

**THE CONSERVATION AND MANAGEMENT OF FRESHWATER FISHES IN THE
GREATER ADDO ELEPHANT NATIONAL PARK**

A thesis submitted in fulfilment of the requirements for the degree of

MASTER OF SCIENCE

of

RHODES UNIVERSITY

by

Graham Ronald Louis Traas

January 2009

Table of Contents

Acknowledgements	5
Abstract.....	6
Chapter 1. Conservation status of freshwater fish.....	9
1. Introduction	9
2. A global perspective.....	10
2.1. North America.....	10
2.2. South America.....	12
2.3. Europe	14
2.4. Middle East	17
2.5. Asia	17
2.6. The Antipodes	20
2.7. Africa.....	21
3. Freshwater fish conservation in South Africa.....	22
4. Thesis outline	24
Chapter 2. Study site, sampling procedures and data sources.....	26
2.1. Study area.....	26
Location, climate and vegetation	26
Hydrology	27
2.2. Site selection	36
Mapping	36
Migration corridors	38
2.3. Ichthyofauna of the Sundays River System	38
Indigenous freshwater species	38
Indigenous estuarine species.....	40
Catadromous species.....	41
Alien freshwater species	42
2.3. General sampling procedures	45
Pre-sampling procedure	45

Passive gear.....	46
Active gear	48
Measuring and data collection	49
2.4. Meta-data sources.....	50
Chapter 3. Distribution and relative abundance of freshwater fishes within the Sundays River System	51
3.1. Introduction	51
3.2. Materials and methods	52
3.3. Results	55
Distribution	56
Relative abundance	66
3.4. Discussion	71
Chapter 4. Spoon or scalpel? Species diversity indices as tools for dissecting community assemblage data.....	75
4.1. Introduction	75
4.2. Materials and methods	77
Diversity indices	78
Community assemblage models	80
Model implementation and interpretation.....	83
4.3. Results	84
Species diversity indices	84
Community assemblage models	87
4.4. Discussion	91
Chapter 5. Density of indigenous species in stagnant pools from catch-effort data and depletion models	95
5.1. Introduction	95
5.2. Materials and methods	96
Depletion models	98
Density calculations	101
5.3. Results	102

AIC and Likelihood ratio tests.....	102
5.4. Discussion	104
Chapter 6: Movement of alien fish in the Greater Addo Elephant National Park.....	107
6.1. Introduction	107
6.2. Materials and methods	108
6.3. Results	109
Potential impacts of alien fish.....	114
6.4. Discussion	116
Chapter 7. General discussion	121
7.1. Overview	121
7.2. Conservation goals	122
7.3. Management options	123
River rehabilitation	123
Eradication	124
Control of aliens.....	125
Special cases	126
Education and awareness	127
7.4. Future monitoring and research	128
Monitoring	128
Future research.....	129
Chapter 8. References.....	131
Appendix 1. Raw Data (Dates and GPS co-ordinates).....	145
Appendix 2. Raw data (Water quality data)	148
Appendix 3. Raw data (Substrate & vegetation type)	151
Appendix 4. Raw Data (number of fish sampled per site)	155

Acknowledgements

I would like to thank Tony Booth and Olaf Weyl, my supervisors, for their guidance, assistance and direction. I thank Tony for his help with the analysis of the data, and the assistance he gave me during the write-up phase, both of which were invaluable. I thank Olaf Weyl for his assistance in setting up and conducting the fieldwork, and his help during the write-up phase. Without him, none of this would have been possible.

I would like to thank Bruce Ellender for his assistance in the field, useful discussions on various topics, proof-reading, and the occasional beer.

I would like to thank Olaf Weyl, Tony Booth, Henning Winker, Sam Bodell, Phil Weyl, Richie Peel and the Honours Class of 2008 for their help in the field.

To my mom and dad, I would like to extend my deepest gratitude for supporting me through this stage of my life. Their support and assistance was more than I could ever expect, and I will forever be indebted to them. I would also like to acknowledge my mom and dad for their assistance in the field. To my sister, Lins, thanks for providing a sympathetic ear when I needed it, and for proof-reading some of this thesis. Kirsty, thanks for the encouragement and support, and for making me laugh!

To my DIFS office mates, Pete, Jess, Andrea, Bruce, Orange and Tinto, the office banter kept me sane. Thanks are also due to Henning Winker for useful discussions on various topics covered by this thesis.

To the SANParks staff, John Adendorff, Riaan Nel, Nollie Bosman and Oupa Chauke, I would like to extend my gratitude for their support and interest in the field.

Finally, thanks are due to SANParks and the National Research Foundation (grant # FA2005021600012) for funding this project.

Abstract

Freshwater fish are the most imperilled vertebrate group with a high projected extinction rate. In general, the world's freshwater ichthyofauna is in a poor state and is threatened by habitat destruction, pollution, the effects of alien species, damming, water extraction and overfishing. Within South Africa, legislation supporting the conservation of aquatic habitats and its associated fauna is strong, but there is a lack of capacity and poor co-ordination between agencies responsible for the conservation and management.

The Greater Addo Elephant National Park (GAENP) includes the upper catchments of five of the Sundays River tributaries. Since the optimal method of conserving freshwater aquatic biota is to conserve the headwaters of either the river or its tributaries, there is an opportunity to conserve the native ichthyofauna of the Sundays River system. The principal objective of this thesis was, therefore, to provide information pertinent to the conservation and management of the fishes in the Sundays River system.

Sixteen species of freshwater and estuarine fish were sampled, of which eight were alien. Species composition in each of the rivers, with the exception of the Kabouga River, had changed significantly from previous studies. The distribution ranges of several indigenous species had reduced, while those of some alien species was increased. Two species not previously recorded, *Labeo capensis* and *Gambusia affinis*, were sampled. Alien species were sampled from all the tributaries.

Diversity indices and community assemblage models were identified as approaches for monitoring the ichthyofauna in this species depauperate system. Diversity indices were not recommended for use as a monitoring tool, as they provided insufficient detail on community composition and structure.

A modified depletion estimator was applied to multiple pass electrofishing data to determine the effects of various physico-chemical variables on the absolute abundance of *Pseudobarbus afer*, an endangered species. *Pseudobarbus afer* density was found to be positively correlated to the presence of boulders. Where *P. afer* were present, they were abundant.

Of the total sampled catch, 21.8% was comprised of alien fishes. *Clarias gariepinus* was the most successful invader having penetrated all but one of the tributaries. *Micropterus salmoides* changed the species composition of the Wit River significantly through predation, and is potentially the most dangerous of the alien species in the Sundays River system, capable of extirpating a population of *P. afer* in as little as three days. Only one man-made barrier to the upstream migration of alien fishes was present on the tributaries, and several additional barriers are required to safeguard the populations of indigenous fish in the headwaters of these tributaries.

Several management and conservation tools are presented and discussed, including river rehabilitation, translocation, eradication of alien species, erection of barrier weirs and captive breeding programmes. Public awareness and education is stressed because without support from

the communities surrounding the GAENP, management plans for conservation of the indigenous ichthyofauna will, in all likelihood, fail.

Chapter 1. Conservation status of freshwater fish

1. Introduction

Freshwater fish are the most imperilled vertebrate group with a projected extinction rate of five times that of terrestrial fauna and three times that of marine mammals (Duncan and Lockwood, 2001, Argent et al., 2003, Cooke et al., 2005). In the United States of America alone, over the last century, three genera, 27 species and 13 sub-species have become locally extinct (Argent et al., 2003). Complete extinction is, however, rarely immediate as it is a gradual process, it is difficult to detect and it is usually preceded by local and regional extinctions (Angermeier, 1995). Due to a lack of efficient monitoring programmes, conservation plans are mostly reactive, only being initiated after evidence becomes available that a species has seriously declined in abundance (Angermeier, 1995). At this stage, it may be too late to save the species from complete extinction.

An implicit problem with the conservation of freshwater fishes is that fish evoke little emotional impact (Coad, 1981, Coyle, 1993). Fish are seldom seen, and people have difficulty understanding why commercially and recreationally unimportant fish are worth conserving. The fact that minnow safaris are not as popular as activities such as whale-watching and shark-diving is clear evidence of this. In some parts of the world, this perspective is changing, and the public are beginning to involve themselves in the conservation of freshwater habitats and their biological communities (Nishizawa et al., 2006).

2. A global perspective

The conservation status of freshwater fishes around the world is varied. In general, the prognosis is poor. Vulnerability of freshwater fish to both localised and complete extinction is a function of large body size, high trophic position, poor dispersal and/or colonization ability, restricted geographical range, migratory habits and trophic specialisation (Angermeier, 1995).

Anthropogenic actions that alter either the habitat or species composition of an environment will be especially detrimental to species with trophic or habitat specializations. Species with poor dispersal abilities are easily displaced by generalist species with good dispersal abilities (Maitland, 1985). These generalist species are usually non-native species introduced by man.

Anthropogenic activities, include, but are not limited to, habitat destruction and alteration, overfishing and the introduction of alien fish species and alien diseases and parasites (Allendorf, 1988). Habitat destruction and alteration can include pollution, construction of dams and weirs, poor farming practices, channelisation of water beds, deforestation and inter-basin water transfer schemes (IBWT). IBWT schemes have the potential to introduce not only alien fish, invertebrates, parasites and other diseases, but changes to both water quality and quantity.

2.1. North America

The extent to which the population of the United States utilises its water resources is such that less than 2% of the USA's rivers are currently considered pristine (Ricciardi and Rasmussen, 1999). This extensive habitat alteration has led to an estimated freshwater fish extinction rate of 4% per decade (Ricciardi and Rasmussen, 1999), despite the promulgation of several federal laws

and recovery programs (Moyle, 1995, Brower et al., 2001). A lack of staff, and funding has been identified as a weak link in conservation efforts (Moyle, 1995), together with the inability of field workers to identify species, a lack of available biological and ecological information and an indifferent public (Cooke et al., 2005).

The USA's history of conserving its aquatic biota is poor. In California, during the late 1800s, the largest freshwater lake in the state was drained to provide land for agriculture (Moyle, 1995), and as recently as 1962, a seven mile stretch of water on the Green River above the Flaming Gorge Dam in Wyoming was rotenoned to remove indigenous species so that a sport fishery could be established (Vanicek, 1970).

The introduction of alien species started between 1850 and 1900 (Moyle, 1995). These species have subsequently become established. Of the 25 species collected in the Cosumnes River in California, 17 were alien to the river system (Moyle et al., 2003), and in the Solomon River in Kansas, 51% of the fish community was comprised of alien species (Eberle et al., 2002).

Even though there are recovery programs for the restoration and protection of some waterways in the USA, these are not always effective. The Colorado River Recovery Implementation Program (CRRIP) was established in 1987 to conserve the native ichthyofauna of the Colorado River while still allowing development and use of the water resources provided by the river (Brower et al., 2001). The stakeholders in this program included the US Fish & Wildlife Service, US Bureau of Reclamation, the States of Colorado, Wyoming and Utah, water users, and environmental

groups. In 14 years, the CRRIP endorsed 200 water development projects, and the populations of native fishes continued to decline, mainly due to political motivations of the stakeholders, despite stakeholders having a voice in the decision-making process (Brower et al., 2001).

In Mexico, there are 200 species of freshwater fish, 120 of which are under threat, and 15 are already locally extinct through anthropogenic actions (Contreras-B and Lozano-V, 1994).

Pollution of the various waterways is high, and fish kills have been reported. There is legal protection for freshwater fish in the Mexican Act, which formed the National Committee on Knowledge and Use of Biodiversity, but this legislation has not been particularly useful, as the populations of native species are declining due to a rapidly expanding human population and associated aquatic pollution (Contreras-B and Lozano-V, 1994).

2.2. South America

Generally, in South America, the conservation of freshwater fishes is poor, mainly due to non-compliance by industry, an indifferent public, lack of technical and scientific knowledge and inadequate protected areas (Lopez-Rojas and Bonilla-Rivero, 2000, Agostinho et al., 2005, Laurance, 2005, Habit et al., 2006).

Around Lake Valencia, Venezuela, rapid industrial and urban growth has led to a 60% decrease in fish diversity due to the degradation of the water supplying the lake (Lopez-Rojas and Bonilla-Rivero, 2000). This region has been identified as one of the three most critical areas in terms of the numbers of endangered animals inhabiting it. Despite its status, little attention has been paid

to the problems affecting the decline of freshwater fish in Lake Valencia and its associated rivers. The contamination of the lake includes agricultural run-off with pesticides and fertilizers, urban and industrial run-off, water abstraction, and siltation from construction projects (Lopez-Rojas and Bonilla-Rivero, 2000).

In Chile, the Biobío River is one of the most important rivers for human utilisation (Habit et al., 2006). The river is used for the provision of drinking water, irrigation, sewage disposal, hydropower generation and industrial consumption. Over 80% of the native fish species inhabiting this river are listed as vulnerable or endangered and their distributions are largely declining. The recent introduction of legislation requiring the improvement of the water quality of both industrial and sewage effluent has had an immediate positive effect on water quality, but due to the presence of introduced species, is not likely to reverse the decline of native species (Habit et al., 2006).

In Brazil, 134 freshwater fish species are listed as threatened (Agostinho et al., 2005). The causes of these declines include pollution and eutrophication, siltation, impoundments and flood control, over-exploited fisheries and species introductions. The majority of protected areas in Brazil were created to protect terrestrial fauna and flora, but incidentally protect important rivers and wetlands. The management of these reserves is generally opportunistic (Laurance, 2005), and is based on poor technical and scientific knowledge, and conservation goals are generally not met due to a lack of information on the fish populations, insufficient funds and limited enforcement (Agostinho et al., 2005).

2.3. Europe

Of the 72 freshwater fish species occurring naturally in France, 27 are IUCN Red-Listed (Keith, 2000). The major cause of the decline of freshwater fish in France is habitat destruction (Changeux and Pont, 1995). Although there are a number of protected areas in France, many of these areas were established for mammal, bird or plant conservation, and any fish conserved in the area were done so incidentally (Keith, 2000). Indeed, authorities are uncertain of the ichthyological species composition in many of the nature reserves. Only one of the Red-Listed species occurs within the core areas of six French national parks, and of the 122 nature reserves, only 51 have fish within their boundaries. Only 16 of the 27 threatened species are found within the boundaries of these nature reserves (Keith, 2000). One of the major problems facing the conservation of freshwater fish in France is the lack of scientific interest in the subject. Keith (2000) stated that there was no research being conducted on freshwater fish in French protected areas, and that the management of freshwater fish conservation was therefore not progressing in a suitable manner.

There are 27 endemic freshwater fish species in Spain, of which 17 are Red-Listed (Elvira, 1995). Spanish fishes are threatened by a variety of factors including construction of dams, pollution, overfishing, habitat destruction, predation by alien fish, water abstraction, hybridization with alien fish, use as bait for angling and competition by alien fish. National conservation programmes are non-existent and the conservation of freshwater fish is not taken seriously by either the government or the public (Elvira, 1995).

Of the 39 freshwater fish species found in Portugal, 11 are alien and 16 are endangered (Almaca, 1995). The high level of endemism in the country has not prevented industry from polluting freshwater rivers heavily, frequently causing mass fish mortalities. Sand extraction and poorly constructed fishways are significant conservation threats. The most serious threat to endemic freshwater fish is the impact of introduced alien fishes, such as largemouth bass, *Micropterus salmoides*, and bluegill sunfish, *Lepomis macrochirus* (Almaca, 1995). No national programmes for the conservation of endemic freshwater fishes exist in Portugal.

In the British Isles, 54 freshwater species are known to occur, of which 12 are introduced (Maitland and Lyle, 1991). A trend in this region, as with many other regions in the world, is the position that fish are not worthy of conservation attention, either because they have no economic value, or because a 'cash value' cannot be placed on them (Maitland and Lyle, 1991). Perhaps the most important threat is high recreational fishing pressure (3 million freshwater anglers) and the introduction of freshwater sport-fish such as rainbow trout, *Oncorhynchus mykiss* and largemouth bass, *M. salmoides* (Maitland, 2004). Pollution is not considered to be a major threat to true freshwater fish, since most pollution occurs in estuaries but this trend has serious implications for fish that require estuaries to complete their life-cycles (Maitland and Lyle, 1991). The construction of dams and weirs are causing losses in species diversity due to the loss of upstream habitats and the barriers to migration that these structures present. Water abstraction from ponds, small streams and ox-bow lakes is removing the refugia that many of the smaller indigenous species inhabited. Despite strong legislation, and numerous reviews of the declining status of the native fishes of Ireland, little action has been taken to protect these species (Maitland, 2004).

The freshwater habitats in Albania are threatened by industrial pollution, eutrophication, habitat alteration and the effects of alien species (Rakaj and Flloko, 1995). The extent of the decline of the populations of native species is unknown due to limited research effort.

Of the 79 fish species native to Greece, 22 are threatened. The main reasons for the decline of these species are agricultural and industrial pollution, overfishing, partial or total destruction of habitat, water abstraction, dams and alien species. Despite these threats, few of the native fish species are locally endangered, and most are protected by legislation. This legislation, however, is ineffective, because practical measures for its implementation are lacking (Economidis, 1995).

In Italy, only 45 of the 71 species present are native to Italian freshwater. Freshwater conservation is not a high public or scientific priority. Indeed, most conservationists are either unaware or unconcerned about the declines of native fish populations, and the introductions of alien species are carried out legally and are publicly lauded (Bianco, 1995).

In Turkey, industrial and urban pollution are the most serious factors affecting freshwater fishes, however there is insufficient knowledge of the biology and ecology of the native species to infer the status of these species (Balik, 1995).

2.4. Middle East

Few reviews of freshwater fish conservation have been conducted in the countries of the Middle East region. Coad (1981) reviewed the conservation status of freshwater fishes in Iran. Conservation strategies in this country seem to be non-existent and most of the conservation measures for Iranian catchments that have been introduced are on the Russian side of the borders where the two countries share a catchment. There is some control of the freshwater fisheries but, as with other parts of the world, regulation of fishing legislation is poor at best. Some hatchery experiments have been conducted to improve the stock of commercially important species, but non-commercial and recreationally unimportant species remain largely ignored (Coad, 1981). A major problem with the conservation of freshwater fishes of Iran is the lack of scientific knowledge. The taxonomy of these fishes is poorly understood, and besides the fishes of the Caspian Sea, an identification key for Iranian fishes has yet to be devised. Eutrophication, damming, water abstraction, poaching of migrating fish species, pollution and the effects of alien species are all considered responsible for the decline of freshwater fishes in this region (Goren and Ortal, 1999). In Israel, although there are 200 protected areas, no freshwater species have been formally protected by legislation (Goren and Ortal, 1999).

2.5. Asia

Many of the rivers in Asia are grossly polluted and the drainage basins of these rivers have been extensively deforested or altered (Dudgeon, 2005). Due to a burgeoning impoverished human population, governments have been forced to focus on economic rather than conservation development measures. The result has been an expansion in the construction of irrigation systems

and IBWT, damming and the unchecked overexploitation of fish stocks. There is also a general lack of public awareness of the plight of Asia's freshwater systems (Dudgeon, 2000).

