review Current Management Regulations, Cape Town, *Unpublished report, Marine &Coastal Managemen*t, Cape Town.

Griffiths MH, Attwood CG, Thompson R. 1999. New management protocol for the South African linefishery. In: Mann BQ (ed.), Proceedings of the third Southern African Marine Linefish Symposium, 28 April–1 May 1999, Arniston, South Africa. SANCOR Occasional

Report no. 5. Arniston: *South African Network for Coastal and Oceanic Research* 145-156.

Griffiths MH. 2000. Long-term trends in catch and effort of commercial linefish off South Africa's Cape Province: Snapshots of the 20th century. *South African Journal of Marine Science* 22:81-110.

Griffiths MH, Lamberth SJ.2002. Evaluating the marine recreational fishery in South Africa.In: Pitcher TJ, Hollingworth C (eds).*Recreational Fisheries Ecological, Economic and Social Evaluation*. Blackwell Science Ltd., United Kingdom.

Hak T, Moldan B, Dahl AL. 2012.Editorial Ecological Indicators 17:1-3.

Heino M, Godø OR.2002. Fisheries-induced selection pressures in the context of sustainable fisheries. Bulletin of Marine Science 70:639-656.

Halpern BS, McLeod KL, Rosenberg AA, Crowder LB.2008. Managing for cumulative impacts in ecosystem-based management through ocean zoning?. *Marine Policy* 51:203-211.

Halpern, B.S., Selkoe, K.A., Micheli, F. and Kappel, C.V.2007. Evaluating and ranking the vulnerability of global marine ecosystems to anthropogenic threats. *Conservation Biology* 21:1301-1315.

Harris JM , Branch GM, Clark BM, Cockcroft AC , Coetzee C, Dye AH , Hauck M, Johnson A, Kati-Kati L, Maseko Z, Salo K, Sauer WHH, Siqwana-Ndulo N, Sowman M.2002. Recommendations for the management of subsistence fisheries in South Africa. *South African Journal of Marine Science* 24:503-523.

Harris LR. 2012. An ecosystem-based spatial conservation plan for the South African sandy beaches. PhD thesis. Nelson Mandela Metropolitan University, South Africa.

Hermoso V, Cattarino L, Kennard MJ, Watts M, Linke S.2015. Catchment zoning for freshwater conservation: refining plans to enhance action on the ground. *Journal of Applied Ecology* 52:940-949.

Heupel MR, Semmens JM, Hobday AJ.2006. Automated acoustic tracking of aquatic animals: scales, design and deployment of listening station arrays. *Marine and Freshwater Research* 57:1-13.

Heupel MR, Semmens JM, Hobday AJ.2006. Automated acoustic tracking of aquatic animals: scales, design and deployment of listening station arrays. *Marine and Freshwater Research* 57:1-13.

Hilborn R, Stokes K, Maguire J, Smith T, Botsford LW, Mangal M, Orensanz J, Parma A, Rice J, Bell J, Cochrane KL, Garcia S, Hall SJ, Kirkwood GP, Sainsbury K, Stefansson G, Walters C. 2004. When can marine reserves improve fisheries management? *Ocean and Coastal Management* 47:197-205.

Hilborn R. 2007. Defining success in fisheries and conflicts in objectives. *Marine Policy* 31:153-158.

Holness SD, Biggs HG.2011.Systematic conservation planning and adaptive management. *Koedoe* 53:34-42.

Hughs TP, Bellwood DR, Folke C, Steneck RS, Wilson J. 2005. New paradigms for supporting the resilience of marine ecosystems. Trends in Ecology and Evolution 20: 380-386.

ICMA (Integrated Coastal Management Act).2008. Act 24 of 2008. South Africa. Government Gazette Vol. 524, No. 31884 of 2009.

Jennings S, Lock JM. 1996. Population and ecosystem effects of reef fishing. In: Polunin, N. V. C. & C. M Roberts (eds). *Reef Fisheries*. Chapman and Hall, London.

Johnson DR, Funicelli NA, Bohnsack JA. 1999. Effectiveness of an existing estuarine no-take fish sanctuary within the Kennedy Space Center, Florida. *North American Journal of Fisheries Management* 19:436-453.

Jones KMM.2005. Home ranges areas and activity centres in six species of Caribbean wrasses (Labridae). *Journal of Fish Biology* 66:150-166.