Pandit (1991) conducted a study on the wetland flora and fauna of the Kashmir province in India and found that these delicate ecosystems are being heavily affected by anthropogenic actions. Of particular concern is the draining of wetlands for urban water use and reclamation of land for agricultural purposes. The latter is endorsed by the Indian government. Other impacts include channelisation of main rivers leading into the wetlands, overgrazing of land near wetlands, pollution, siltation and poaching of edible fish species. No mention was made of any steps being taken to conserve these wetlands by government (Pandit, 1991).

Of the 15 endemic freshwater fish species found in Sri Lanka, two are rare, seven are vulnerable and two are endangered. Therefore, 73% of Sri Lanka's freshwater fish species are threatened with extinction (Wikramanayake, 1990). The main threats to endemic species are mining, deforestation, agriculture, damming and urbanisation. Conservation efforts in Sri Lanka have been focussed on captive breeding of commercially important species, regulation of fisheries, better watershed management, and, perhaps controversially, translocation of sensitive species to establish refuge populations (Wikramanayake, 1990).

In China, sturgeons, large catfishes and cyprinids are threatened by overfishing and damming, which has limited their ability to access suitable spawning and feeding grounds (Wei et al., 1997). Damming and the associated altered hydrological regime is the most significant threat to

its freshwater fishes (Fu et al., 2003). Several captive breeding programmes have been initiated, but these were with commercial rather than conservational intent. Most of the captive breeding programmes have, however, failed. Although the sturgeons are protected by legislature (Dudgeon, 2005), the conservation of these fishes has failed due to insufficient management (Wei et al., 1997).

In Bangladesh, flood control, damming and irrigation schemes are limiting the amount of water that flows into the floodplains, thus reducing the amount of spawning ground available for potamodromous species (Craig et al., 2004). These control and abstraction practises may result in the loss of approximately 151 300 tonnes of fish per annum. The use of a co-management approach has failed due to a lack of a sense of ownership, resulting in overfishing and reckless use of water resources (Craig et al., 2004).

In Laos, the increasing pressures of increasing human population, and associated pollution, over-exploitation, and habitat disturbance has led to the implementation of large numbers of “Fish Conservation Zones” and no-take fish sanctuaries. These reserves have had a beneficial effect for populations of both sedentary and migratory species along the portions of the Mekong River that flow through Laos (Baird and Flaherty, 2005).

Jang *et al.* (2003) report that the conservation of freshwater fish has been an area of concern for both scientists and the public in South Korea since the 1980s. Waterways in South Korea are affected mainly by agriculture and the associated water abstraction, but also by damming, mining

and construction. Weirs, while interrupting the continuity of rivers and hence migration routes, are important refugia for fishes during the dry season. Endemic species are well-represented in the national parks of South Korea, especially in headwater sections of catchments which are well-protected. Although there are a number of exotic species in the rivers of South Korea, little information is available on the effects these species are having on endemic species. A problem common throughout South Korea is the use of throw- and cast-nets on the borders of the national parks. The potential for fish introductions to these areas is high and may have serious implications for endemic species if these introduced species infiltrate the national parks (Jang et al., 2003). On the whole, South Korea's freshwater fish species seem to benefit the best protection of all the countries within Asia.

2.6. The Antipodes

In New Zealand, the threats to native freshwater are pollution from agriculture, habitat alteration from alluvial mining, forestry, pollution and alien species (Townsend and Crowl, 1991, Townsend and Winterbourn, 1992, Jowett et al., 1996, Townsend, 1996, Ling, 2004, Neilson et al., 2004). The effects of alien fish species, especially rainbow trout and brown trout, *Salmo trutta*, threaten several species of native galaxiids (Townsend and Crowl, 1991, Townsend and Winterbourn, 1992, Townsend, 1996). Mosquitofish, *Gambusia affinis*, have had deleterious effects on the populations of native fish (Ling, 2004). The eradication of these alien species has been attempted through the use of the piscicide rotenone and intensive gill netting using monofilament gill nets (Neilson et al., 2004).

In Australia, 192 freshwater fish species have been recorded, with 65 of these considered vulnerable (Maitland, 1995). The major threats to these native species include hydroelectric and irrigation schemes, channelization, overfishing, pollution and exotic species (Lloyd and Walker, 1986, Morgan et al., 2004). Some of Australia's freshwater fishes are protected by legislation, but conservation efforts are hampered by lack of scientific knowledge, indifference from the public, inconsistency in policy and the lack of a co-ordinated national management plan (Koehn and MacKenzie, 2004). Despite this, captive breeding programmes have been implemented for five threatened species (Maitland, 1995).

2.7. Africa

Native ichthyofauna in African countries are exposed to a wide range of threats including exotic species, overfishing (sometimes with the use of chemicals), deforestation, siltation and eutrophication, agricultural, urban and industrial pollution, policy deficiencies, inadequate conservation planning, conversion of sites for agriculture & aquaculture, water diversion and abstraction and IBWT schemes (Shumway, 1999). The conservation of freshwater ichthyofauna in Africa has been largely ignored, with the exception of Madagascar and South Africa (Skelton, 1983, Skelton, 1990, Reinthal and Stiassny, 1991, Maitland, 1995, Skelton, 2002, Benstead et al., 2003).

In Madagascar, the decline of the native fauna has been attributed to the presence of alien species, siltation from extensive forest clearing and the associated erosion, pollution and water abstraction from agriculture (Reinthal and Stiassny, 1991, Benstead et al., 2003). In some areas,

overfishing by subsistence fishermen remains a significant problem (Reinthal and Stiassny, 1991). Although the Malagasy government has expanded the network of formal protected areas (Benstead et al., 2003), these conservation efforts are focused on terrestrial fauna and flora. Aquatic habitats remain largely ignored (Reinthal and Stiassny, 1991).

In central Africa, overfishing by destructive methods such as piscicides and even dynamite is common (Udoidiong, 1988, Castelo, 1994, Inogwabini, 2005). While some governments have taken action and banned certain fishing methods, the lack of law enforcement is hampering efforts by conservationists to curb the decline of the native ichthyofauna within the region (Castelo, 1994, Inogwabini, 2005).

In Namibia, aquatic habitats are affected by water abstraction, river flow regulation, pollution, siltation and over-exploitation (Curtis et al., 1998). Four wetland sites have been designated as Ramsar sites, but no aquatic fauna is formally protected (Curtis et al., 1998). As with many other regions in Africa, there is a lack of law enforcement of and public interest in aquatic conservation (Curtis et al., 1998).

3. Freshwater fish conservation in South Africa

There are less than 100 freshwater fish species in South Africa, of which 37 are threatened to some extent (Skelton et al., 1995). These fishes are threatened by channelization of stream beds, stream diversion and water extraction, especially in the Cape Fold Mountains, catchment changes from agriculture, overgrazing, forestry and rural human settlement, erosion and sedimentation,

construction of dams and weirs, IBWT schemes, which changes both quantity and quality of water, and the biota of the receiving system, pollution and alien species (Skelton, 1990). While aquatic systems are given high conservation status via legislation, such as the Environment Conservation Act No. 73 of 1989, the National Environmental Management Act No. 107 of 1998, the National Water Act No. 36 of 1998 and National Environmental Management: Biodiversity Act No. 10 of 2004, there is a lack of capacity and poor co-ordination between the agencies responsible for conservation and water management. The lack of awareness of the importance of biodiversity for the economy also hampers conservation efforts (Cowling et al., 2003).

Although South Africa has many protected areas, few are ideal for fish conservation because they do not protect entire catchments (Skelton, 1990). A further problem is that conservation managers have little or no control over the quality of water flowing through their protected areas (Skelton et al., 1995). Those mountain catchment reserves that are able to protect native ichthyofauna from pollution and habitat destruction are often plagued with the problem posed by alien species (Skelton et al., 1995). Despite the issues of conservation within protected areas, water-resource managers are recognising that conservation managers are important stakeholders and must be considered when changes are to be made to a watercourse (Skelton, 1990).

In South Africa, the conservation status of some river systems still remains unknown as there is a severe paucity of biological information and locality records from these systems. The Sundays River system is one such river system (Roux et al., 2002). The Sundays River is affected by an IBWT scheme that transfers water from the Orange River to the Great Fish River and then into the Sundays River. The Sundays River, which is highly altered due to this IBWT scheme, flows

through a formally protected area, the Greater Addo Elephant National Park (GAENP). This same protected area contains the headwaters of five of the Sundays Rivers' tributaries. Since the optimal method of conserving freshwater aquatic biota is to conserve the headwaters of either the river or its tributaries (Angermeier and Winston, 1999, Saunders et al., 2002, Abell et al., 2007), there is potential for the conservation of the native ichthyofauna of the Sundays River system within this national park.

4. Thesis outline

The objective of this thesis is to contribute to a conservation management plan for the freshwater fishes of the Greater Addo Elephant National Park (GAENP). This was achieved by examining the current distribution and relative abundance of fishes in the GAENP, examining potential monitoring tools, assessing the impacts of alien species and reviewing some of the tools that have previously been used to assess community composition.

This thesis has been structured into seven chapters. Chapter 2 describes the study site, each of the rivers examined, pre-sampling procedure, materials and general methods and sampling site selection. Chapter 3 describes the current relative abundance and distribution of the freshwater fishes in the Sundays River system, and compares these results to previous studies. Chapter 4 reviews two tools for analysing community assemblage data, species diversity indices and community assemblage models, and attempts to determine the most effective tool for analysing community data in ichthyologically depauperate environments. Chapter 5 examines the densities of the smaller indigenous fish species in the tributaries in an attempt to define population sizes

for each of the streams. Chapter 6 describes the movement of alien fish species in the Sundays River system, focusing on the Wit River, where *M. salmoides* are present. In Chapter 7, various management options for the conservation of freshwater ichthyofauna are discussed.

Chapter 2. Study site, sampling procedures and data sources

2.1. Study area

Location, climate and vegetation

The Greater Addo Elephant National Park (GAENP) is situated in the Eastern Cape, South Africa (Figure 2.1). The park is large, incorporating 6 860 km², a third of which is marine. The GAENP extends from the semi-arid plains around Darlington Dam in the north-west, extending south east over the Zuurberg mountains to its southeastern boundary in Algoa Bay (SANParks, 2006).

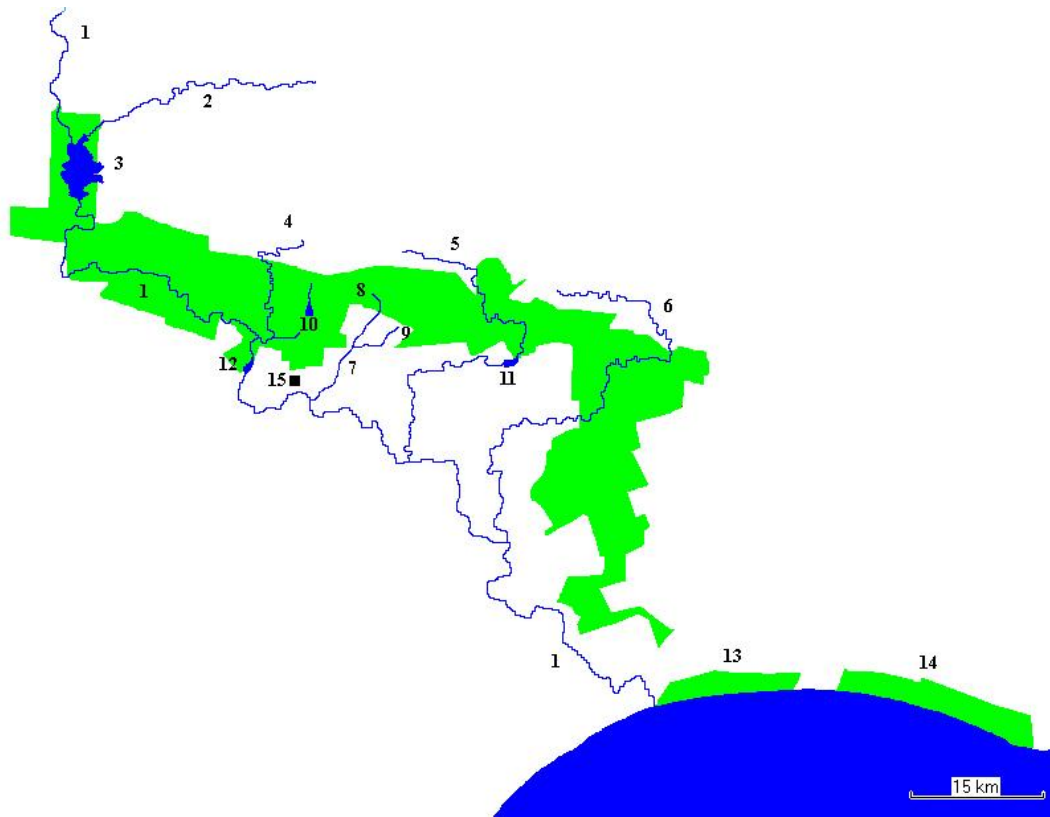


Figure 2.1. The Greater Addo Elephant National Park, with marine section, showing the Sundays River, and tributaries. 1 = Sundays River; 2 = Skoenmakers River; 3 = Darlington Dam; 4 = Kabouga River; 5 = Wit River; 6 = Krom River; 7 = Uie River; 8 = Klein Uie River; 9 = Groot Uie River; 10 = Woodlands Dam; 11 = Slagboom Dam; 12 = Korhaansdrift Dam; 13 = GAENP, Colchester section; 14 = GAENP, Woody Cape section; 15 = Kirkwood.

The GAENP is situated in a warm temperate climate belt that receives season-dependent rainfall (SANParks, 2006). The summer rainfall area is predominantly in the northern Zuurberg mountain range and year-round rainfall occurs in the southern belt of the Zuurberg Mountain range (SANParks, 2006). Mean annual rainfall varies greatly within the Park, ranging from 50 mm per annum in the Darlington section to 550 mm in the Nyathi section (Roux et al., 2002). Air temperatures range between 15° C and 45° C in summer, and between 5° C and 18° C in winter (Lombard et al., 2001).

Due to differences in altitude, climate, topology, geology and soil type, the vegetation within the GAENP is diverse with five of the seven biomes found within South Africa (Roux et al., 2002). These include the Nama Karoo, Fynbos, Forest, Thicket and Grassland biomes (Lombard et al., 2001). North of Darlington Dam, succulent Noorsveld type, karoid vegetation and Spekboom Succulent Thicket is found (Roux et al., 2002). Southeast of Darlington Dam, and north of the Zuurberg Mountain range, Karoo plains are prominent. Grassy and Mountain Fynbos dominate on the relatively nutrient-poor northern slopes of the Zuurberg Mountain range (Roux et al., 2002). On the southern slopes of the Zuurberg Mountain range, Xeric and Mesic Succulent Thicket dominate (SANParks, 2006).

Hydrology

The GAENP contains two main rivers, the Sundays and Skoenmakers, and five tributary rivers, the Kabouga, Klein Uie, Groot Uie, Wit and Krom, within its boundaries. There are four large impoundments. Darlington Dam (previously named Lake Mentz) and Korhaansdrift Dam are

located on the Sundays River, Woodlands Dam is located on the Kabouga River and Slagboom Dam is located on the Wit River (Fig 2.1).

Sundays River

The Sundays River (33° 43.312' S, 25° 51.152' E: estuary mouth co-ordinates) is the only major river within the boundaries of the GAENP, with 44 km of its reach lying within the Park's boundaries. The Sundays River has a catchment size of approximately 20 792 km² (Russell, 1999). Upstream of Darlington Dam, the Sundays River is seasonal, flowing only during the rainy season. Below Darlington Dam, the Sundays River is perennial (Roux et al., 2002). There are two impoundments on the Sundays River, namely Darlington Dam and Korhaansdrift Dam. Darlington Dam has a 4350 ha area at full capacity, but is managed at 30% of full supply level by the Sundays River Irrigation Board, while Korhaansdrift Dam is approximately 7.6 ha. Darlington Dam was initially constructed in 1922 as a water storage facility for the intensive citrus agriculture in the lower Sundays River valley. This impoundment is supplied by the Skoenmakers River, which is augmented by water transfer from the Great Fish River, which is, in turn, augmented by water transfer from Lake Gariep on the Orange River system (Cambray and Jubb, 1977).

Water releases from Darlington Dam are controlled by the Sundays River Irrigation Board and the Department of Water Affairs and Forestry to avoid overflow conditions. Water released from Darlington Dam flows into Korhaansdrift Dam, 45 km downstream. Water from Korhaansdrift Dam is extracted into a canal system which is used for irrigation of crops in the Sundays River

valley. These releases are carefully controlled such that there is little overflow (Russell, 1999). Water from the canal systems flows back into the Sundays River via outflow pipes approximately 70 km downstream from Darlington Dam and groundwater return flows from irrigation. Between Korhaansdrift Dam and the canal outflow pipes, there is no flow in the Sundays River unless the Kabouga River floods, although there is standing water in this section.

The Sundays River is warm, the water slightly alkaline (pH 7.06 – 8.75), and has high water conductivity (553 μ S – 765 μ S) and dissolved oxygen concentrations (Russell, 1999). Russell (1999) mentions that the concentration of the ionic salts sodium, sulphate, bicarbonate and chloride are high in the lower Sundays River, reflecting the adverse effects that intensive agriculture is having on water quality. Aquatic macrophytes are dense, with only the main channel, and deep sections of river being open. Aquatic macrophytes are dominated by tall (± 3 m tall) *Phragmites* reeds. Riverine substrate is dominated by boulders (rocks > 30cm diameter), with small patches of muddy or silted areas. The banks of the river are generally steep, with overhanging trees and assorted shrubs (Figure 2.2).



Figure 2.2. Habitat types on the Sundays River. On the left, a slow-flowing pool with dense *Phragmites* beds and on the right, a fast-flowing stretch of river with dense macrophyte growth on the banks.

Kabouga River

The Kabouga River (33° 18.351 S, 25° 22.765 E) is a small tributary of the Sundays River, about 28 km in length, of which 15.1 km lies within GAENP. It enters the Sundays River just above Korhaansdrift Dam. The river is episodic, only flowing after heavy rains (Roux et al., 2002) for approximately ten days (N. Bosman, SANParks Section Ranger, Kabouga, *pers. comm.*, 2007). During this survey, the only water that was present within the GAENP was in a deep depression, on the shaded side of a cliff face.

Water temperature ranged between 15° C and 17.6° C (Table 2.1), slightly lower than that noted by Russell (1999) at between 17.7° C and 21.3° C. The water in the Kabouga River was slightly acidic (pH: 6.92 – 7.45), in contrast to the alkaline conditions (pH: 8.3 – 8.6) noted by Russell (1999). The substrate is dominated by boulders, with some stones (rocks between 5 cm and 30cm

in diameter) and small gravel beds (pebbles < 5 cm in diameter) (Figure 2.3). The small pool that was present during the dry season was turbid. Canopy cover along the course of the Kabouga River is sparse. The river banks are steep with overhanging trees and shrubs. The land adjacent to the river channel is natural with no evidence of anthropogenic actions. Roux *et al.* (2002) described the Kabouga River as being in a pristine condition.



Figure 2.3. Habitat types along the Kabouga River. On the left, boulders, stones and gravel substrate, and on the right, the only pool found along the Kabouga River in the sampled area that was turbid with a muddy substrate.

Uie River and its tributaries

The Uie River (33° 22.747' S, 25° 27.545' E) is 7.8 km in length and flows into the Sundays canal system 2 km east of Kirkwood. No part of this river lies within the Park's boundaries. The Uie River splits into the Klein Uie and Groot Uie Rivers. The Klein Uie River (33° 19.814' S, 25° 29.495' E) is 15.6 km in length, of which 8.4 km lies within the Park's boundaries, and the Groot Uie (33° 20.972 S, 25° 32.148 E) is 6.6 km in length, and does not flow through the Park,

although SANParks has identified this area for future inclusion in the GAENP. The three rivers are episodic, flowing only after heavy rainfall (Roux et al., 2002). In the dry season, pools form ranging from 10 cm to over 2 m in depth. The majority of pools are *ca* 0.5 m deep. The Uie River dries up completely, except for a small pool inside a weir. The Uie River and its tributaries have the smallest of the catchments of the rivers flowing through the Park.

The water in the Uie River was cold and acidic with low conductivity and total dissolved solid concentrations (Table 2.1). The substrate is dominated by boulders, with some stones and bedrock (continuous sheets of rock). Macrophyte growth along the riverbeds is minimal, but where it occurs, is dominated by small reedbeds. Canopy cover is thick on the Klein Uie and Groot Uie rivers (Figure 2.4), but almost absent on the Uie River. Adjacent land use along the Uie River and its tributaries is dominated by game ranching activities. The lower section of the Uie River flows past a small settlement on the outskirts of Kirkwood (Friedagevonden, 33° 23.238 S, 25° 26.993 E).



Figure 2.4. Habitat types along the Groot Uie River showing dense canopy cover (left) and submerged macrophyte growth on large boulders (right).

Wit River

The Wit River (33° 20.903' S, 25° 41.359' E) is seasonal, and about 60 km in length flowing into the Sundays River 20 km southeast of Kirkwood, 16.3 km of its length lies within the GAENP.

The Wit River has the second largest catchment of the Sundays River tributaries that flow through the Park. After summer rainfall, the Wit River flows for up to six months (O. Chauke, SANParks Section Ranger, Zuurberg, pers. comm., 2008). In the dry season, the river ceases flowing, and forms numerous pools, with some that are over three metres deep. A single impoundment, the 10 ha Slagboom Dam, is located towards the lower section of this river. The Wit River is dry for most of its length between Slagboom Dam and its confluence with the Sundays River, with only a few pools present. This is mostly due to water abstraction from Slagboom Dam itself, and from a deeper pool 150 m downstream from the dam wall.

During the study period, the Wit River water had a temperature range of 11.6° C to 29.2° C and a pH of between 6.04 and 8.07. Conductivity was found to range between 257 and 686 µS. Total

dissolved solids (TDS) ranged between $128 \text{ mg}\cdot\text{l}^{-1}$ and $341 \text{ mg}\cdot\text{l}^{-1}$ (Table 2.1). Aquatic macrophytes are rare in the Wit River, and where they do occur, emergent macrophytes in the form of reeds dominate. In the upper reaches, algae covers the entire substrate. Stones dominate the substrate type, with some gravel, boulders and bedrock (Figure 2.5). Muddy pools are found on the Wit River, but are rare. Canopy cover is limited along most of the Wit River, but it is thick above a small tributary that runs into the Wit River from the east. Adjacent land use in the upper reaches is mostly extensive cattle ranching, and in the middle reaches near Slagboom Dam, there is a small citrus plantation. In the lower reaches, adjacent land use is intensive citrus agriculture, and there is a small settlement, Enon, situated on the banks of the Wit River. Roux *et al.* (2002) described the Wit River as being moderately transformed.



Figure 2.5. Habitat types along the Wit River, exhibiting a substrate of stones and boulders.

Krom River

The Krom River ($33^{\circ} 24.278 \text{ S}$, $25^{\circ} 47.017 \text{ E}$) is 77.5 km long, with 26.6 km of its length lying within the Park's boundaries. It flows into the Sundays River 23 km southeast of Kirkwood. It is

seasonal, flowing for up to eight months after the rainy season (O. Chauke, SANParks Section Ranger, Zuurberg, pers. comm., 2008). The Krom River has the largest catchment of the rivers flowing through the GAENP.

Water temperatures in the Krom River ranged between 15.9° C and 27.6° C, pH between 6.65 and 8.38, conductivity between 680 μS and the upper limit of the instrument, which was 3999 μS , and total dissolved solid concentrations between 347 $\text{mg}\cdot\text{l}^{-1}$ and 2000 $\text{mg}\cdot\text{l}^{-1}$ (Table 2.1). In the middle reaches of the Krom River, the substrate is predominantly muddy with some stone patches, aquatic macrophytes are rare, and are present only in a single large pool, in the form of lily pads and canopy cover is sparse (Figure 2.6). In the upper and middle reaches of the Krom River, adjacent land use is reserved for conservation, but along the lower reaches, the Krom River passes through intensive citrus agriculture land. Roux *et al.* (2002) described the Krom River as moderately transformed by anthropogenic actions.



Figure 2.6. Habitat types along the Krom River, showing a clear pool (left) and an abundance of water lilies (circled) in the muddy Gwarrie Pan (right).

Table 2.1. Water quality data (water temperature, pH, conductivity, dissolved oxygen, percentage oxygen saturation and total dissolved solids) from various rivers and their associated impoundments measured between September 2007 and April 2008. Values presented in parentheses refer to values recorded by Russell (1999). Methods used to obtain these parameters are detailed in section 2.3.

	Sampling dates	Temperature (°C)	pH	Conductivity (µS)	DO (mg·l ⁻¹)	% Saturation	TDS (ppm)
Darlington	Nov 07; Feb 08	20.1 - 25.6 (-)	8.7 - 9.27 (-)	560 - 720 (-)	9.2 - 12.9 (-)	106.2 - 158.8 (-)	285 - 364 (-)
Groot Uie	Sep 07	13.4 - 18.5 (-)	6.9 - 7.13 (-)	225 - 292 (-)	3.2 - 8.4 (-)	32.5 - 91.6 (-)	113 - 153 (-)
Kabouga	Sep 07	15 - 17.6 (17.7 - 21.3)	6.92 - 7.45 (8.3 - 8.6)	533 - 1089 (990 - 1000)	2.7 - 11 (-)	27.7 - 122 (-)	266 - 545 (395 - 611)
Klein Uie	Sep 07	13.7 - 22.2 (14.5 - 16.2)	6.37 - 7.4 (6.0 - 6.2)	249 - 600 (281 - 297)	5.4 - 10.1 (-)	53.9 - 107.8 (-)	124 - 300 (171 - 185)
Krom	Nov 07	15.9 - 27.6 (14.8 - 16.0)	6.65 - 8.38 (7.1 - 7.8)	680 - 3999 (660 - 720)	1.8 - 16.2 (-)	18.7 - 207 (-)	347 - 2000 (508 - 541)
Sundays	Sept 07; Nov 07; Mar 08	19 - 26.7 (17.1)	7.06 - 8.75 (8.2)	553 - 765 (890)	9.6 - 12.9 (-)	105 - 164.3 (-)	289 - 390 (593)
Wit	Sep 07; Feb 08; Mar 08	11.6 - 29.2 (15.1 - 18.1)	6.04 - 8.07 (6.1 - 7.0)	257 - 686 (262 - 650)	4.2 - 11.6 (-)	44.1 - 141.2 (-)	128 - 341 (142 - 497)

2.2. Site selection

Mapping

Sampling site selection was based on mapping, previously sampled sites and field assessments. Previously sampled sites used by Russell (1999) and Swartz (South African Institute of Aquatic Biodiversity, unpublished data) were plotted on a “South Africa – StreetMaps v3.06” map in Garmin© MapSource© and intersections of roads and rivers noted. While in the field, these sites

were assessed in terms of water availability and vehicle and hiking accessibility. Sampling sites that were either dry, or had no access were eliminated from the list of available sites. In total, 73 sampling sites were selected and sampled once each between September 2007 and March 2008. Seven sites were selected on the Skoenmakers River, in Darlington Dam five sites were selected from different habitat types and three sites were selected along the Sundays River. Three suitable sites along the Kabouga River were identified and sampled. Six sites were sampled along the Groot Uie River, and 12 sites were sampled along the Klein Uie River. The only available water body on the Uie River was sampled. A total of 21 sites on the Wit River and seven sites along the Krom River were identified and sampled (Figure 2.7).

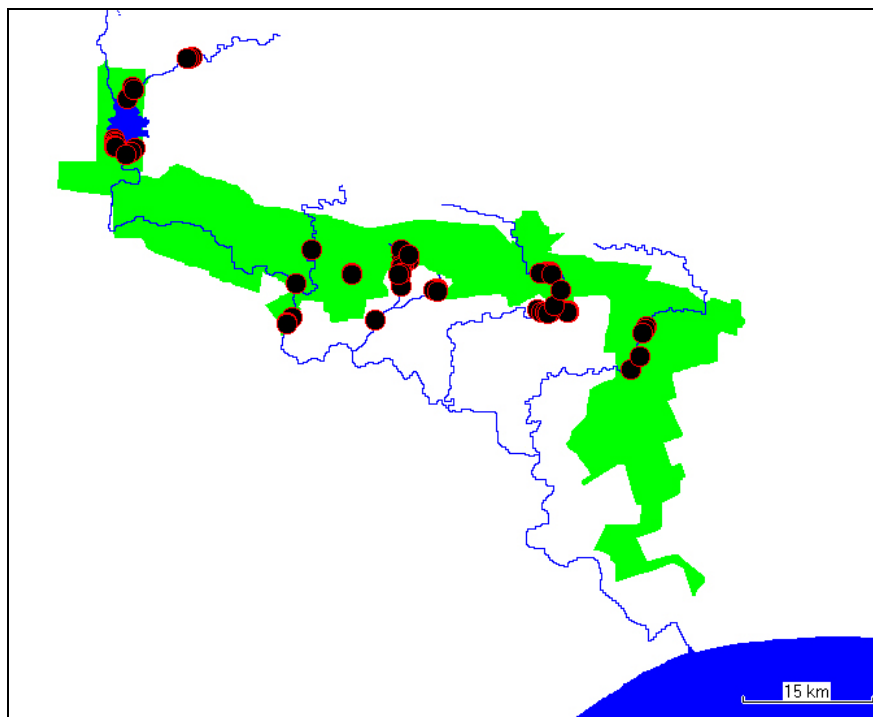


Figure 2.7. Seventy-three sampling sites used during the ichthyological survey of the Greater Addo Elephant National Park.

Migration corridors

Sampling both above and below dam walls and weirs was considered important when determining whether indigenous species were being prevented from moving upstream, and to determine whether alien species had moved upstream past these barriers. At all dam walls under consideration, sampling was conducted once both immediately above the wall, and immediately below the wall. If alien species were found on one of the tributaries of the Sundays River, attempts were made to find all natural barriers that would possibly impede the movement of these invaders upstream.

2.3. Ichthyofauna of the Sundays River System

The ichthyofauna of the Sundays River system can be considered depauperate, consisting of 17 species of which eight species are alien. Of the indigenous species sampled, four were primarily freshwater in origin, three were estuarine and two species were catadromous.

Indigenous freshwater species

The chubbyhead barb, *Barbus anoplus*, is widely distributed in southern Africa, occurring in the Limpopo, upper Orange, Olifants, Gourits, Gamtoos, Sundays and Great Fish Rivers (Skelton, 1993). Within the Sundays River System, it is known to occur in the Sundays and Skoenmakers Rivers. It is a shoaling fish that prefers cooler waters and inhabits a diverse array of habitats ranging from small rivers to lakes and dams. It is a structure oriented species and is frequently found amongst fallen trees and in marginal vegetation. It feeds on insects, zooplankton, seeds,

algae and diatoms (Skelton, 1993). It is potamodromous, moving upstream during flood conditions to breed (Cambray, 1990).

The goldie barb, *Barbus pallidus*, can be found in the coastal streams of the Eastern Cape, from the Great Fish River to the Krom River and in some tributaries of the Vaal, Limpopo, Tugela and Phongolo Rivers (Skelton, 1993). Voucher specimens indicate that this species occurs in the Sundays and Wit Rivers. What is known about this species is that it can be found in the pools of clear rocky streams with emergent marginal vegetation. It breeds in summer (Skelton, 1993).

The moggel, *Labeo umbratus*, attains a maximum size of 50 cm TL and 2.8 kg and occurs in the Orange-Vaal, Gourits, Gamtoos, Sundays, Great Fish and Bushmans River systems (Skelton, 1993). It has been introduced into the Keiskamma, Buffalo and Olifants-Limpopo River systems. *Labeo umbratus* prefers still or slow-flowing waters, and does extremely well in small impoundments (Skelton, 1993, Potts et al., 2005). It feeds on detritus and algae (Potts, 2003) and is a facultative potamodromous species (Cambray, 1990, Skelton, 1993).

The Eastern Cape redfin, *Pseudobarbus afer*, occurs in the upper catchments of rivers from Algoa Bay to Mossel Bay (Skelton, 1993). It is a shoaling species that inhabits clear rocky pools (Skelton, 1993). It is omnivorous, feeding on algae and small invertebrates (Skelton, 1993). It breeds in summer (Cambray, 1990, Skelton, 1993). Information on age and growth (Cambray and Hecht, 1995) indicates that this species grows to 95 mm SL, and up to 6 years of age. The Eastern

Cape redfin is IUCN Red-Listed due to its limited natural range and the threats posed by alien species (Swartz and Impson, 2007).

Indigenous estuarine species

The estuarine roundherring, *Gilchristella aestuaria*, is endemic to southern Africa (Skelton, 1993). It ranges from the Kosi system on the east coast of southern Africa to the Orange River estuary on the western coast (Whitfield, 1998). It is a shoaling, zooplanktivorous species that is commonly found in both turbid and clear estuaries in southern Africa. It is also common in some freshwater coastal lakes (Whitfield, 1998). It attains 9 cm SL, matures after 7 months and breeds throughout the year with peak in spring and summer (Skelton, 1993, Whitfield, 1998).

The river goby, *Glossogobius callidus*, is found in both freshwater and estuarine systems (Whitfield, 1998). It occurs from Malawi to the Swartvlei region in the Western Cape. It is tolerant of both lotic and lentic waters. It attains sexual maturity at 35 mm SL, and breeds in spring (Whitfield, 1998). In freshwater, this species feeds mainly on chironomid, trichopteran and ephemeropteran larvae, and in estuaries, its diet consists of amphipods and chironomid larvae (Whitfield, 1998).

Catadromous species

Longfin eel, *Anguilla mossambica*, is a catadromous species found in the eastern coastal rivers of Africa and grows to 150 cm TL and up to 5 kg (Skelton, 1993). Voucher specimens from the Albany Museum and SAIAB indicate that this species occurs within the Sundays River and in the Wit River. It has been suggested that adults breed east of Madagascar and the Agulhas Current transports the larvae and leptocephali south along the eastern African coastline (Skelton, 1993). Elvers then migrate up rivers, overcoming obstacles such as dam walls and waterfalls (Bruton et al., 1987). Once in freshwater, elvers will feed and grow there for at least ten years before returning to sea to breed (Skelton, 1993). When *A. mossambica* are smaller than 20 cm TL, they consume invertebrates, and above this length, crustacea and fish (Bruton et al., 1987). The length of residency in freshwater differs for males and females, with males spending between eight and ten years, and females between 15 and 20 years in freshwater (Bruton et al., 1987). The maximum age reported for *A. mossambica* is 18 years (Balon, 1975).

The catadromous freshwater mullet, *Myxus capensis*, is endemic to southern Africa and occurs from the Breë River in the Cape to the Kosi River in KwaZulu-Natal (Skelton, 1993). This species is catadromous, migrating from freshwater to the sea to spawn (Skelton, 1993). Because dams and weirs restrict their migrations upstream, this species has become rare, and is now on the IUCN Red List (Skelton, 1993). It feeds on algae and benthic invertebrates (Bok, 1983).

Alien freshwater species

Largemouth black bass, *Micropterus salmoides*, were first introduced into South Africa in 1928 for angling purposes, and has since become widespread throughout the country (de Moor and Bruton, 1988, Skelton, 1993). *Micropterus salmoides* have been introduced into approximately 69 countries, as well as suitable reservoirs throughout western North America (Iguchi et al., 2004). *Micropterus salmoides* can tolerate temperatures from 5° to 36.5° C, with an optimum temperature of 28° C (de Moor and Bruton, 1988, Skelton, 1993). It prefers clear, slow-flowing or still waters with an abundance of aquatic macrophytes (Skelton, 1993, Iguchi et al., 2004). *Micropterus salmoides* is primarily piscivorous but will feed on any animal prey, including insect larvae, frogs, crabs, snakes and even small mammals (Iguchi et al., 2004, Garcia-Berthou, 2002).

African sharptooth catfish, *Clarias gariepinus*, is naturally distributed from Turkey south to the Orange River (de Moor and Bruton, 1988, Skelton, 1993). It have been translocated into the Eastern and Western Cape provinces (de Moor and Bruton, 1988, Cambray, 2003b) and introduced into rivers as far afield as Brazil (Cambray, 2005). This species is potamodromous, can tolerate wide fluctuations of environmental conditions, including low oxygen conditions and high turbidity, and has a catholic diet (Spataru et al., 1987, de Moor and Bruton, 1988, Skelton, 1993). De Moor & Bruton (1988) indicate that, although it is suspected that this species is likely to have a detrimental impact on native fauna, there is insufficient quantitative data to corroborate this.

Common carp, *Cyprinus carpio*, was introduced into South Africa during the 1700s and 1800s for angling purposes, and are widespread throughout the country (de Moor and Bruton, 1988, Skelton, 1993). It has a wide tolerance of environmental conditions, and favours slow-flowing or still, turbid waters (de Moor and Bruton, 1988, Skelton, 1993). It is omnivorous, feeding mainly on insects and plant matter in sediments, but will consume the eggs and larvae of other fish (Crivelli, 1981, de Moor and Bruton, 1988). This feeding habit makes this species a pest in many environments because it agitates sediments and out competes and suppresses the spawning efficacy of native species (de Moor and Bruton, 1988, Pimentel et al., 2005). It is listed as one of the world's 100 worst alien species (Lowe et al., 2000).

Western mosquitofish, *Gambusia affinis*, has a scattered, isolated distribution in South Africa, with populations in Gauteng, the Eastern Cape and the Western Cape (Skelton, 1993). It inhabits still and slow-flowing waters with suitable aquatic vegetation (de Moor and Bruton, 1988, Skelton, 1993). It feeds mainly on small live organisms, including mosquito and fish larvae (Rehage et al., 2005). It was introduced into South Africa in 1936 by aquarists, but the Cape Inland Fisheries Department distributed it countrywide in an attempt to curb problematic mosquito populations (Skelton, 1993). In South Africa, their impact is believed to be minimal (de Moor and Bruton, 1988), although it has been responsible for local extinctions of native topminnows elsewhere (Meffe, 1984, Caiola and DeSostoa, 2005). It is listed as one of the world's 100 worst alien species (Lowe et al., 2000).

Orange River mudfish, *Labeo capensis*, are native to the Orange-Vaal system, but have been translocated to the Great Fish and Sundays river systems via inter-basin water transfer schemes

(Cambray and Jubb, 1977). It prefers slow-moving or still waters and is herbivorous (de Moor and Bruton, 1988, Skelton, 1993). It has been known to hybridise with the native moggel, *Labeo umbratus* (Tweddle and Ter Morshuizen, 1996).