Kark S, Levin N, Grantham HS, Possingham HP. 2009. Between-country collaboration and consideration of costs increase conservation planning efficiency in the Mediterranean Basin. *Proceedings of the National Academy of Sciences* 106:15368-15373.

Keeney RL.1982. Decision analysis: An overview. *Operations Research* 30:803-838.

Kerwath SE, Götz A, Attwood CG, Sauer WHH,Wilke CG.2007.Area utilisation and activity patterns of roman Chrysoblephus laticeps (Sparidae) in a small marine protected area. *African Journal of Marine Science* 29:259-270.

Kerwath SE, Thorstad EB, Naesje TF, Cowley PD, Okland F, Wilke C, Atwood CG. 2009. Crossing Invisible Boundaries: the Effectiveness of the Langebaan Lagoon Marine Protected Area as a Harvest Refuge for a Migratory Fish Species in South Africa. *Conservation Biology* 23:653-661.

Kerwath SE, Winker H, Götz A, Attwood CG.2013. Marine protected area improves yield without disadvantaging fishers. *Nature Communications 4: 2347* doi:10.1038/ncomms3347.

Kiker GA, Bridges TS, Varghese A, Seager PT, Linkov I.2005. Application of multicriteria decision analysis in environmental decision making. *Integrated Environmental Assessment and Management* 1:95-108.

Killmer AW, Harrison TH.2015.Real-world progress in overcoming the challenges of adaptive spatial planning in marine protected areas. *Biological Conservation* 181:54-63.

King JR, McFarlane GA, Punt AE.2015. Shifts in fisheries management: adapting to regime shifts. *Philosophical Translations*. 370: 20130277.

Klein CJ, Chan A, Kircher L, Cundiff AJ, Gardner N, Hrovat Y, Scholz A, Kendall BE, Airamé S.2008. Striking a balance between biodiversity conservation and socioeconomic viability in the design of marine protected areas. *Conservation Biology* 22:691-700.

Klein CJ, Chan A, Kircher L, Cundiff A, Gardner N, Hrovat Y, Scholz A, Kendall B, Airame S.2008. Striking a balance between biodiversity conservation and socioeconomic viability in the design of marine protected areas. *Conservation Biology* 22:691-700.

Knol M.2010. Scientific advice in integrated ocean management: the process towards the Barents Sea plan. *Marine Policy* 34:252-260.

Kyle R.1988. Aspects of the ecology and exploitation of the fishes of the Kosi Bay system, KwaZulu, South Africa. Ph.D. thesis, University of Natal, Pietermaritzburg.

Kremen C.2008.Conservation with caveats—response. *Science* 321:341-342.

Lamberth S, Turpie J.2003. The role of estuaries in South African fisheries: economic importance and management implications. *African Journal of Marine Science* 25: 131-157.

Lamberth SJ, Joubert AR.2014. Prioritising species for research, conservation and management: a case study of exploited fish species. *African Journal of Marine Science* 36:345-360.

Lamberth SJ, Turpie JK.2003. The Role of Estuaries in South African Fisheries:Economic Importance and Management Implications. *African Journal of Marine Science* 25:131-157.

Lee DE, Stephen GH, Mario Du Preez. 2013. Valuing User Preferences for Improvements in Public Nature Trails Around the Sundays River Estuary, Eastern Cape, South Africa. *Economic Research Southern Africa* (*ESRA*) 53:1-21.

Lee DE, Hosking SG Du Preez MA. 2014. Choice experiment application to estimate willingness to pay for controlling excessive recreational fishing demand at the Sundays River Estuary, South Africa. *Water SA* 40:39-40.

Leslie H, Ruckelshaus M, Ball IR.2003. Using siting algorithms in the design of marine reserve networks. *Ecological Applications* 13:185-90.

Lester, S., B. Halpern, and K. Grorud-Colvert. 2009. Biological effects within no-take marine reserves: a global synthesis. *Marine Ecology Progress Series* 384:33-46.

Levin N, Watson EM, Joseph N, Grantham HS, Hadar L, Apel e N, Perevolotsky A, DeMalach N, Possingham HP, Kark S.2013. A framework for systematic conservation planning and management of Mediterranean landscapes. *Biological Conservation* 158: 371-383.