Smallmouth yellowfish, *Labeobarbus aeneus*, are native to the Orange-Vaal system, but have been introduced into the Limpopo River and Lake Kyle, Zimbabwe (de Moor and Bruton, 1988, Skelton, 1993). It has been translocated to the Great Fish and Sundays systems via inter-basin water transfer schemes (Cambray and Jubb, 1977). It inhabits clear, flowing waters, but can survive in turbid waters (de Moor and Bruton, 1988, Skelton, 1993). It is omnivorous, feeding mainly on insects, algae and other plant material (Stadtlander, 2006). The potential impacts of this species in non-native environments is not known (de Moor and Bruton, 1988).

Mozambique tilapia, *Oreochromis mossambicus*, is found naturally in the eastern coastal rivers, from the lower Zambezi system, south to the Bushmans River (Skelton, 1993). It has been widely dispersed throughout the country (de Moor and Bruton, 1988, Skelton, 1993). It inhabits still to moderately fast-flowing waters, and is tolerant of a wide range of environmental conditions (de Moor and Bruton, 1988, Skelton, 1993). It is limited by low temperature conditions, during which mass mortalities have been recorded (de Moor and Bruton, 1988). Its diet consists of algae and plant matter, but it will prey on insects and small fish (de Moor et al., 1986, Skelton, 1993). It has been listed as one of the world's 100 worst alien species (Lowe et al., 2000).

Banded tilapia, *Tilapia sparrmanii*, are widely distributed from the Orange River north to Lake Malawi (de Moor and Bruton, 1988, Skelton, 1993). They have been translocated into the Eastern and Western Cape provinces (de Moor and Bruton, 1988, Skelton, 1993). It inhabits still waters with submerged or emergent aquatic macrophytes, is tolerant of a wide range of environmental conditions, and feeds on a wide variety of prey items, including small invertebrates and small fish (de Moor and Bruton, 1988, Skelton, 1993, Winemiller and Kelso-Winemiller, 2003). It was widely distributed as a forage fish for *Micropterus* species (de Moor and Bruton, 1988). Due to competition with and predation on native ichthyofauna, this species has had a detrimental impact in both the Gourits and Olifants rivers (de Moor and Bruton, 1988). It may also be a vector for alien parasites (de Moor and Bruton, 1988).

2.3. General sampling procedures

Pre-sampling procedure

At each site, GPS co-ordinates were recorded on a Garmin© E-trex Legend series GPS unit. Accuracy of this unit was usually within five metres. Oxygen concentration and saturation and water temperature was measured using an OxyGuard oxygen probe. Conductivity, total dissolved solid concentrations and pH were measured with a Hanna HI98129 Combo pH and EC meter. Mean width, length and depth of each sampling site was visually estimated. The hydrology of the sampling site was recorded as either a pool, dam, riffle or run. According to the classification given by Fievet et al. (1999), a riffle was defined as a stretch of running water that exhibits “white” water, a run as a stretch of running water that did not exhibit “white” water, and a pool as a stretch of water exhibiting no movement at all. Substrate type was separated into several

categories, namely mud, sand, gravel, stones, boulders and bedrock. The substrate of each sampling site was classified according to a percentage of each of the substrate categories.

Aquatic macrophyte growth was classified according to the percentage of total area that that particular type of macrophyte covered in relation to the total surface area of water in the sampling site. The composition of bankside vegetation was recorded for each sampling site. The presence of black wattle, *Acacia mearnsii*, was noted wherever it occurred. The land use adjacent to each sampling site was noted. If the adjacent land was not in a natural state, the type of land use was noted (i.e. citrus agriculture, camping site, picnic area, cattle ranching). Other pertinent information, such as the presence of dangerous game (buffalo, elephant, venomous snakes) , equipment failure and weather conditions, was noted for future reference.

Passive gear

Double-ended fyke nets, with eight metre long lead lines and 20mm stretched mesh, were set in water deeper than 60 cm. Due to the presence of Cape clawless otter, *Aonyx capensis*, in the area, otter guards of 5 cm diameter plastic mesh were inserted into the entrance of each fyke net. The fyke nets were set in the late afternoon, and retrieved at sunrise the next morning. Catches from different fyke nets were recorded separately in order to calculate differences in catch-per-unit-effort.

Initially, two fish traps were constructed out of wire and mosquito netting. A further four fish traps were constructed out of two litre soft-drink bottles. These traps were baited with bread,

weighted down with six ounce teardrop sinkers and set in areas where aquatic macrophytes were plentiful. The fish traps were set overnight and during the day. All traps were unsuccessful in capturing fish, and were eventually discarded as a sampling technique.

A fleet of gill nets was used on the larger pools in the tributaries of the Sundays River and in the Sundays River. A fleet of 45 m gill nets was used in the Sundays River, while a fleet of 30 m gill nets was used on the Wit River. The 45 m gill nets were constructed with five 9 m panels of 44, 60, 75, 100 and 144 mm stretch multifilament mesh, while the 30 metre gill nets were constructed with three 10 m panels of 60, 75 and 100 mm stretch multifilament mesh. In Darlington Dam, three 45 m gill nets were deployed, while in the Sundays River between Darlington Dam and Korhaansdrift Dam, two 45 m gill nets and three 30 m gill nets were used. In Korhaansdrift Dam and in the Sundays River below the impoundment, a single 45 m gill net was used. A single 30 m gill net was used in a large pool on the Wit River. Gill nets were set in the late afternoon and retrieved early the next morning.

Two different types of longlines were used each with a distinct function. The first longline was designed to sample larger *C. gariepinus* and *A. mossambica* in the Sundays River and its associated impoundments, while the second longline was designed to sample for smaller predatory fish in the pools of the tributaries of the Sundays River. The more robust longline, used in the Sundays River and its impoundments, was constructed with polyethylene rope, with snoods constructed with 115kg test nylon and 9/0 Mustad[®] circle hooks. These longlines were baited with fish fillets or chicken offal. This longline was weighted down with concrete anchors, and scuba diving floats were used to keep the longline visible. The longline designed for the

tributaries was made out of 27 kg test Maxima[®] Ultra-Green[®] nylon, and the snoods were constructed with 13 kg test Maxima[®] Ultra-Green[®] nylon and No. 2 and No. 4 Bandit[®] Octopus[®] hooks. These longlines were baited with small fish or chicken offal. One end of this longline was tied onto a tree trunk near the pool to be sampled, and the other end was weighed down with a six-ounce teardrop sinker and thrown into the pool. Due to the presence of crabs, bubble floats were attached to the snoods to keep the baits off the substrate. Longlines were set in the late afternoon and retrieved early the next morning.

Active gear

An 8 m x 2 m seine net with 3 mm stretch mesh was used on the Sundays River. A single pull was conducted, immediately below Darlington Dam, but due to the amount of boulders found on the substrate in the majority of sampling sites, seine netting was deemed to be ineffective and was therefore not further utilised.

Electrofishing was conducted throughout the Sundays River system. A Samus[®] 725G backpack electrofisher, attached to a 12 V electric gate motor battery, was used. The settings for the electrofisher were standardised at a duration of 0.3 milliseconds and a frequency of 80 Hz. In shallow pools (< 70 cm deep), three-pass depletion electrofishing was conducted. In deeper pools (> 70 cm deep), single-pass electrofishing was conducted on the fringes of the pool in order to collect presence/absence data.

Micropterus salmoides, which are present in Slagboom Dam are difficult to sample with gill nets, and seem to be able to effectively avoid electrofishing, so artificial angling was used to sample for this species. *Cyprinus carpio*, which was found throughout the Sundays River, was also difficult to sample effectively with standard methods (i.e. gill netting, fyke nets, electrofishing). Angling for this species was conducted in Darlington Dam, Korhaansdrift Dam, Slagboom Dam and the Sundays River using maize meal as bait.

Measuring and data collection

According to SANParks Standard Operating Procedures, all alien fish species that were sampled were sacrificed. *Cyprinus carpio*, *L. capensis*, *M. cephalus*, *M. salmoides* and *L. aeneus* were measured for fork length (FL), while *O. mossambicus*, *C. gariepinus*, and *T. sparrmanii* were measured for total length (TL). *Gambusia affinis* and *G. aestuaria* were measured for standard length (SL). All measurements were to the nearest millimetre. Larger fish were sexed and the gonads were macroscopically staged. Otoliths were also removed from these fish. Smaller fish (*G. affinis*, *G. aestuaria* and *T. sparrmanii*) were preserved in 10% formalin and accessioned into the South African Institute for Aquatic Biodiversity's (SAIAB) fish collection for locality records.

Indigenous fish species were all released, with the exception of limited specimens retained for accessioning into SAIAB's fish collection for locality records. *Labeo umbratus* were measured for fork length (FL), *A. mossambica* for total length (TL) and *P. afer*, *B. pallidus* and *G. callidus* were measured for standard length (SL). *Labeo umbratus* captured in gill nets were sacrificed for

biological examination as they were usually too damaged to survive release. These fish were sexed and visually staged, and asteriscus otoliths were removed. The smaller barb species (*P. afer* and *B. pallidus*) were examined for external parasites.

2.4. Meta-data sources

Past fish distribution data were gathered from various sources, including an *Atlas of southern Africa's freshwater fishes* (Scott et al., 2006), SAIAB's fish collection database and the Albany Museum's fish collection database. Information on relative abundance and distribution for Addo Elephant National Park rivers was taken from Russell (1999). The SANParks Addo Elephant National Park management plan (2006) was used to gather information on the extent of the Park, and on the various types of water bodies that were found in the Park. Information on the flow regimes of the various rivers was found in Roux *et al.* (2002) and corroborated with anecdotal accounts from the section rangers who govern the various rivers. Information on the Orange-Fish tunnel and its effects on the Sundays River was gathered from Cambray and Jubb (1977).

Chapter 3. Distribution and relative abundance of freshwater fishes within the Sundays River System

3.1. Introduction

There are fewer than 100 indigenous freshwater fish species in southern Africa, of which about half are endemic (Skelton, 2002). Twenty-eight of these endemic species are listed in the South African Red Data Book (Skelton et al., 1995). In a recent assessment, it was determined that the conservation status of South Africa's freshwater fish had declined since its last assessment in 1987 (Skelton, 2002). The major threats to these fishes include habitat destruction, dams and weirs, water abstraction, pollution and the effects of introduced alien species (Skelton, 1983, Skelton, 1987, Skelton, 2002, Woodford and Impson, 2004).

A high-level objective of the Greater Addo Elephant National Park (GAENP) is to “conserve a representative sample of Eastern Cape ecological patterns and processes in a contiguous arrangement by establishing a connected land-seascape enabling natural variation in structure, function and composition in space and time” (SANParks, 2006, pg 9). The Sundays River and its tributaries are considered essential components of the ecosystem, and improving and restoring hydrological regimes and natural functioning of the Sundays River is a high priority within the GAENP management plan (SANParks, 2006). The optimal method of conserving freshwater biota is to conserve either entire watersheds (Skelton, 1990, Angermeier and Winston, 1999, Keith, 2000), or the headwaters of tributaries (Saunders et al., 2002, Abell et al., 2007). Since the GAENP contains within its boundaries the headwaters of four of the Sundays River tributaries,

the GAENP has an opportunity to contribute to the conservation of the indigenous freshwater ichthyofauna (Russell, 1999).

It is recognised that the inclusion of freshwater fish into the overall management of the park will require an understanding of the distribution, biology, ecology and impact of alien fish species on the freshwater biota within the park. Unfortunately, there is a severe paucity of such information (Roux et al., 2002). Roux *et al.* (2002), in their biodiversity assessment of the freshwaters of the AENP, found that biological information was extremely limited. As a result, their assessment was based on landscape and ecosystem patterns and implied that stream biota would be protected by conserving habitat heterogeneity and pattern. However, comprehensive management requires higher level understanding. For this reason, important baseline information needs to be collected and a major component of this baseline information is the determination of the distribution of alien and indigenous species in the Sundays River system. Information on the geographical distribution of the fish species in the system will provide important information for conservation interventions and management of the ichthyofauna within the GAENP.

3.2. Materials and methods

Locality records from SAIAB and the Albany Museum, both in Grahamstown, were accessed to determine the past distribution of species in the Sundays River catchment. For each species, a new Garmin[®] MapSource[®] file was created, and the locality points for that species were entered into the file as waypoints. Data from an *Atlas of southern African freshwater fish species* (Scott et al., 2006) were then examined and any relevant locality points were added to the appropriate

species' MapSource[®] file. Since many authors conducting distribution studies do not accession their findings into museums and fish collections, it was important to determine if any of the studies that had been conducted in the Sundays River catchment could be used to add to the past distribution records. Two such studies were found, in Tweddle and Ter Morshuizen (1996), and Russell (1999). Any additional locality records from these studies were added to the appropriate species' MapSource[®] file. Relative abundances of species from past studies were obtained from Tweddle and Ter Morshuizen (1996) for the Skoenmakers and Sundays rivers, and from Russell (1999) for the Sundays, Kabouga, Uie, Wit and Krom rivers.

Data deficient areas were identified and prioritised for sampling effort from these distribution maps. The areas that were given the highest priority were those rivers that contained IUCN Red-Listed species, or that were potentially continually introducing alien species into the Sundays River catchment. Rivers containing conservation critical species were the Kabouga River, the Uie River and its two tributaries, the Wit River and the Krom River, and the river that was identified as potentially introducing alien species into the catchment was the Skoenmakers River (Figure 3.1).

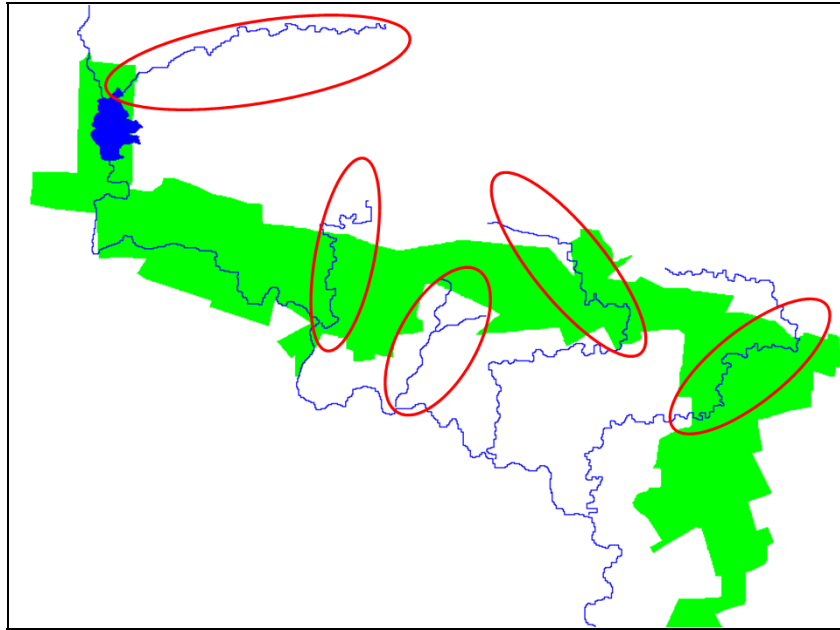


Figure 3.1. Data deficient areas in the GAENP as identified by locality records from SAIAB and the Albany Museum, and from previous ichthyological surveys. The areas include, from left to right, the Skoenmakers, Kabouga, Uie, Wit and Krom rivers.

An ichthyological survey of the rivers in the GAENP was conducted between September 2007 and February 2008 to determine the current distribution of freshwater fishes.

The GPS co-ordinates of the sampling sites from where each species was sampled were entered into the appropriate species' MapSource[®] file. Relative abundance data was graphed alongside the relative abundance data from Tweddle and Ter Morshuizen (1996) and Russell (1999) in SigmaPlot[®] 8. Relative abundance data, in the form of numbers of fish per sampling gear type per river, from previous studies by Tweddle and Ter Morshuizen (1996) and Russell (1999) were

compared to relative abundance data from this study, using a 3 × 16 species contingency table.

Expected values for the contingency tables were calculated by using:

$$\hat{e} = \frac{\sum n_y \sum n_s}{\sum n}$$

where $\hat{e}$ = expected value, n_y = sum of numbers of all species sampled during the y^{th} study, n_s =

sum of the numbers of that species captured throughout all study periods and n = sum of numbers of all species captured during all study periods.

3.3. Results

A total of 3313 fish, comprising 16 species (Table 3.1), were captured from the 73 sampled sites.

A total of three species showed evidence of an increase in the extent of their distribution, while six species were not sampled from parts of their previous distributions. Six species exhibited no change to their distribution, and two species were found that had not previously been recorded in the system.

Table 3.1. Checklist of freshwater and estuarine fish species found in the freshwater section of the GAENP. I = indigenous, C = catadromous, A = alien, T = translocated through the Orange-Fish water transfer scheme, E = estuarine, S = stocked. 1 = decrease in range, 2 = increase in range, 3 = no change in distribution, 4 = new species.

Species	Status	Tweddle & Ter Morshuizen 1996	Russell (1999)	This study
<i>Anguilla mossambica</i> ²	I; D	-	X	X
<i>Barbus anoplus</i> ¹	I	X	X	X
<i>Barbus pallidus</i> ¹	I	-	X	X
<i>Clarias gariepinus</i> ²	A; T	X	X	X
<i>Cyprinus carpio</i> ³	A; T	X	X	X
<i>Gambusia affinis</i> ⁴	A; T	-	-	X
<i>Gilchristella aestuaria</i> ³	I; E; S	X	X	X
<i>Glossogobius callidus</i> ²	I; E	X	X	X
<i>Labeo capensis</i> ⁴	A; T	-	-	X
<i>Labeo umbratus</i> ¹	I	X	X	X
<i>Labeobarbus aeneus</i> ¹	A; T	-	X	X
<i>Micropterus salmoides</i> ³	A; S	-	-	X
<i>Mugil cephalus</i> ³	I; E; S	-	-	X
<i>Myxus capensis</i> ³	I; E	-	X	X
<i>Oreochromis mossambicus</i> ¹	A; T	X	X	X
<i>Pseudobarbus afer</i> ³	I	-	X	X
<i>Tilapia sparrmanii</i> ¹	A; T	-	-	X

Distribution

Indigenous cyprinids

Prior to this study, the distribution of *B. anoplus* was widespread throughout the Sundays River, extending up into Darlington Dam, the Skoenmakers River, and also the Kabouga River.

However, *B. anoplus* were found at only one site on the Kabouga River (Figure 3.3). The site at which this species was found was a stagnant pool (Figure 3.2) with low oxygen concentrations.

This suggests that this species may be able to persist through long periods of drought.

Barbus pallidus was found to be widespread throughout the tributaries, but absent from the Sundays River at sites sampled during this survey. Previous records have indicated that this species was at one time present in the Sundays River (Figure 3.3).

Locality records from previous studies indicate that *L. umbratus* was widespread throughout the Sundays River and Darlington Dam, and could be found in the Skoenmakers, Krom, Kabouga and Wit rivers (Figure 3.3). *Labeo umbratus* was found at all sites sampled along the Sundays River, and at in Darlington Dam. It was also found in the Skoenmakers River and in Slagboom Dam on the Wit River. It was not present in the Krom or the Kabouga rivers.

Pseudobarbus afer were previously found in the Kabouga, Wit, Klein Uie, Groot Uie and Krom rivers. This species was found to be widespread in the Wit, Klein Uie, Groot Uie and Krom rivers in the current study, but were not sampled from the single site sampled on the Kabouga River (Figure 3.3).

Other indigenous species

In previous studies, *A. mossambica* was sampled from Darlington Dam, the Sundays River and the Wit River. Based on the sampling conducted during this study, new locality records for this species were noted for the Kabouga, Uie and Klein Uie rivers (Figure 3.4) This species was found to endure the harshest of habitats and can be expected to persist in the tributaries through the worst drought conditions (Figure 3.2.). This species was not sampled in the Krom River.

Gilchristella aestuaria was found in Darlington Dam where they were stocked (Dr. A. Bok, *pers. comm.* 2007) and in the Sundays River, downstream to Korhaansdrift Dam (Figure 3.4). They were absent from a pool on the Sundays River directly downstream from Korhaansdrift Dam. This species was previously recorded from this site, and also from the Wit River but no evidence of their presence was found during this study.

Glossogobius callidus is widespread throughout the Sundays River system (Figure 3.4). Based on the data collected during this study, this species' recorded range has expanded to include the Skoenmakers River.

Alien species

Clarias gariepinus was translocated from the Great Fish River through the Orange-Fish tunnel system into the Skoenmakers River and then into Darlington Dam and the Sundays River. It has extended its range to all of the tributaries that flow through the GAENP (Figure 3.5). *Clarias gariepinus* were found to inhabit the harshest of environments, including low oxygen concentrations and extreme drought conditions (Figure 3.2).

Cyprinus carpio was sampled in Darlington Dam, the Sundays River and the Skoenmakers River (Figure 3.5), a distribution range similar to previous studies. This species was also sampled from Slagboom Dam on the Wit River.

Gambusia affinis has not been recorded in the system prior to this study (Figure 3.5). Samples were collected in the Skoenmakers, Sundays and Kabouga rivers, as well as in Darlington Dam.

Labeo capensis has not been recorded in the Sundays catchment prior to this study. It is now widespread throughout the Sundays River, including Darlington Dam, and the Skoenmakers River (Figure 3.5), having been translocated via the Great Fish River through the Orange-Fish water transfer scheme.

Labeobarbus aeneus was only sampled in the Skoenmakers River and at the confluence of the Sundays and Kabouga rivers previously. Tweddle and Ter Morshuizen (1996) predicted that this species would colonise Darlington Dam and the lower Sundays River below Darlington Dam, a prediction confirmed during the course of this study (Figure 3.6). A single juvenile was also sampled at one site along the Kabouga River.

Locality records indicated that *Micropterus salmoides* was limited in distribution to the lower Wit River (Figure 3.6). This species was present in Slagboom Dam, at one site just upstream of the dam, and below the Slagboom Dam wall. This species may have, at one time, occurred in the

upper Wit River (M. Manthe, Wells Park farm owner, *pers. comm.*, 2007). Sampling, however, did not corroborate this.

Based on locality records from previous ichthyological surveys, *Oreochromis mossambicus* is present in the Skoenmakers, Sundays, Wit and Krom rivers, and in Darlington Dam. Data from this survey revealed that *O. mossambicus* is present in the Skoenmakers and Sundays rivers and in Darlington Dam (Figure 3.6).

Tilapia sparrmanii were previously recorded from the Skoenmakers, Sundays and Wit rivers, and Darlington Dam. In this study, the species was found to occur at only one sampling site on the Skoenmakers River (Figure 3.6).



Figure 3.2. Environments in which the longfin eel, *Anguilla mossambica*, was found to persist through drought conditions. On the left, a small weir structure from which one individual was sampled with an electrofisher and a further two were observed. A single sharptooth catfish, *Clarias gariepinus*, in poor condition, was also sampled from this weir. On the right, a pool on the Kabouga River, from which a single *A. mossambica* was sampled with a small longline. Electrofishing at this site produced numerous *Barbus anoplus*, some *Gambusia affinis* and a single *Labeobarbus aeneus*.

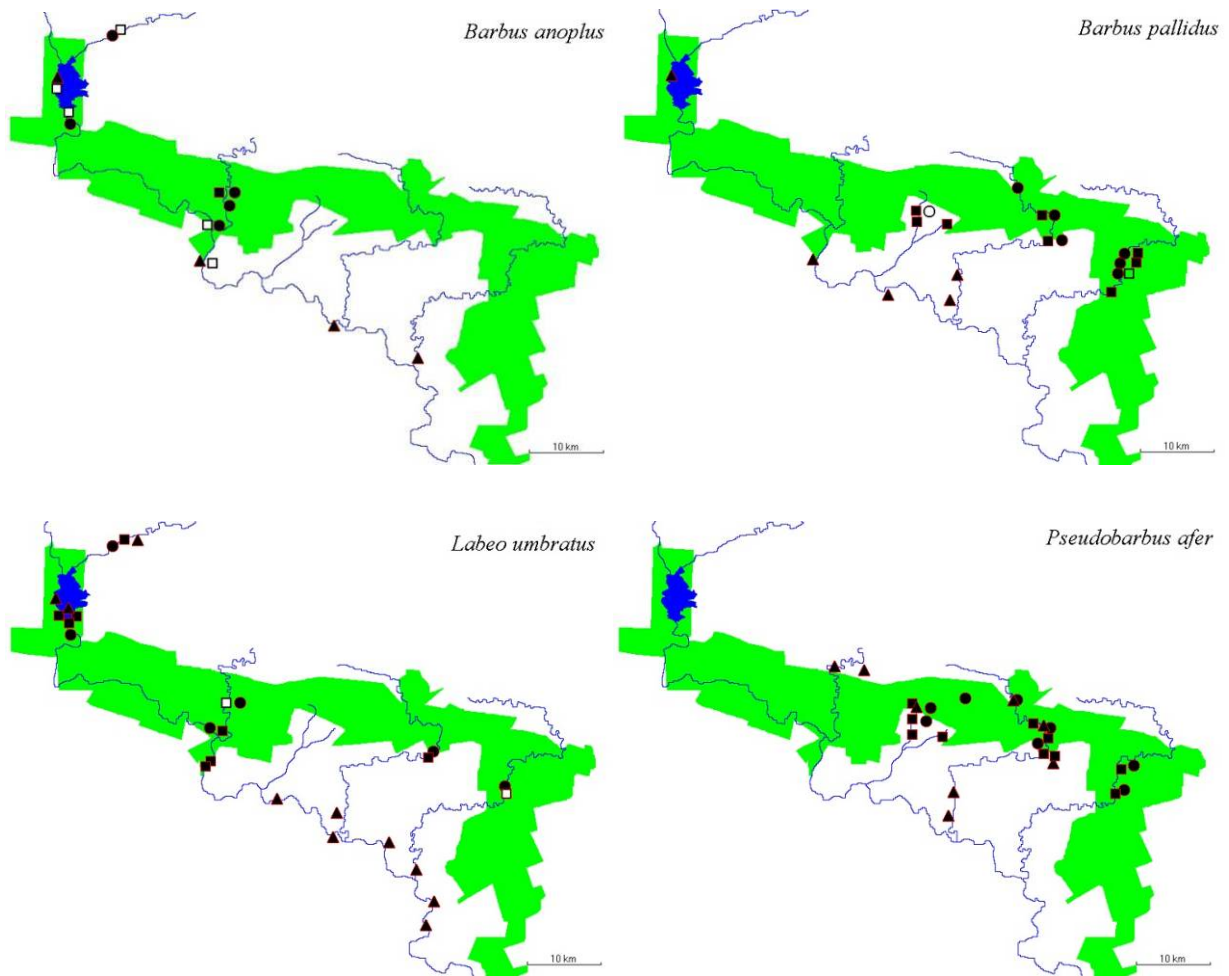


Figure 3.3. Distribution of indigenous cyprinids within the Sundays River System. ● indicate localities where the species was sampled during previous studies, ○ indicate localities where sampling was conducted in the past, but the species was not sampled. ■ indicate localities where the species was sampled during this study. □ indicate localities sampled during this study where species were previously sampled, but were not sampled in this study. ▲ indicate locality records from SAIAB and the Albany Museum.

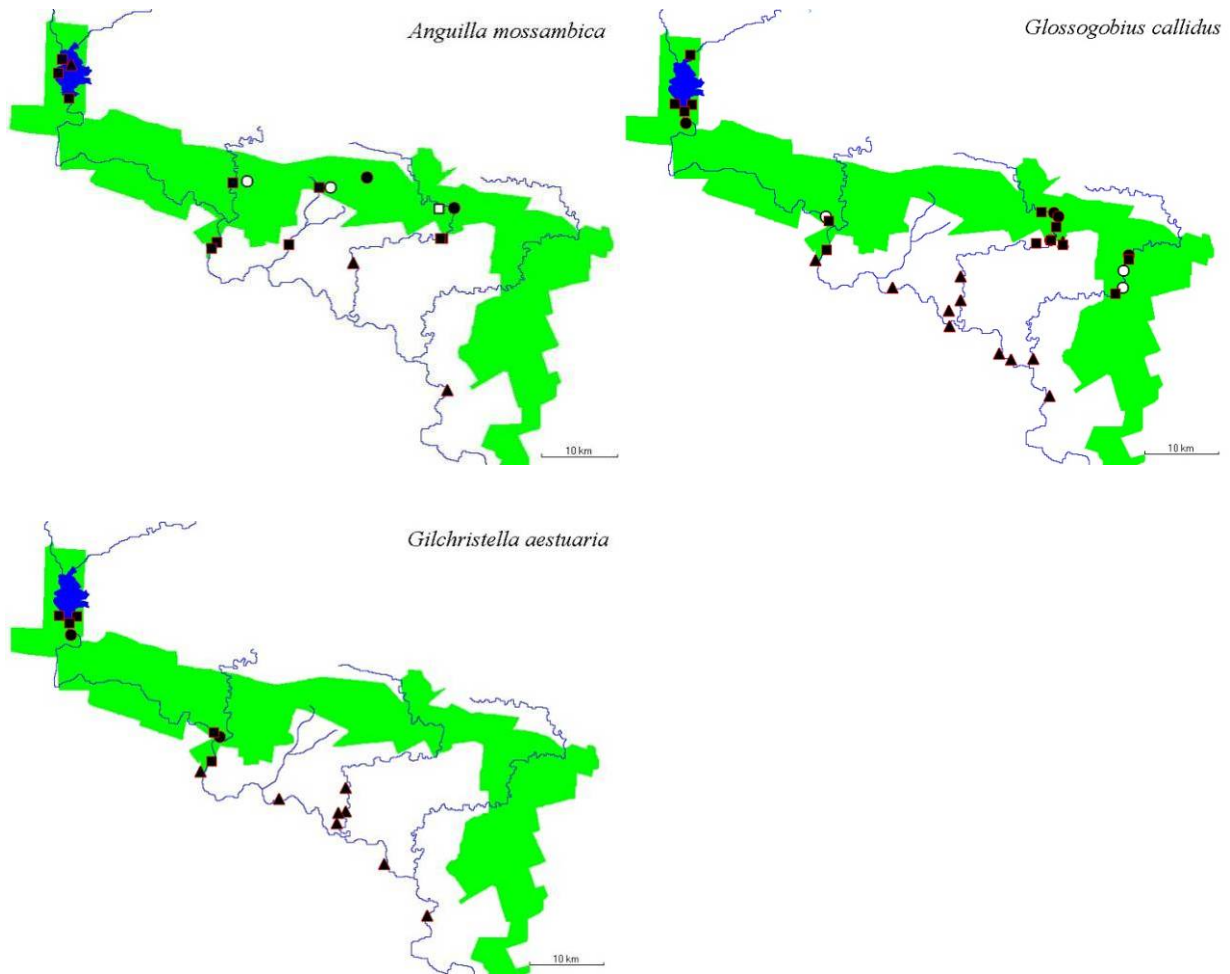


Figure 3.4. Distribution of other indigenous species within the Sundays River System. ● indicate localities where the species was sampled during previous studies, ○ indicate localities where sampling was conducted in the past, but the species was not sampled. ■ indicate localities where the species was sampled during this study. □ indicate localities sampled during this study where species were previously sampled, but were not sampled in this study. ▲ indicate locality records from SAIAB and the Albany Museum.

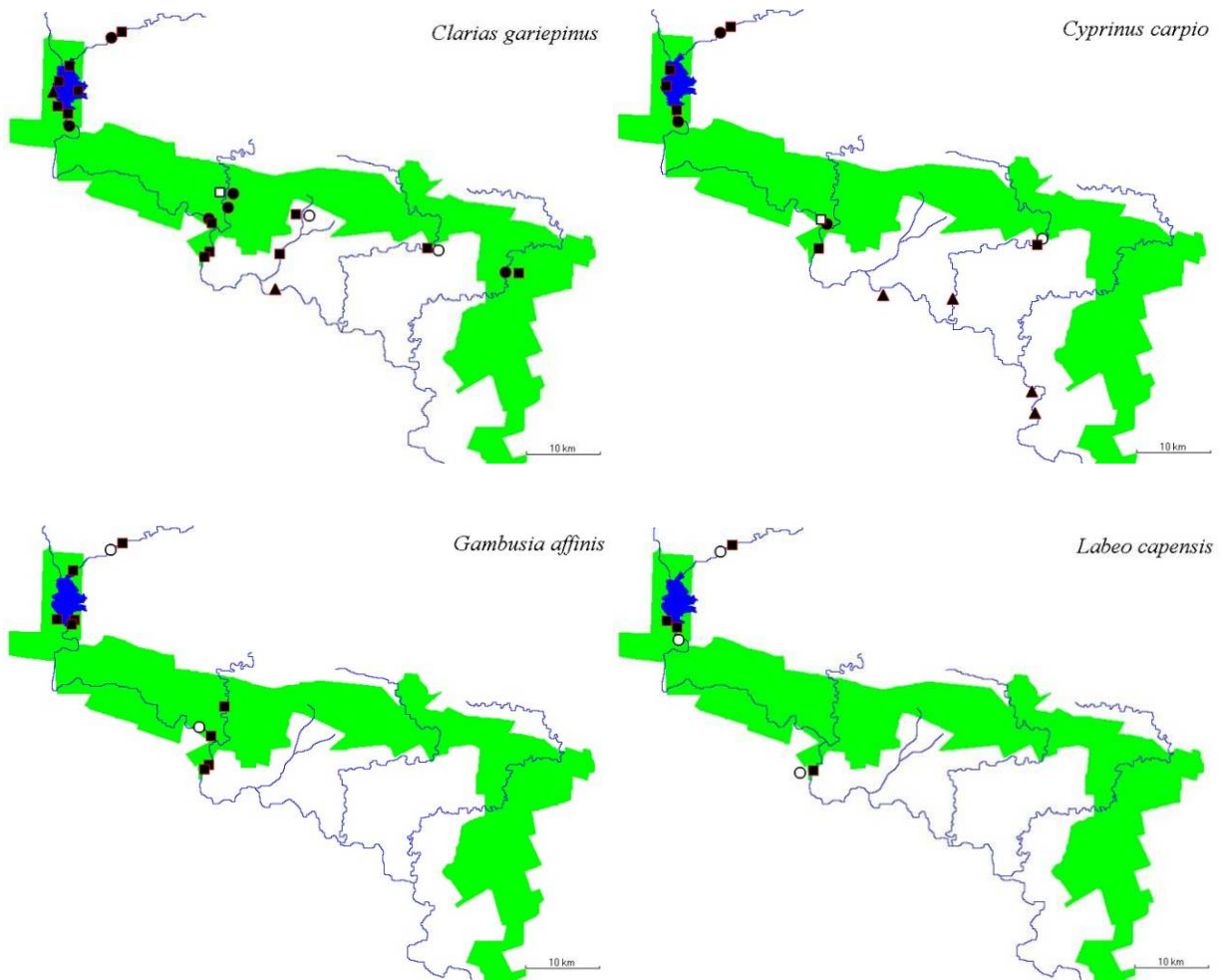


Figure 3.5. Distribution of alien species, *Clarias gariepinus*, *Cyprinus carpio*, *Gambusia affinis* and *Labeo capensis*, within the Sundays River System. ● indicate localities where the species was sampled during previous studies, ○ indicate localities where sampling was conducted in the past, but the species was not sampled. ■ indicate localities where the species was sampled during this study. □ indicate localities sampled during this study where species were previously sampled, but were not sampled in this study. ▲ indicate locality records from SAIAB and the Albany Museum.

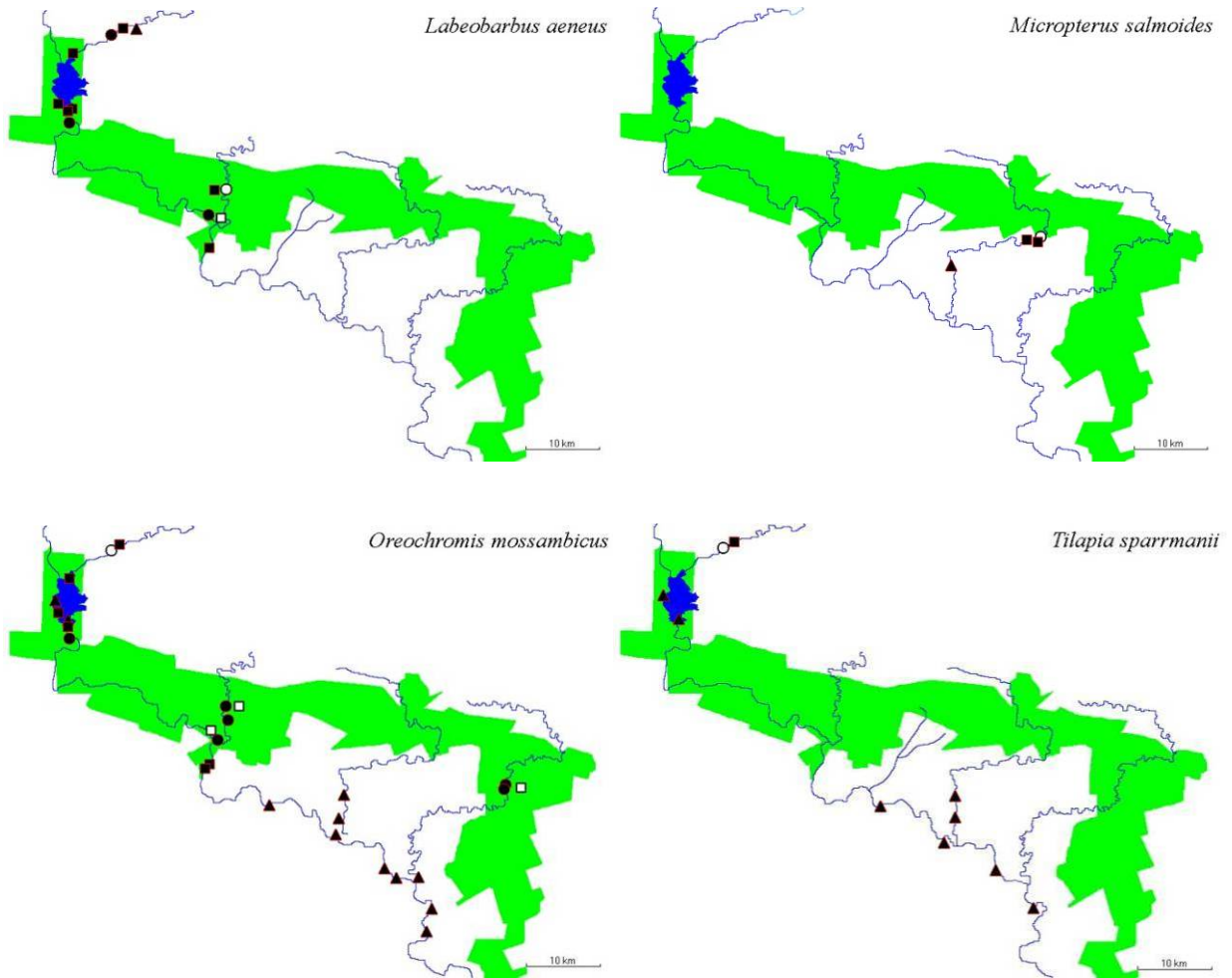


Figure 3.6. Distribution of alien species, *Labeobarbus aeneus*, *Micropterus salmoides*, *Oreochromis mossambicus* and *Tilapia sparrmanii*, within the Sundays River System. ● indicate localities where the species was sampled during previous studies, ○ indicate localities where sampling was conducted in the past, but the species was not sampled. ■ indicate localities where the species was sampled during this study. □ indicate localities sampled during this study where species were previously sampled, but were not sampled in this study. ▲ indicate locality records from SAIAB and the Albany Museum.