Levin N, Watson JEM, Jospeh LN, Grantham HS, Hadar L, Apel N, Perevolotsky A, DeMalach N, Possingham HP, Kark S.2013. A framework for systematic conservation planning and management of Mediterranean landscapes. *Biological Conservation* 158: 371-383.

Levin PS, Mollmann C.2015. Marine ecosystem regime shifts: challenges and opportunities for ecosystem-based management. *Philosophical Transactions* 370:2013-0275.

Lieberknecht LM, Carwardine J, Connor DW, Vincent MA, Atkins SM, Lumb CM. 2004. The Irish Sea Pilot - Report on the Identification of Nationally Important Marine Areas in the Irish Sea. *JNCC* Report no. 347:28-36.

Liebold M, van Zyl CJ.2008. The economic impact of sport and recreational angling in the Republic of South Africa, Development Strategies International PTY(Ltd), Cape Town.

Linke S, Turak E and Nel JL.2011. Freshwater conservation planning: The case for systematic approaches. *Freshwater Biology* 56:6-20.

Lombard AT, Reyers B, Schongevel LY, Cooper J, Smith-Adao LB, Nel DC, Froneman PW, Ansorge IJ, Bester MN, Tosh CA, Strauss T, Akkers T, Gon O, Leslie RW, Chown SL.2007. Conserving pattern and process in the Southern Ocean: designing a marine protected area for the Prince Edward Islands. *Antarctic Science* 19: 39-54.

Lombard AT, Attwood C, Sink K, Grantham H.2010. Use of Marxan to identify potential closed areas to reduce bycatch in the South African inshore trawl fishery. Technical Report for WWF South Africa and the Responsible Fisheries Alliance. *WWF South Africa.*

Lombard AT, Strauss T, Harris J, Sink K, Attwood C, Hutchings L.2004. South African National Spatial Biodiversity Assessment Technical Report. Volume 4: Marine Component. South African National Biodiversity Institute (SANBI), Pretoria.

Lombard AT,Attwood C, Sink K, Grantham H.2010. Use of Marxan to identify potential closed areas to reduce bycatch in the South African inshore trawl fishery. Technical Report for WWF South Africa and the Responsible Fisheries Alliance. *WWF South Africa.*

Loos S. 2001. Exploration of MARXAN for Utility in Marine Protected Area Zoning. Master's thesis, University of Victoria, Australia.

Mackay HM, Schumann EH.1990. Mixing and circulation in the Sundays River Estuary, South Africa. *Estuarine, Coastal and Shelf Science* 31:203-216.

Maggs JQ, Mann BQ, Cowley PD.2013. Contribution of a large no-take zone to the management of vulnerable reef fishes in the South-West Indian Ocean. *Fisheries Research* 144:38-47.

Maggs JQ, Mann BQ, Potts WM, Dunlop SW. 2015. Traditional management strategies fail to arrest a decline in the catch-per-unit-effort of an iconic marine recreational fishery species with evidence of hyperstability. *Fisheries Management and Ecology*. DOI: 10.1111/fme.12125.

Mann BQ (eds).2000. Southern African marine linefish status reports. *Oceanographic Research Institute Special Publication* No. 7: 257.

Mann BQ (eds).(2013). Southern African marine linefish species profiles. *Oceanographic Research Institute Special Publication* No. 9: 343.

Mann BQ, Cowley PD, Fennessy ST. 2015. Movement patterns of surf-zone fish species in a sub-tropical World Heritage Site on the east coast of South Africa. *African Journal of Marine Science* 37:99-114.

Marais JFK.1981. Seasonal abundance, distribution, and catch per unt effort using gill-nets, of fishes in the Sundays estuary. *South African Journal of Zoology* 16:144-150.

Margules CR, Pressey RL.2000. Systematic conservation planning. *Nature* 405: 243-253.

Marshell A, Mills JS, Rhodes KL, McIlwain J.2011. Passive acoustic telemetry reveals highly variable home range and movement patterns among unicornfish within a marine reserve. *Coral Reefs* 30: 631-642.

McCord M, Lamberth SJ.2009. Catching and tracking the world's largest Zambezi (bull) shark Carcharhinus leucas in the Breede Estuary, South Africa: the first 43 hours. African *Journal of Marine Science* 31:107-111.