Relative abundance

All the ichthyofaunal communities in the Sundays River system showed changes in percentage composition, except for those on the Klein Uie River. *Pseudobarbus afer* was the most dominant species in all of the tributaries, with the exception of the Krom River, where *B. pallidus* was dominant. In the Sundays and Skoenmakers rivers, *O. mossambicus* dominated.

Since 1999, the community composition of the Krom River has changed significantly ($\chi^2 = 124.15$, $df = 5$, $p < 0.001$). Russell (1999) noted that *P. afer* was the dominant species, comprising almost 90% of all individuals sampled (Figure 3.9). Based on the data from this study, *B. pallidus* has become the dominant species in the river. Notable species absences from this study are *O. mossambicus* and *L. umbratus*.

The Klein Uie River had a similar dominance pattern to that previously recorded by Russell (1999), although three new species have been recorded in this river albeit in small numbers (Figure 3.9). *Pseudobarbus afer* was the most abundant species in this river, both in 1999 (Russell, 1999) and in 2007.

The Wit River showed a significant change in species composition from 1999 ($\chi^2 = 232.04$, $df = 6$, $p < 0.001$). *Pseudobarbus afer* abundance decreased, while *G. callidus* abundance increased between 1999 (Russell, 1999) and 2007 (Figure 3.9). *Barbus pallidus* abundance has remained at

a similar level. *Clarias gariepinus* were sampled for the first time in this river, but in small numbers.

The community composition of the Skoenmakers River was significantly changed ($\chi^2 = 222.29$, $df = 9$, $p < 0.001$). *Oreochromis mossambicus* was the most dominant species, followed by *C. gariepinus* (Figure 3.8). *Oreochromis mossambicus* were not present during a survey conducted by Tweddle and Ter Morshuizen (1996). Species recorded in this study that have not been previously recorded were *G. affinis* (9.1% by numbers), *T. sparrmanii* (3.5%), *G. callidus* (2.1%) and *L. capensis* (0.7%). *Barbus anoplus* was not sampled during this survey. It was noted in low abundances in the Skoenmakers River (Tweddle and Ter Morshuizen, 1996).

The community composition in the Sundays River was found to have changed significantly since 1999 ($\chi^2 = 319.92$, $df = 20$, $p < 0.001$). *Oreochromis mossambicus* was the most dominant species, followed by *G. callidus* and *G. affinis* (Figure 3.7). Neither Russell (1999) nor Tweddle and Ter Morshuizen (1996) sampled *G. callidus* or *G. affinis* in the Sundays River. In 1996, *G. aestuaria* was the most dominant species (Tweddle and Ter Morshuizen, 1996), while in 1999, *L. umbratus* was the most abundant species (Russell, 1999). Other species recorded during this study that were not sampled by either Tweddle and Ter Morshuizen (1996) or Russell (1999) were *A. mossambica* and *L. capensis*. Neither of these two species were common. *Barbus anoplus* was not sampled during this study although Russell (1999) noted that this species was not abundant.

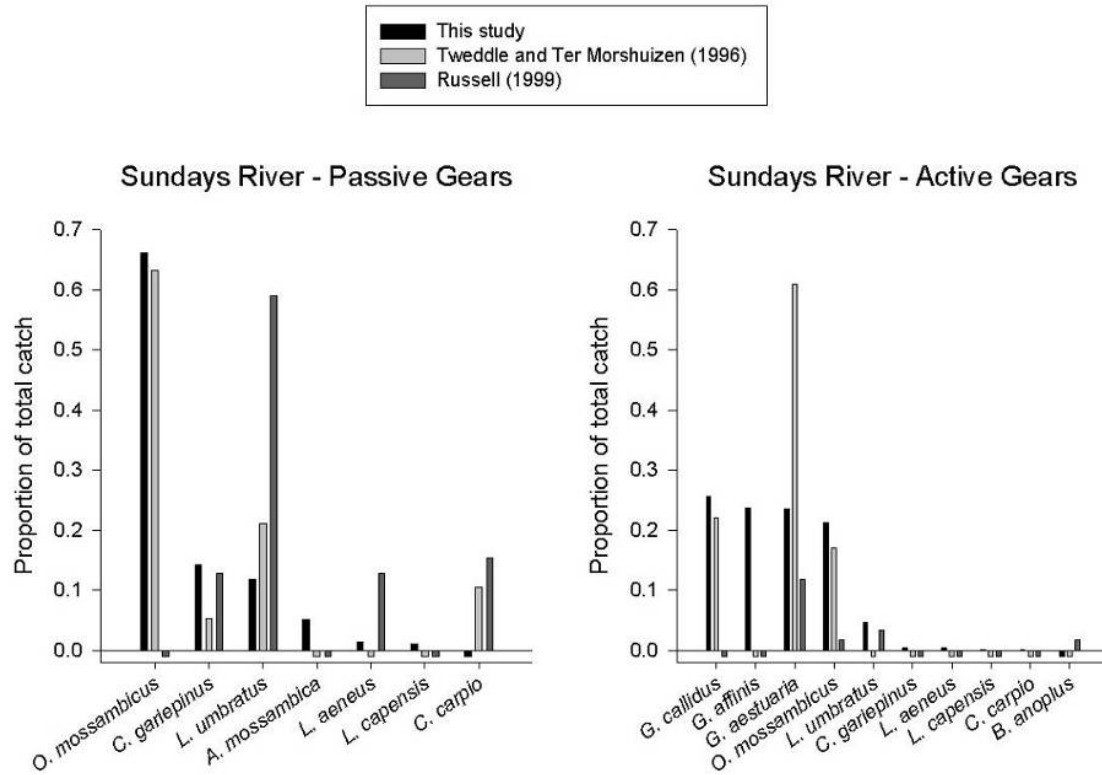


Figure 3.7. Relative abundance data from the Sundays River, including Darlington Dam. Negative values indicate species that were not sampled.

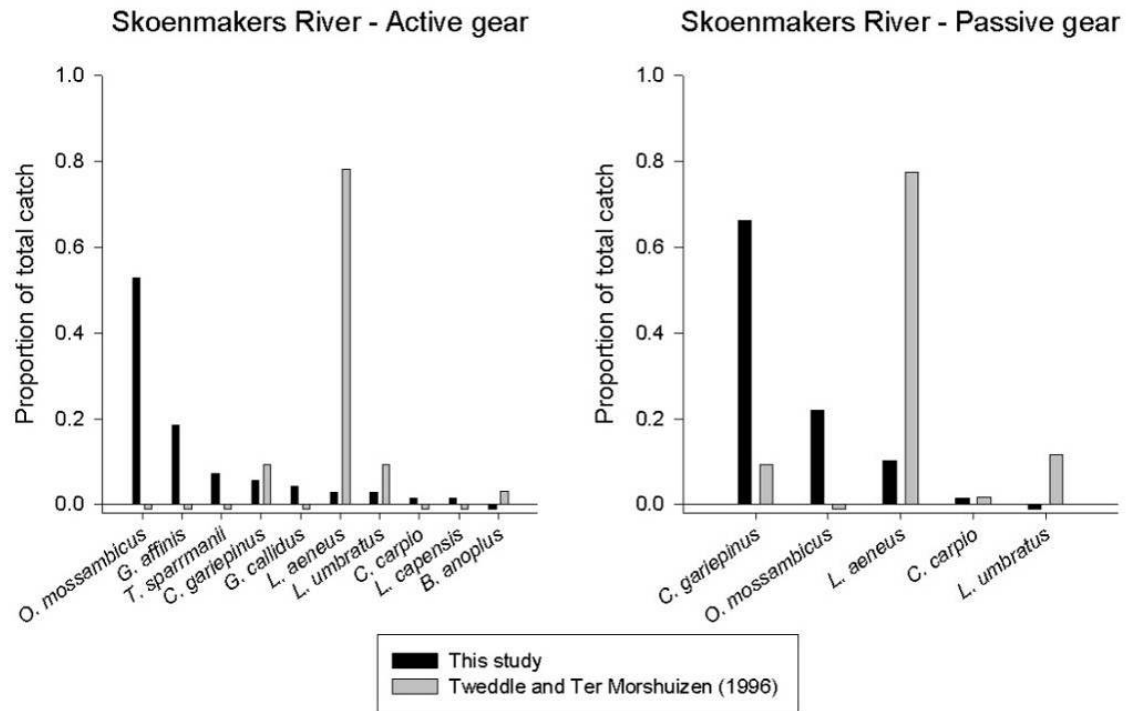


Figure 3.8. Relative abundance data from the Skoenmakers River. Negative values indicate species that were not sampled.

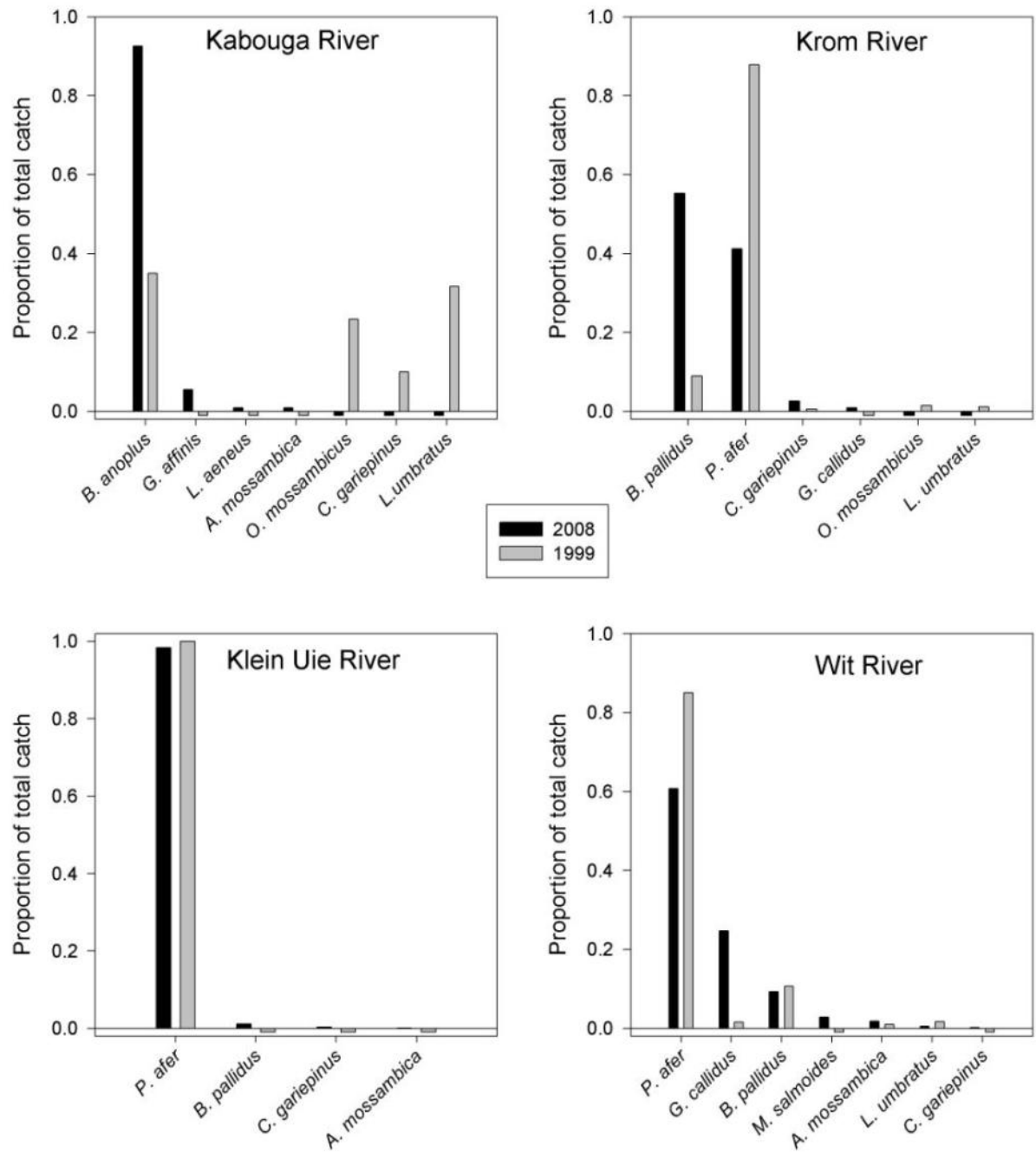


Figure 3.9. Relative abundance data from the tributaries of the Sundays River. Negative values indicate species that were not found. 2008 refers to data collected during this study and 1999 refers to data collected by Russell (1999).

3.4. Discussion

The continuous re-introduction of alien species into the Skoenmakers River via the Orange-Fish-Sundays water transfer scheme (Cambray and Jubb, 1977) has created source populations within Darlington Dam. Over time, fish recolonise areas downstream of the dam wall. From there, these alien species have the invasion potential to move into the Kabouga River, which is the only river where *B. anoplus* were sampled. From Korhaansdrift Dam, alien species move into the canal systems, and then have the potential to colonise other tributaries.

Indigenous fishes in the Sundays River, in general, appear to have decreased their distribution range, whereas alien species have generally increased their ranges. Most of the indigenous species are limited in range to the upper catchments of the tributaries, which implies that, if SANParks are intent on conserving a representative sample of the indigenous Sundays River ichthyofauna (SANParks, 2006), conservation of the upper catchments of the tributaries are critical to achieving their goal. Alien species are present in all the tributaries, with *G. affinis* and *L. aeneus* sampled from the Kabouga River, *C. gariepinus* from the Klein Uie, Uie, Wit and Krom rivers, and *M. salmoides* and *C. carpio* from the Wit River.

In the Sundays and Skoenmakers rivers, *O. mossambicus* was dominant. These rivers are highly altered due to the Orange-Fish-Sundays IBWT scheme. This altered state lends itself to invasibility and colonisation by alien species (Baltz and Moyle, 1993, Moyle and Light, 1996a, Moyle and Light, 1996b, Vitousek et al., 1997, Townsend, 2003). In areas where the streams remain unaltered, indigenous species dominate.

The presence of *A. mossambica* in Darlington Dam is an indication that the elvers of this species are capable of climbing over both the Korhaansdrift and Darlington Dam walls, the highest of which is 48 metres high. Dams are the leading cause of declines in the numbers of migratory fish (Angermeier, 1995). So, while the numbers of *A. mossambica* may have declined, these fish have not been extirpated from these environments due to the building of the dam walls. Similar conclusions can be made for the Slagboom Dam wall on the Wit River. Further barriers to the migration of *A. mossambica* can be found on the Klein Uie River, where a small weir in conjunction with a long section of steep smooth rocky cliffs exist. *Anguilla mossambica* are, however, able to overcome this barrier as two individuals were found above this structure.

The appearance of *G. affinis* in the Sundays River poses a threat to most of the ichthyofauna in the system, because of its potential predation on larval fishes. The most likely explanation for the appearance of *G. affinis* in the Sundays River is a rapid colonization from the Great Fish River, via the water transfer scheme, and a subsequent population growth. This species has been listed as one of the world's most damaging alien invasive species (Lowe et al., 2000). *Gambusia affinis* was capable of extirpating a topminnow (*Poeciliopsis occidentalis*) within one to three years of the introduction of *G. affinis* into the Santa Cruz System in Arizona (Meffe, 1984). This was entirely due to the predation of *G. affinis* on larvae and juveniles. The presence of *G. affinis* in the only remaining water of the Kabouga River has serious consequences for the health of the *B. anoplus* found there. *Gambusia affinis*, when found sympatrically with the black mudfish, *Neochanna diversus*, had negative impacts on both juvenile survival and juvenile growth rate (Ling and Willis, 2005). It is likely that *G. affinis* would have the same effect on *B. anoplus* and

since the Kabouga River was the only site where *B. anoplus* were found, the long term survival of *B. anoplus* is threatened.

The presence of *L. capensis* in the Skoenmakers and Sundays rivers poses a threat to the conservation of the genetic purity of the *L. umbratus* stocks in these same rivers. *L. capensis* readily hybridises with *L. umbratus* thus contaminating the genetic purity of the *L. umbratus* population (Cambray and Jubb, 1977). This hybridisation has already occurred in the Little Fish River, where *L. umbratus* is native and *L. capensis* is alien (Tweddle and Ter Morshuizen, 1996). Initial data from a genetic study on *L. umbratus* suggests that hybridization between *L. umbratus* and *L. capensis* has occurred in Darlington Dam (M. Ramoejane, MSc Student, Rhodes University, unpublished data).

Micropterus salmoides is restricted to the lower reaches of the Wit River, and has not moved upstream into the upper Wit River. Large numbers of juvenile *M. salmoides* were observed in the shallows of Slagboom Dam, and in some pools upstream of the dam in early February 2008, which indicates that this species is breeding successfully. The risk that a member of the public will physically move these fish upstream of the barriers is high, and will result in a similar situation to that found in the Blindekloof River in the Eastern Cape (Skelton, 2000). In the Blindekloof, *M. salmoides* were confined to a number of pools when the river stopped flowing. They would prey extensively on the minnow species in that pool until they had extirpated them. The bass would sustain themselves through cannibalisation, until the river flowed again, and they could move either upstream or downstream in search of new food sources. The result was that the pools had one or two large bass in them, instead of a community dominated by minnows.

There was insufficient data gathered during this study from the Kabouga River to make comparisons to past surveys, despite *B. anoplus* being the dominant species (Figure 3.9). Russell (1999) recorded the presence of *O. mossambicus*, *C. gariepinus* and *L. umbratus* in this river, but during this study these species were absent. Russell (1999) noted that the abundance of *B. anoplus* was similar to the abundance of *L. umbratus*. Since only one small pool of water was present in the Kabouga River, the species that were absent from this study may have been extirpated due to a lack of habitat space, and predation by otters.