Mills M, Weeks R, Pressey RL, Gleason MG, Eisma-Osorio RL, Lombard AT, Harris JM, Mo¨llmann C, Lindegren M, Blenckner T, Bergstro¨m L, Casini M, Diekmann R, Flinkman J, Mu¨ller-Karulis B, Neuenfeldt S, Schmidt J. O, Tomczak M, Voss R, Ga˚rdmar A.2013.Implementing ecosystem-based fisheries management: from single-species to integrated ecosystem assessment and advice for Baltic Sea fish stocks. *Journal of Marine Science* doi:10.1093/icesjms/fst123.

Musick JA.1999. Criteria to define extinction risk in marine fishes. *Fisheries* 24: 6-14.

Naesje TF, Childs A-R, Cowley PD, Potts WM, Thorstad EB, Okland F.2007. Movements of undersized spotted grunter (Pomadasys commersonnii) in the Great Fish Estuary, South Africa: implications for fisheries management. *Hydrobiologia* 582:25-34.

Noss RF.1990. Indicators for monitoring biodiversity: A hierarchical approach. *Conservation Biology* 4:355-364.

Nsubuga YN. 2004. Towards sustainable utilisation of the fishery resources of the Kowie Estuary, South Africa. MSc Thesis. Rhodes University, Grahamstown, South Africa.

OECD. 1993. OECD Core Set of Indicators for Environmental Performance Reviews. *Organisation for Economic Co-operation and Development*, Paris, France.

Oosthuizen A, Holness S, Chalmers R.2011. Draft Management Plan for the Proposed Marine Protected Area: Addo Elephant National Park. *Prepared for Park Planning & Development*, South African National Parks.

Parsons DM, Babcock RC, Hankin RKS, Willis TJ, Aitken JP, O'Dor RK, Jackson GD.2003. Snapper Pagrus auratus (Sparidae) home range dynamics: acoustic tagging studies in a marine reserve. *Marine Ecology Progress Series* 262:253-265.

Pajak P.2000. Sustainability, ecosystem management, and indicators: thinking globally and acting locally in the 21st century. *Fisheries* 25:16-30.

Pauly, D. 1995. Anecdotes and the shifting base-line of fisheries. *Trends in Ecology and Evolution*, 10:430.

Pauly D.1997. When is fisheries management needed? In: Adams TJH, Dalzell PJ, Robert SA (eds). South Pacific Commission and Forum Fisheries Agency Workshop on the Management of South Pacific Inshore Fisheries 3. Nouméa, New Caledonia.

Pitcher TJ, Preikshot D.2001. RAPFISH: a rapid appraisal technique to evaluate the sustainability status of fisheries. *Fisheries Research* 49:255-270.

Pikitch EK, Santora C, Babcock EA, Bakun A, Bonfil R, Conover DO, Dayton P, Doukakis P, Fluharty D, Heneman B, Houde ED, Link J, Livingston PA, Mangel M, McAllister MK, Pope J, Sainsbury KJ. 2004. Ecosystem-based fishery management. *Science* 305: 346-347.

Pitcher TJ, Lam ME, Ainsworth C, Martindale A, Nakamura K, Perry RI, Ward T.2013. Improvements to Rapfish: a rapid evaluation technique for fisheries integrating ecological and human dimensions. *Journal of fish biology* 83: 865-889.

Polunin NVC.2002. Marine protected areas, fish and fisheries. In: Hart PJB, Reynolds JD (eds) *Handbook of Fish Biology and Fisheries*. Blackwell Publishing, UK.

Pomeroy RS, Watson LM, Parks JE, Cid GA.2005. How is your MPA doing? A methodology for evaluating the management effectiveness of marine protected areas. *Ocean and Coastal Management* 48: 485-502.

Possingham H, Ball I, Andelman S.2000. Mathematical Methods for Identifying Representative Reserve Networks 291-305. In: Ferson S, Burgman M (eds). *Quantitative methods for conservation biology*. Springer-Verlag, New York.

Possingham HP, Wilson KA, Andelman SJ, Vynne CJ. 2006. Protected areas: goals, limitations, and design 509-533. In: Groom MJ, Meefe GK, Carroll CR (eds). *Principles of conservation biology*. 3rd edition. Sinauer Associates, Sunderland, Massachusetts.

Potts WM, Cowley PD, Corroyer B, Næsje TF.2005. Trends in fishery resource utilisation on the Great Fish Estuary. *NINA Report* 50.