To conclude, the source populations of alien species in Darlington Dam will continue to introduce fish into the Sundays River, and subsequently the tributaries. Barrier weirs are an important management tool for controlling the upstream movement of these alien species (Adams et al., 2001). These weirs should be constructed at suitable points along the tributaries to protect the integrity of the indigenous ichthyofauna of the GAENP.

Chapter 4. Spoon or scalpel? Species diversity indices as tools for dissecting community assemblage data

4.1. Introduction

The distribution of an organism within an aquatic system depends firstly on its natural biogeographical range, and secondly on its tolerance limits of abiotic factors such as temperature and oxygen concentration (Putman, 1994). Of species that are able to tolerate abiotic factors within a certain environment, some will be unable to establish viable populations due to the absence of suitable food organisms and the presence of superior competitors, predators or parasitic organisms (Putman, 1994). The interaction between biotic and abiotic factors determine the constituency of the community in question. In general, species richness (a count of the species present) has been shown to increase with increasing ecosystem size (King, 1964, Hill, 1973, Putman, 1994), while diversity (the relative number of species present) has been shown to increase with decreasing latitude (Hill, 1973, Putman, 1994).

Two factors have been proposed as constraints of environmental capacity for diversity. These are the productivity of the ecosystem and its spatial complexity (Putman, 1994). This trend is not an absolute certainty, as ecosystems with lower productivity can have a higher diversity than ecosystems with higher productivity (Putman, 1994). This is because low productivity environments limit the abundances of all species to the level where the species do not interact strongly with each other. But, with more resources available, certain species may dominate by out-competing other species that leads to a reduction in diversity (Putman, 1994). In addition, the more predictable an environment, the more diverse it is likely to be (Caswell, 1976).

From a freshwater fish perspective, the three most important factors determining species richness and diversity across global scales are; the size of the river system's drainage basin, the mean annual discharge of the river system and the primary productivity within the river system (Oberdorff et al., 1995). Oberdorff et al. (1995) also noted that biotic factors became more important within stable systems.

Species richness is one metric to characterize a community by simply counting the number of species present (Begon et al., 2006). A simple count of species is not always accurate since, invariably, through sampling, some species, especially rarer ones, are missed (Hill, 1973, Begon et al., 2006). In order to compare richness between samples, communities should be compared with the same selectivity. Some studies, when faced with this problem, have treated the sample as a complete census. This eliminates the need to consider variation and bias (Tokeshi, 1993). The most appropriate method to sample for species abundance patterns is therefore to sample randomly with the assumption that every individual in the community has an equal probability of being sampled (Tokeshi, 1993). Furthermore, species can easily be stochastically under- or over-represented during sampling. Rare species are particularly susceptible to under-representation.

If a simple count of species is used, a major characteristic of the community is omitted, that is, some species are rare while others are common. A measure of the evenness or equitability of the distribution of the individuals amongst species is required in addition to richness to quantify diversity properly (Putman, 1994). For example, consider a community with 10 individuals and 5

species. Let $|$ represent a species and i an individual. A community with a distribution of two individuals per species, $|ii|ii|ii|ii|ii$, is more diverse than a community where the community is dominated by one species, $|iiiiii|ii|i|i|i$, even though the species richness of the community is identical (Begon et al., 2006). Several diversity metrics are applied to terrestrial and aquatic communities. The most commonly applied being Simpson's and Shannon's indices. Attempts to classify a community with a single number has drawbacks as much of the detail about the community is lost (Begon et al., 2006). If a more descriptive analysis of the community is required, community assemblage models can be fitted to the data.

The objective of this chapter is to compare species diversity indices and community assemblage models, to determine which of the two were more suitable for use as community assemblage analysis in species depauperate environments.

4.2. Materials and methods

Diversity indices were applied to 48 individual sites and eight rivers, while, for the community assemblage analyses, 40 individual sites and three rivers were analysed. These sites were spread throughout the GAENP, and focused on the Skoenmakers, Sundays, Wit and Klein Uie rivers. Darlington Dam was also considered in the analyses, using data from both electrofishing and monthly gill-net samples.

Sampling was undertaken using gill nets, fyke nets, long lines, seine nets, electrofishing and angling. Individuals captured were separated into species, and measured. These measurements also acted as a count of all individuals captured in the sample. For more details on sampling procedure, refer to Chapter 2.

Species sampled with passive gear types were separated from species sampled with active gear types for the community assemblage analysis as it is considered unreasonable to compare species sampled with a passive, trap-like gear to species sampled with active targeting. Passive gears were defined as gears that required no human intervention or supervision, other than setting or retrieving of the gear. Active gears, under this definition, included angling, seine netting and electrofishing and passive gears included gill nets, fyke nets and long-lines.

Diversity indices

Two diversity indices were applied to the datasets. These were Simpsons' and Shannon's indices (Begon et al., 1996).

Simpson's diversity index takes into account both richness and evenness and is calculated by determining the proportion of individuals that each species contributes to the total number of individuals in the sample. The index is effectively the probability that two individuals picked at random from a community will belong to different species (DeJong, 1975). The index is unfortunately strongly affected by the abundance of the two or three most abundant species in the

community and does not take into account the rarer species (Hill, 1973, DeJong, 1975).

Shannon's diversity index, derived from information theory, has similar properties to Simpson's diversity index and it effectively quantifies the uncertainty that an individual, picked from an infinite population, belongs to a particular species. This index of diversity is strongly influenced by species richness (DeJong, 1975).

One important point about the diversity indices is that S , the total number of species, is intended to represent the number of species in the universe, but in biological studies that use this form of analysis, S is taken to represent the number of species that were sampled (Peet, 1975). As a consequence of this, the results from these studies consistently overestimate the evenness of the community, and that the differences in species number, because of the inherent bias of sampling, can affect the resulting value of the index (Peet, 1975).

Shannon's (H) and Simpson's diversity indices (D) were calculated as:

$$H = -\sum_{i=1}^S p_i \ln p_i \text{ and } D = \frac{1}{\sum_{i=1}^S p_i^2}$$

where p_i is the proportion of individuals for the i th species and S is the total number of species.

Simpson's evenness (J) and Shannon's evenness (E) were calculated as

$$J = \frac{H}{H_{\max}} = \frac{-\sum_{i=1}^S p_i \ln p_i}{\ln S} \text{ and } E = \frac{D}{D_{\max}} = \frac{1}{\sum_{i=1}^S p_i^2}, \text{ respectively.}$$

Species richness and evenness in the tributaries were compared to species richness and evenness in the Sundays River, Skoenmakers River and Darlington Dam using a Welch-modified two-sample t-test to determine under the null hypothesis whether richness and evenness were equal. The Welch-modified two-sample t-test was conducted due to heteroscedasticity between sites.

Community assemblage models

Community assemblage models, as with diversity indices, are abstractions of a complex community structure. The principal difference is that community assemblage models attempt to provide insight into the processes that structure communities (Begon et al., 2006).

Community assemblage models are categorised as being either biologically or statistically oriented (Tokeshi, 1993). Biologically oriented models can be further subdivided into niche oriented and non-niche oriented models. Statistically oriented models are usually applied to species rich communities that have vague spatial or temporal boundaries. In contrast, niche oriented models are applied to small communities containing related species that share the same resources.

In all likelihood, a community will not be governed by the rules of one community assemblage model. Tokeshi (1990) recognized this, and stated that the three most abundant species in the community would most likely work on some niche apportionment basis, while the rest would

most likely work on a random assortment basis of niche division. Four niche oriented, biological models were therefore considered for evaluation. There were the dominance pre-emption, random fraction, MacArthur fraction and random assortment models (Figure 4.1).

The *dominance pre-emption model* a single species' dominance over all other species in the environment is guaranteed (Tokeshi, 1990, Tokeshi, 1993, Fesl, 2002). The model therefore assumes that the first species takes a proportion, k , of the total niche, where k can take a value between 0.5 to 1. The second species takes a proportion, k , of the remaining niche, where, again, k can take a value of 0.5 to 1, and so on.

The *random fraction model* splits the total niche randomly amongst the species in the community (Tokeshi, 1990). The niche is divided into two fractions. One of the two sections is then chosen randomly, and divided into two. One of the three resulting fractions is then chosen at random and divided. In terms of a newly invading species, this model suggests that the invading species will invade the niche of a species taken at random and take a fraction of that niche (Tokeshi, 1993).

The *MacArthur fraction model* assumes that a newly invading species will take a portion of the niche occupied by the most abundant species (Tokeshi, 1990, Tokeshi, 1993). The model therefore divides each niche at random into two fractions. Then, one of the two resulting fractions is chosen probabilistically based on its length, and divided again at random. One of the three

fractions resulting from the previous division is chosen probabilistically based on length and divided again, and so on (Tokeshi, 1990).

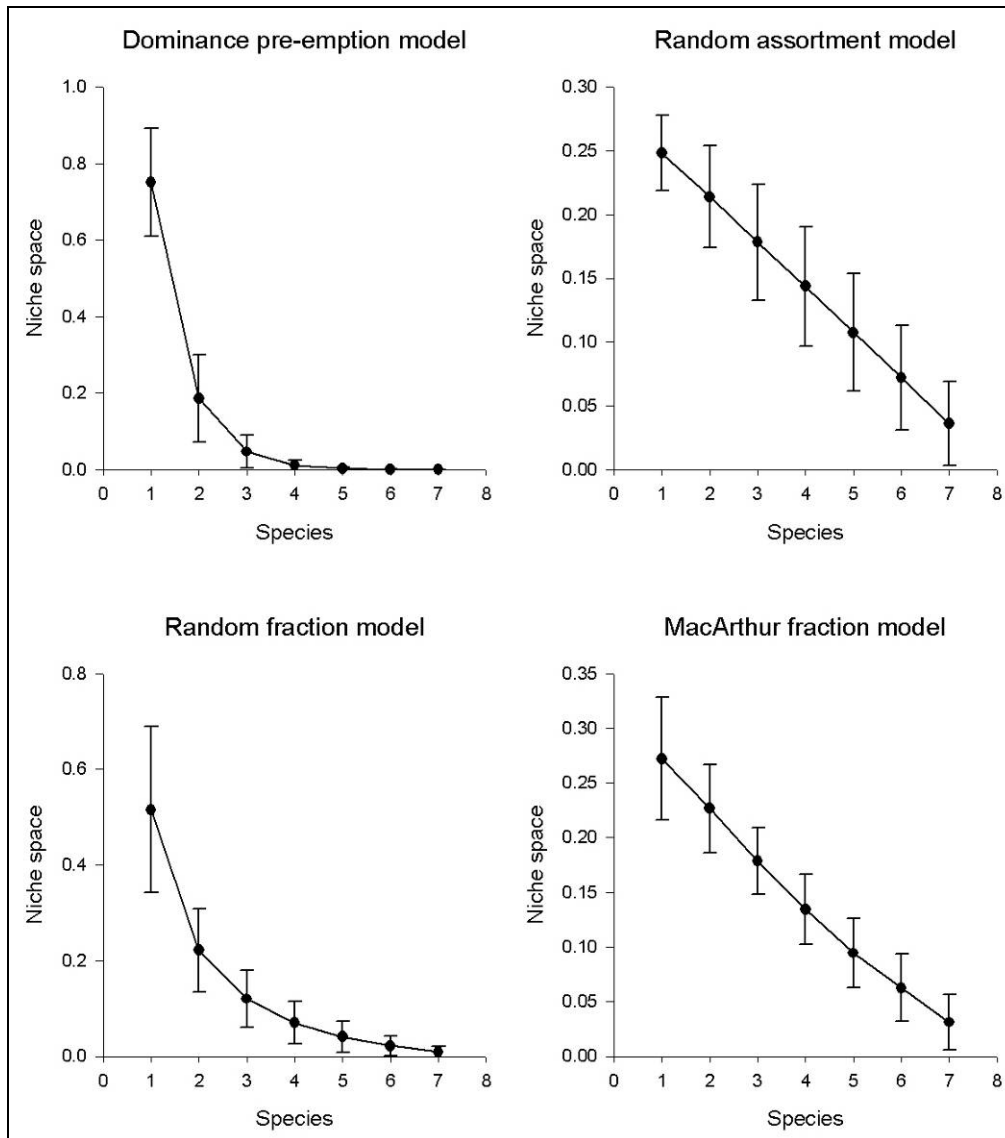


Figure 4.1. Community assemblage models for communities composed of 7 species, each showing different species compositions within the community

In the *random assortment model* abundances of the species in a community are independent of one another. This could be viewed either as non-correspondence of niche apportionment and species abundance, or as a dynamic niche division among species because of a highly variable environment (Tokeshi, 1990, Tokeshi, 1993). The variation inherent in such an environment leads to the total niche space changing constantly over time. In this situation, each species will take a portion of the niche independently of one another (Tokeshi, 1990). Because of the continual change in niche space, the niche is unlikely to be completely used at all times (Tokeshi, 1993, Fesl, 2002). Because of the variability of the environment, there is insufficient time for inter-specific competition to develop (Tokeshi, 1993).

Model implementation and interpretation

Each model was coded in the Sun Microsystems© Java 2 Standard Edition programming language and run on the J2SE virtual machine. Each model was run for a number of different communities, including two through seven, nine, ten and fifteen species communities. Each of these variations was run 1000 times, and the mean and standard deviation of niche space occupied at each level (i.e. most dominant, second most dominant, etc.) was obtained using the Apache Commons Math 1.2 API extension. Each model produces a different type of community structure (Figure 4.1) which can then be compared to observed data. Data from each site where two or more species were sampled was compared to each model using a least residual sum of squares approach. The model with the least residual sum of squares was selected as that model which most closely resembled the community under investigation.

This approach is unfortunately heavily influenced by subjectivity. Tokeshi (1990) suggested that, for niche oriented models, n (>10) random samples should be taken of the community being investigated. Mean abundance for each of the species encountered is then determined. Using a simulation approach, a large number (>1000) of theoretical communities of the same size as the samples are generated for each of the models to be tested. These simulations will have a mean and variance, which can then be used to calculate the confidence limits for each model. These theoretical values can then be compared with the observed means to determine the fit of the data. A problem with the above method is that the total number of species, S , must remain constant between samples. If this does not occur, Tokeshi (1993) recommends taking the minimum S from the observed data. He warns that when taking the minimum S , the total number of species encountered between samples should not vary too greatly. Tokeshi (1993) further states that the S taken for the simulation should account for at least 90% of the community in terms of abundance.

4.3. Results

Species diversity indices

If calculations from Shannon's evenness index yield a low value, dominance can be expected. A Welch-modified two-sample t-test determined that Shannon diversity index values were lower for the tributaries of the Sundays River than for the Sundays River itself ($df = 45.87$, $p < 0.01$). Shannon evenness values were also lower in the tributaries than in the Sundays River (Welch-modified two-sample t-test, $df = 45.464$, $p < 0.02$) (Table 4.1).

Simpson's diversity index showed a similar trend to that of Shannon's index in that species richness was significantly lower in the tributaries than in the Sundays River ($df = 34.305$, $p < 0.01$) (Table 4.1). Unlike Shannon's evenness index, Simpson's evenness index was not significantly different for tributary sites as opposed to sites from the Sundays River ($df = 26.004$, $p = 0.102$). A high evenness value for Simpson's index is an indicator of dominance.

When the sites were combined into their respective rivers, and the species diversity indices were calculated, the values showed similar trends to when the indices were calculated for each individual site with tributaries having lower diversity and evenness values. One exception was the Krom River that was considered species rich and relatively even (Table 4.2).

Table 4.1. Diversity indices values for 48 sites from the GAENP. H = Shannon's species richness index, J = Shannon's evenness index, D = Simpson's species richness index, E = Simpson's evenness index. Low values for Shannon evenness index and high values for Simpson's evenness index indicate dominance

Site	Watercourse	Species	Shannon		Simpson	
			H	J	D	E
GUIE-822-138	Groot Uie	2	0.13	0.15	1.06	1156.0
GUIE-871-185	Groot Uie	2	0.15	0.17	1.07	802.8
GUIE-904-890	Groot Uie	2	0.04	0.05	1.02	18906.3
GUIE-972-148	Groot Uie	2	0.25	0.26	1.14	218.4
KAB-351-765	Kabouga	4	0.32	0.18	1.16	306.9
KUIE-634-498	Klein Uie	2	0.36	0.36	1.26	74.0
KUIE-695-069	Klein Uie	2	0.03	0.04	1.01	33489.0
KUIE-814-495	Klein Uie	2	0.09	0.11	1.04	3025.0
KUIE-821-476	Klein Uie	2	0.21	0.23	1.11	348.4
KUIE-835-444	Klein Uie	2	0.64	0.53	1.80	9.0
KRO-167-761	Krom	2	0.33	0.33	1.22	100.0
KRO-310-710	Krom	2	0.31	0.31	1.20	121.0
KRO-550-520	Krom	2	0.11	0.12	1.05	1936.0
SKO-239-421	Skoenmakers	3	0.85	0.50	2.10	8.7
SKO-437-534	Skoenmakers	3	0.80	0.50	1.85	20.0
SKO-440-851	Skoenmakers	5	1.37	0.62	3.31	9.5
SKO-463-727	Skoenmakers	3	0.44	0.30	1.27	144.5
SKO-464-718	Skoenmakers	4	0.57	0.31	1.38	90.1
SKO-492-541	Skoenmakers	4	1.17	0.59	2.78	9.1
DAR-040-630	Sundays	5	1.43	0.67	4.01	5.8
DAR-349-348	Sundays	3	0.96	0.57	2.38	7.9
DAR-509-132	Sundays	3	0.89	0.53	2.14	10.5
DAR-676-130	Sundays	3	0.78	0.50	1.78	26.3
DAR-784-175	Sundays	2	0.54	0.49	1.55	18.8
DAR-919-092	Sundays	2	0.26	0.27	1.15	196.0
DAR-930-111	Sundays	3	1.02	0.62	2.61	7.5
DAR-988-115	Sundays	3	0.87	0.47	2.22	5.0
DAR-989-137	Sundays	4	0.77	0.39	1.77	18.4
KOR-566-362	Sundays	9	1.56	0.55	3.18	19.3
SUN-439-974	Sundays	3	1.05	0.63	2.71	7.0
SUN-502-601	Sundays	5	1.53	0.72	4.37	7.2
SUN-519-999	Sundays	6	0.97	0.36	2.22	6.9
SUN-904-001	Sundays	3	0.94	0.54	2.38	5.7
SUN-974-961	Sundays	6	0.67	0.29	1.40	148.0
UIE-747-545	Uie	2	0.56	0.50	1.60	16.0
WIT-027-032	Wit	3	0.63	0.39	1.57	26.0
WIT-213-958	Wit	2	0.18	0.20	1.09	529.0
WIT-285-612	Wit	2	0.50	0.46	1.47	25.0
WIT-317-443	Wit	4	1.32	0.69	3.59	6.0
WIT-790-321	Wit	3	0.90	0.54	2.17	10.0
WIT-798-435	Wit	3	1.06	0.63	2.78	5.0
WIT-803-591	Wit	2	0.45	0.43	1.39	36.0
WIT-855-123	Wit	2	0.35	0.35	1.25	81.0
WIT-858-812	Wit	3	0.92	0.53	2.28	7.1
WIT-884-925	Wit	2	0.09	0.11	1.04	2916.0
WIT-898-683	Wit	2	0.43	0.41	1.35	43.6
WIT-903-359	Wit	3	0.70	0.45	1.62	37.1
WIT-926-349	Wit	3	0.38	0.26	1.22	215.5

Table 4.2. Diversity indices values for the rivers of the GAENP. H = Shannon's species richness index, J = Shannon's evenness index, D = Simpson's species richness index, E = Simpson's evenness index. Low values for Shannon evenness index and high values for Simpson's evenness index indicate dominance

Site	Watercourse	Alien	Indigenous	Shannon		Simpson	
				H	J	D	E
Darlington	Sundays	6	4	1.38	0.64	3.32	11.2
Groot Uie	Uie	0	2	0.12	0.13	1.05	1610.0
Krom	Krom	1	3	0.83	0.36	2.10	5.9
Klein Uie	Uie	1	3	0.09	0.06	1.03	6488.5
Skoenmakers	Skoenmakers	7	2	1.55	0.54	3.64	7.1
Sundays	Sundays	5	4	1.78	0.62	5.01	9.1
Wit	Wit	2	5	1.08	0.40	2.28	14.2
Uie (incl. Groot & Klein)	Uie	1	3	0.13	0.07	1.05	2985.3

Community assemblage models

The Random assortment model appeared to be the most applicable to describe the Sundays and Skoenmakers river's communities, including Darlington Dam, and Korhaansdrift Dam (nine communities). The Dominance pre-emption model was the second most applicable model, with five communities being the most adequately described. The Random fraction model was similar to six of the Sundays River communities, and the MacArthur fraction model was similar to five of the Sundays River communities. All the communities along the Krom (three communities), Kabouga (one community), Groot Uie (four communities) and Klein Uie (four communities) rivers and most (six of seven) of the communities from the Wit River were similar to the Dominance pre-emption model. A single community along the Wit river was similar to the Random assortment model.