Preikshot DB, Nsiku E, Pitcher TJ, Pauly D.1998. An interdisciplinary evaluation of the status and health of African lake fisheries using a rapid appraisal technique. *Journal of Fish Biology* 53:382-393.

Pressey R., Watts ME, Barrett TW, Ridges MJ.2009. The C-Plan Conservation Planning System: Origins, Applications and Possible Futures. In: Moilanen A, Wilson KA, Possingham HP (eds). *Spatial Conservation Prioritisation*. Oxford University Press, New York, USA.

Pressey RL, Logan VS.1998. Size of Selection Units for Future Reserves and its Influence on Actual vs Targeted Representation of Features: A Case Study in Western New South Wales. *Biological Conservation* 85:305-319.

Pressey RL, Cowling RM, Rouget M.2003. Formulating conservation targets for biodiversity pattern and process in the Cape Floristic Region, South Africa. *Biological Conservation* 112: 99-127.

Pressey RL, Tully SL.1994. The cost of ad hoc reservation: a case study in western New South Wales. *Australian Journal of Ecology* 19:375-384.

Pressey RL.1999. Editorial-systematic conservation planning for the real world. *Parks* 9:1-6.

Punt AE.1993.Use of spawner-biomass-per-recruit in the management of linefisheries. In Fish, Fishers and Fisheries.*Proceedings of the Second South African Marine Linefish Symposium*, Durban, October.

Rigney H .1990.Marine reserves – blueprint for protection. *Australian Fisheries* 49:18-22.

Rice JC, Rochet MJ.2005. A framework for selecting a suite of indicators for fisheries management.*ICES Journal of Marine Science*62:516-527.

Roberts C.2007. *The unnatural history of the sea*. Island Press/Shearwater Books, Washington D.C.

Rojas-Nazar U, Gaymer CF, Squeo FA, Garay-Flühmann R, López D.2012. Combining information from benthic community analysis and social studies to establish no-take zones within a multiple uses marine protected area. *Aquatic Conservation of Marine and Freshwater Ecosystems* 22:74-86.

RSA. 2008. Integrated Coastal management Act. No. 24. Government Gazette vol. 524, No. 138, Notice. 31884. Environmental Management Act, Republic of South Africa.

Rudd M.2003. An institutional framework for designing and monitoring ecosystem-based fisheries management policy experiments. *Ecological Economics* 48: 109-124.

SANP (South African National Parks). 2015. Addo Elephant National Parks draft management plan for the period of 2015-2025, under the National Environmental Management Protected Areas Act (Act no 57 of 2008). SANParks, Pretoria.

Sainsbury KJ, Sumalia UR. 2003. Incorporating ecosystem objectives into management of sustainable marine fisheries, including 'best practice' reference points and use of Marine Protected Areas. In Sinclair M, Valdimarsson G (eds). *Responsible Fisheries in the Marine Ecosystem.* CABI Publishing, Oxon, UK.

Sale E.2002. A general model for designing networks of marine reserves. *Science* 298:1991-1993.

Sale PF, Cowen RK, Danilowicz BS, Jones GP, Kritzer JP, Lindeman KC, Planes S, Poulin NVC, Russ GR, Sadovy YJ, Steneck RS.2005. Critical science gaps impede use of no-take fishery reserves. *Trends in Ecology and Evolution* 20: 74-80.

SANBI (South African National Biodiversity Institute).2009. National protected area expansion strategy for South Africa. Department of Environmental Affairs & South African National Biodiversity Institute.

SANP (South African National Parks).2008. 'Policy context: SANParks' mandate and values', in South African National Parks, A framework for developing and implementing management plans for South African National Parks.

Sauer WHH, Hecht T, Britz PJ, Mather D. 2003.An Economic and Sectoral Study of the South African Fishing Industry. *Volume 2: Fishery profiles.*

Schaffer V.2016. Understanding the influence of social capital on social sustainability in an Australian trawl fishery. *International Journal of Sustainable Development* 19: 36-53.

Scott JM, Schipper J. 2006. Gap analysis: a spatial tool for conservation planning. In: Groom MJ, Meffe GK, Ronald C (eds). *Principles of Conservation Biology* (3rd ed.). Sunderland, MA.

Scrogin D, Boyle K, Parsons G, Plantinga AJ. 2004. Effects of regulations on expected catch, expected harvest, and site choice of recreational fishers. *American Journal of Agricultural Economics* 86: 963-974.

Sinclair M, Valdimarsson G (eds).2003. *Responsible fisheries in marine ecosystems*. UK: Biddles ltd.

Sink KJ, Attwood CG, Lombard AT, Grantham H, Leslie R, Samaai T, Kerwath S, Majiedt P, Fairweather T, Hutchings L, van der Lingen C, Atkinson LJ, Wilkinson S, Holness S, Wolf T. 2011. Spatial planning to identify focus areas for offshore biodiversity protection in South Africa. Unpublished Report. Cape Town: South African National Biodiversity Institute.

Sjostedt M, Sundstrom A.2015. Coping with illegal fishing: An institutional account of success and failure in Namibia and South Africa. *Biological Conservation* 180: 78-85.

Smith MK.2005. Towards a new approach for coastal governance with an assessment of the Plettenberg Bay linefisheries. MSc thesis, Rhodes University, South Africa.

Smith MKS, King CM, Sauer WHH, Cowley PD. Development of a fishery indicators for local management initiatives- a case study for Pettenberg bay, South Africa. *African Journal of Marine Science* 29: 511-525.

Smith D. 2008. Movement, growth and stock assessment of the coastal fish Lichia amia (Teleostei: Carangidae) off the South African coast. Masters thesis, University of KwaZulu-Natal, Durban, South Africa.

Solomon JN, Gavin MC, Gore ML. 2015. Detecting and understanding non-compliance with conservation rules. *Biological Conservation* 189:1–4.

Solomon J, Jacobson SK, Wald KD, Gavin M.2007. Estimating illegal resource use at a Ugandan park with the randomised response technique. *Human Dimensions of Wildlife* 12: 75-88.

Solomon J, Michael GC, Gore ML.2015. Detecting and understanding non-compliance with conservation rules. *Biological Conservation* 189: 1-4.

Sowman M.2011. New perspectives in small-scale fisheries management: challenges and prospects for implementation in South Africa. *African Journal of Marine Science* 33: 297-311.

Stewart RR, Noyce T, Possingham HP.2003. Opportunity Cost of Ad Hoc Marine Reserve Design Decisions: An Example from South Australia. *Marine Ecology Progress Series* 253:25-38.

Stewart RR, Possingham HP. 2005. Efficiency, costs and trade-offs in marine reserve system design. *Environmental Modelling and Assessment* 10:203-213.

Stewart RR.2003. Opportunity cost of ad hoc marine reserve design decisions: an example from South Australia. *MarineEcology Progress Series* 253: 25-38.

Strydom HA, King ND.2009. Environmental management in South Africa. 2nd edition. Juta Law. Cape Town.

Sutcliffe PR, Klein CJ, Pitcher CR, Possingham HP.2015. The effectiveness of marine reserve systems constructed using different surrogates of biodiversity. *Conservation Biology* 29: 657–667.

Svancara LK, Brannon R, Scott JM, Groves CR, Noss RF, Pressey RL.2005. Policydriven versus evidence-driven conservation: A review of political targets and biological needs. *Biological Science* 55: 989-995.

Taylor P, Sauer WHH, Penney AJ, Erasmus C, Mann BQ, Brouwer SL, Lamberth J, Stewart TJ.2010. Science An evaluation of attitudes and responses to monitoring and management measures for the South African boat-based linefishery and management measures for the South African boat. *South African Journal of Marine*: 37–41.

Tesfamichael D, Pitcher TJ.2006. Multidisciplinary evaluation of the sustainability of Red Sea fisheries using Rapfish. *Fisheries Research* 78: 227-235.

Tobey J, Volk R.2002. Learning frontiers in the practice of Integrated Coastal Management. *Coastal Management* 30:285-298.

Topping DT, Lowe CG, Caselle JE.2005. Home range and habitat utilisation of adult California sheephead, Semicossyphus pulcher (*Labridae*), in a temperate no-take marine reserve. *Marine Biology* 147:301-311.

Turpie J, Wilson G,Van Niekerk L.2011. National Biodiversity Assessment. *National Estuary Biodiversity Plan for South Africa Technical Report*.

Turpie JK.2004. South African National Spatial Biodiversity Assessment 2004: Technical Report. Volume 3: *Estuary Component. Pretoria*: *South African National Biodiversity Institute.*

Turpie JK, Wilson G, Van Niekerk L.2012. National Biodiversity Assessment. National Estuary Biodiversity Plan for South Africa. Anchor Environmental Consultants Report No AEC2012/01, Cape Town. *Report produced for the Council for Scientific and Industrial Research and the South African National Biodiversity Institute.*

Turpie JK, Goss JR.2014. Potential impacts of alternative regulatory interventions on the recreational value of angling on the Breede River estuary, South Africa. *African Journal of Marine Science*: 36:399-408.

van der Elst RP, Adkin F.1991. Marine linefish – priority species and research objectives in southern Africa. *Oceanographic Research Institute Special Publication.*

van Niekerk L, Turpie JK (eds). 2012. South African National Biodiversity Assessment.2011: Technical Report. Volume 3: Estuary Component. CSIR Report Number CSIR/NRE/ECOS/ER/2011/0045/B. *Council for Scientific and Industrial Research, Stellenbosch.*

Wallace JH.1975.The estuarine fishes of the east coast of South Africa. III. Reproduction. *Investigational Report Oceanographic Research Institute* 41: 1–51.

Watts ME, Ball IR, Stewart RS, Klein CJ, Wilson K, Steinback C, Lourival R, Kircher L, Possingham HP.2009. Marxan with Zones: software for optimal conservation based land- and sea-use zoning. *Environmental Modelling & Software* 24:1513-1521.

Weyl 0. 1999.Fish stock and fisheries of Malawian waters. Resource report for the fisheries department, Malawian Government. *Fisheries Bulletin* No. 39:1-41.

White Paper for Sustainable coastal development in South Africa. 2000. Cape Town: The Department of Environmental Affairs and Tourism

Winker H, Attwood CG, Kerwath S.2015. Assessment of stock abundance of inshore fish resources included in the ‘basket-of-species’ to be allocated under the small-scale fisheries policy. Report of the Linefish Scientific Working Group, LSWG No. 3.Department of Agriculture, Forestry. Cape Town.

Whitfield AK, Cowley PD.2010. The status of fish conservation in South African estuaries. *Journal of Fish Biology* 76: 2067-2089.

Whitfield AK.1997.Fish conservation in South African estuaries. Aquatic Conservation. *Marine and Freshwater Ecosystems* 7: 1-11.

Whitfield AK.1998. Biology and ecology of fishes in southern African estuaries. Ichthyological Monographs of the J.L.B. *Smith Institute of Ichthyology.*

Wintle BA.2008. A review of biodiversity investment prioritisation tools. A report to the Biodiversity Expert Working Group toward the development of the Investment Framework for Environmental Resources.

Wood A, Paterson A, Cowley P.2003. A Classification System for Eastern Cape Estuaries, with Management Guidelines for the Sustainable use of their Living Resources. Eastern Cape Estuaries Management Research Programme Vol 1 & 2. WRC Report No. 1246/1/04.

Wood A, Cowley P, Paterson A.2004. Protocols Contributing to the Management of Estuaries in South Africa, with a Particular Emphasis on the Eastern Cape Province, Volume II, Report C:Classification System for Eastern Cape Estuaries, with Management Guidelines for the Sustainable use of their Living Resources. *Report to the Water Research Commission on behalf of the Institute of Natural Resources.*

Worm B, Hilborn R, Baum JK, Branch TA, Collie JS, Costello C, Fogarty MJ, Fulton EA, Hutchings JA, Jennings S, Jensen O, Lotze HK, Mace PM, McClanahan TR, Minto C, Palumbi SR, Parma AM, Ricard D, Rosenberg AA, Watson R, Zeller D. 2009. Rebuilding Global Fisheries. *Science* 325: 578-585.

Ye, Y, Cochrane K, Bianchi G, Willmann R, Majkowski J, Tandstad M, Carocci F.2013. Rebuilding global fisheries: the World Summit Goal, costs and benefits. *Fish and Fisheries* 14:174-185.

Zhou S, Smith ADM, Punt AE, Richardson AJ, Gibbs MT, Fulton EA, Pascoe S, Bulman CM, Bayliss P, and Sainsbury KJ. 2010. Ecosystem-based fisheries management requires a change to the selective fishing philosophy. *Proceedings of the National Academy of Sciences of the United States of America,* 107: 9485-9.