Table 4.3. Sum of squares values for community assemblage models as compared to data from each of the 40 sites considered for this analysis. Values highlighted in bold text indicate the model selected as most similar to the community at that sampling site. Gear types are passive (P) and active (A). DP = Dominance pre-emption model; RF = random fraction model; MF = MacArthur fraction model; RA = random assortment model

Site Number	Gear	Species	DP	RF	MF	RA
SUN-439-974	P	2	0.115	0.086	0.089	0.028
SUN-439-974	A	3	0.099	0.045	0.012	0.003
SUN-502-601	P	2	0.003	0.0002	0.0001	0.020
SUN-502-601	A	3	0.160	0.088	0.039	0.012
SUN-519-999	A	6	0.077	0.034	0.117	0.141
SUN-904-001	A	3	0.110	0.055	0.019	0.010
SUN-974-961	P	6	0.022	0.111	0.325	0.357
DAR-040-630	A	5	0.300	0.123	0.020	0.010
DAR-509-132	A	3	0.031	0.005	0.002	0.019
DAR-676-130	A	3	0.012	0.017	0.047	0.091
DAR-784-175	A	2	0.002	0.001	0.0003	0.022
DAR-919-092	A	2	0.044	0.062	0.060	0.140
DAR-930-111	A	3	0.094	0.040	0.010	0.001
DAR-988-115	A	3	0.123	0.069	0.033	0.025
DAR-989-137	A	4	0.002	0.015	0.092	0.136
KOR-566-362	P	6	0.055	0.001	0.066	0.083
KOR-566-362	A	4	0.016	0.001	0.046	0.077
SKO-239-421	A	3	0.041	0.013	0.006	0.021
SKO-437-534	A	3	0.005	0.003	0.025	0.063
SKO-440-851	A	5	0.178	0.053	0.005	0.010
SKO-463-727	P	3	0.037	0.083	0.155	0.235
SKO-464-718	P	2	0.029	0.042	0.040	0.109
SKO-464-718	A	2	0.029	0.015	0.016	0.000001
SKO-492-541	A	4	0.081	0.020	0.008	0.020
Darlington Gill-net data	P	7	0.031	0.008	0.126	0.146
KAB-351-765	A	3	0.056	0.116	0.200	0.293
KUIE-634-498	A	2	0.022	0.034	0.032	0.096
KUIE-695-069	A	2	0.092	0.117	0.113	0.218
KUIE-814-495	A	2	0.081	0.105	0.102	0.202
KUIE-821-476	A	2	0.056	0.075	0.072	0.159
GUIE-822-138	A	2	0.072	0.095	0.092	0.188
GUIE-871-185	A	2	0.068	0.090	0.087	0.180
GUIE-904-890	A	2	0.083	0.105	0.102	0.202
GUIE-972-148	A	2	0.047	0.064	0.062	0.144
WIT-027-032	A	3	0.001	0.019	0.059	0.114
WIT-213-958	A	2	0.062	0.083	0.080	0.171
WIT-317-443	P	3	0.139	0.073	0.028	0.008
WIT-884-925	A	2	0.081	0.104	0.101	0.201
WIT-898-683	A	2	0.011	0.018	0.017	0.068
WIT-903-359	A	3	0.008	0.024	0.066	0.120
WIT-926-349	A	3	0.045	0.097	0.174	0.260
KRO-167-761	A	2	0.030	0.043	0.041	0.111
KRO-310-710	A	2	0.034	0.049	0.047	0.120
KRO-550-520	A	2	0.078	0.100	0.097	0.196

When the data from all the sites were combined according to river and gear type, the Random fraction model (Figure 4.2) performed best, fitting the data from two rivers (Table 4.4.). Some data, for example from the Sundays River with active gear, did not fit any particular model well, but models were still chosen for these communities with the least sum of squares method (Figure 4.3).

Table 4.4. Sum of squares values for community assemblage models as compared to data from each of the three rivers considered for this analysis. Values highlighted in bold text indicate the model selected as most similar to the community in that river. Gear types are passive (P) and active (A). DP = Dominance pre-emption model; RF = random fraction model; MF = MacArthur fraction model; RA = random assortment model

River	Gear	Species	DP	RF	MF	RA
Sundays	P	7	0.038	0.006	0.116	0.136
Sundays	A	10	0.334	0.085	0.035	0.041
Klein Uie	A	4	0.093	0.200	0.393	0.473
Wit	P	3	0.043	0.013	0.006	0.018
Wit	A	4	0.018	0.001	0.042	0.073

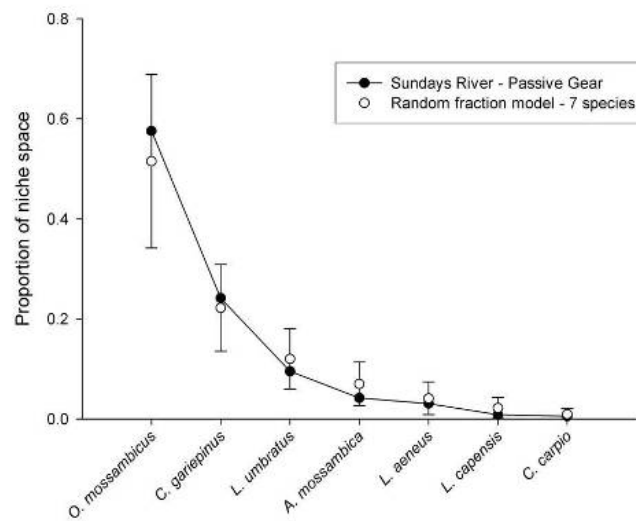


Figure 4.2. Data from species sampled with passive gear in the Sundays River plotted against the 7-species random fraction model, showing a good correlation between the data and the model.

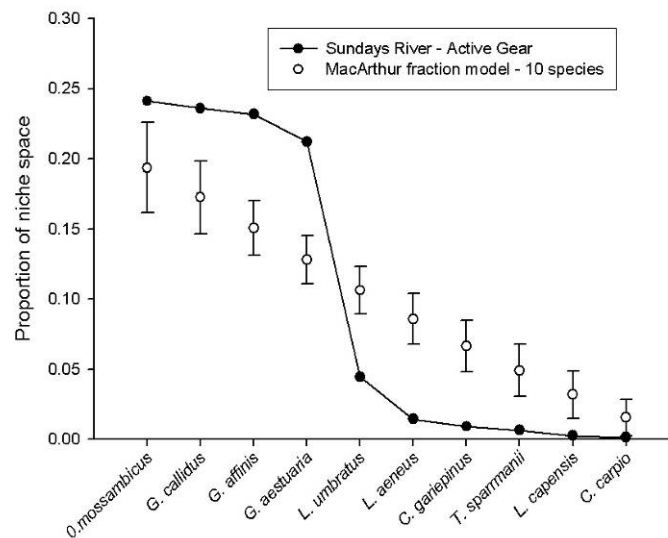


Figure 4.3. Data from species sampled with active gear in the Sundays River plotted against the 10-species MacArthur fraction model, showing the lack of correlation of the data to the model. No model fitted this particular data set well, although the proportional division model was selected based on a least sum of squares approach.

4.4. Discussion

Trends in the Sundays River and its tributaries were similar to those found in other river systems in that the main river had a higher species richness than its tributaries. For example, within western Virginia (Hitt and Angermeier, 2006), Illinois (Pegg and McClelland, 2004), Gabon (Ibanez et al., 2007), and New Zealand (Jowett and Richardson, 1996), large streams (that were at least third-order) had a greater species richness than the smaller first- and second-order streams . Species richness is generally higher in larger rivers due to the increased space, habitat complexity and food availability that these rivers provide (Angermeier and Schlosser, 1989).

Species compositions were more even in the Sundays and Skoenmakers rivers, as opposed to the tributaries where dominance by one species was prevalent. This finding concurred with other 1st and 2nd order streams in New Zealand (Jowett and Richardson, 1996) and the tributaries of the Rio Grande River (Platania, 1991).

Trophic level is not taken into account by any of the diversity indices employed. Cousins (1991) mentions that diversity indices would be more useful if data were separated by trophic level, such that piscivores are separated from detritivores, for example. In species depauperate environments, however, it would be difficult to achieve this, since there is seldom more than one species from each trophic guild, or as is the case in the Sundays tributaries, only one or two species occur naturally.

Hurlbert (1971) and Hill (1973) warn that no clear interpretation should be attached to a value calculated by a diversity index, nor should comparisons between such values be attempted as the distinction between abundance and importance is blurred. For instance, the complaint that the Simpson index is biased towards more common species is invalid (Hurlbert, 1971), since rare species, while contributing a small amount to the community, do not have major impacts on the environment or the community. It should be noted, however, that a single large predator in a community dominated by smaller prey species will have a major impact on the community.

In the Sundays River and its tributaries, the community assemblage models provided some insight into the structuring mechanisms that result in the observed communities. The biological description of the Random assortment model is that each species carves out a niche for itself, independently of other species in the community, because of the variability of the environment (Tokeshi, 1990). The data from sites along the Sundays and Skoenmakers rivers mostly fitted the Random assortment model. This is not unexpected since both of these rivers are in constant flux, due to water releases from the Great Fish River for the Skoenmakers River, and releases from Darlington Dam for the Sundays River. That these data fitted the Random assortment model suggests that there has been insufficient time for interspecific competition to develop in these environments.

The Dominance pre-emption model was the most successful model fitted to the different communities, fitting 25 of the 50 datasets (Table 4.3). Most of these sites had communities that consisted of only two species. All models except the Random assortment model had very similar

means and confidence intervals when run for two-species communities. It would be prudent to consider carefully fitting these models to communities containing two species, as it would be difficult to infer any reasonable conclusions, due to the difficulties associated with separating the different models at the two-species level.

In Darlington Dam, and along the Skoenmakers River, the Dominance pre-emption model was the most appropriate. The reason for this was the dominance of juvenile *O. mossambicus*. These juveniles may die off during winter due to low water temperatures (Skelton, 1993), leaving seasonally different community structures at these sites. Seasonal comparisons of these sites may yield some interesting inferences about community assemblage changes when juvenile fish are absent.

To conclude, species diversity indices provided insufficient detail about a community, as they did not provide any insights into the underlying processes of that community (Hurlbert, 1971, Hill, 1973, Peet, 1975). In addition, the diversity indices provided little insights into each community and were difficult to compare across sampling sites. These indices should, therefore, not be used for biological data because the information they provide is limited with regard to biological inference and comparability with other areas. In addition, the diversity indices can only be compared across sampling sites if the sample sizes are identical (Peet, 1975). The indices, if they are to be used, should be used in conjunction with other methods of community analysis, such as relative abundance plots or community assemblage models. As a biological indicator, Simpson's index appears to be the most useful of the three, but only if it is viewed as a measure of

interspecific contact, rather than interspecific competition. Community assemblage models, however, seem to be more useful in terms of gaining insights into the structure of a community. In a species depauperate environment, community assemblage models appear to function fairly accurately provided the community under examination consists of more than two species. When the community under examination has only two species, the means and confidence limits of the models are so similar that little distinction can be made between them owing to the lack of contrast.

Chapter 5. Density of indigenous species in stagnant pools from catch-effort data and depletion models

5.1. Introduction

Determining the absolute abundance of a species is often necessary when developing conservation strategies (Peterson et al. 2004). If population numbers are low, urgent action needs to be taken, while if population numbers are high, conservation focus can be shifted elsewhere. Acquiring a baseline population size is also important to conservation goals in terms of monitoring such that declines in absolute abundance can be observed and the appropriate action taken.

Depletion methods were first introduced in 1914 (Bishir and Lancia, 1996), and were later extended by Leslie and Davis (1939) and DeLury (1947) into a linear regression framework. Maximum likelihood approaches are now favoured due to advances in statistical theory and the availability of personal computing. Gould and Pollock (1997) later developed a maximum likelihood estimator based on a robust design that enabled the application of these methods to populations where recruitment, migration and mortality are not negligible.

Bishir and Lancia (1996) note that depletion methods are suitable for any situation in which the rate of detection of individuals is a function of the effort applied, and when population size changes based on the amount of effort expended. Depletion methods have most often been applied to commercially exploited fish populations (Bishir and Lancia, 1996) and have also been used with some success on small rivers and streams using multiple pass electrofishing (Seber and

LeCren, 1967, Bohlin and Sundstrom, 1977, Jowett and Richardson, 1996, Peterson et al., 2004).

As a sampling method, electrofishing is considered to be highly effective for assessing small rivers and streams (Bohlin and Sundstrom, 1977, Fievet et al., 1999).

To assess the status of the indigenous fish populations, multiple pass depletion electrofishing was conducted so that an estimate of population size could be determined. To compare populations from different pools, the densities of fish per square metre were calculated. These abundance and density estimates will provide suitable benchmarks for conservation management, against which future estimates can be compared, so that abundance fluctuations can be identified.

5.2. Materials and methods

A Samus[®] 725G backpack electrofisher, attached to a 12 V electric gate motor battery, was used for depletion electrofishing. The settings for the electrofisher were standardised at a duration of 0.3 ms and a frequency of 80 Hz. In shallow pools (< 70 cm deep), three-pass depletion electrofishing was conducted. Three buckets were filled with water and the anaesthetic 2-phenoxyethanol (at a concentration of 0.25 ml.l⁻¹) and placed on the bank of the pool to be sampled. The first pass was conducted and all fish captured during this pass were placed in the first bucket. The second pass was conducted in the opposite direction to the first, and all fish captured during this pass were placed in the second bucket. The third pass was then conducted in the same direction as the first pass, and all fish captured in this pass were placed in the third bucket. Catch-per-unit-effort, measured as fish·m² of water sampled, was then recorded for each pass.

At each site, the length and width of the pool were estimated by pacing. At each site, only a portion of the pool was sampled to minimise potential delayed mortality in that particular pool. The section of the pool that was electrofished was blocked from the rest of the pool with block-nets constructed from 50% shade netting. Captured fish from each electrofishing pass were separated into species, measured to the nearest millimetre in standard length, counted and released. There was less than 1% mortality from all sites sampled. Only *P. afer* data were used for the estimation, since the other indigenous species sampled, *B. pallidus* and *G. callidus*, were not sampled in sufficiently large numbers to apply depletion estimators.

Three-pass depletion electrofishing was attempted at 25 separate sites (Figure 5.1). On the Groot Uie River, six sites were sampled, while on the Klein Uie River, seven sites were sampled. On the Wit River, ten sites were sampled and on the Krom River, two sites were sampled. Only 21 sites produced data that could be incorporated into the depletion estimator. Data from one site on the Groot Uie River and three sites on the Wit River were discarded because no *P. afer* were sampled.

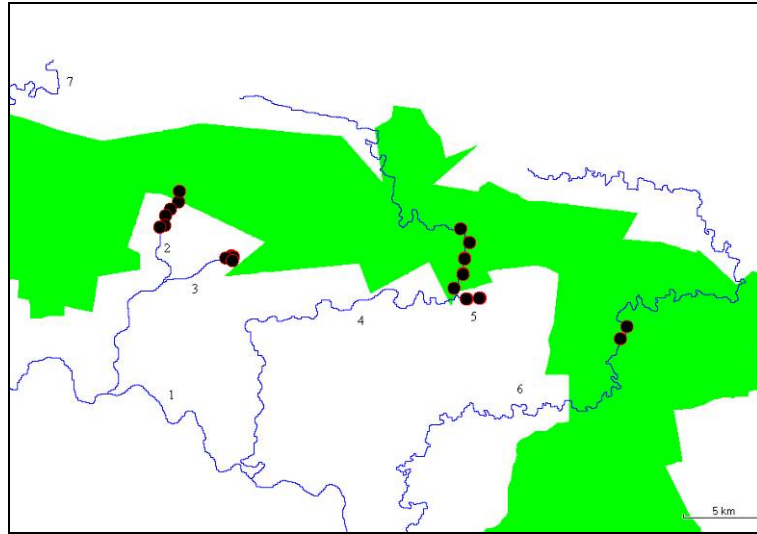


Figure 5. 1. Sampling sites used in three-pass depletion electrofishing experiments. 1 = Sundays River; 2 = Klein Uie River; 3 = Groot Uie River; 4 = Wit River; 5 = Ebersohn tributary, Wit River; 6 = Krom River; 7 = Kabouga River

Depletion models

The maximum likelihood approach proposed by Gould and Pollock (1997) was considered the most appropriate depletion estimator. This model has several assumptions that need to be satisfied before they can be applied.

As with other catch-effort methods, it is assumed that the number of individuals sampled is proportional to the amount of effort that is expended (Seber, 1973). This assumption implies that a single unit of sampling effort will capture a fixed proportion of the population, thus reducing the overall population and hence reducing catch-per-unit-effort. Other estimator assumptions are that the population is closed to recruitment, migration and mortality, and that the catchability

coefficient is assumed to be constant throughout the whole experiment, and the same for each individual such that all individuals have the same probability of capture (Seber, 1973).

As with other maximum likelihood estimators, the estimated parameter vector θ is estimated from some known data vector x by maximising a likelihood function of the form $L(\theta|x)$. In

Gould and Pollock's (1997) approach, $\theta = [N, k]$ and $x = [n, f]$, where N is the population size, k the catchability coefficient, x the vector of observed removals each corresponding to a vector f of sampling effort. Gould and Pollock's (1997) approach assumes that removals are multinomially distributed conditioned on the probability of catching a fish at least once. Specific likelihood details are found in Gould and Pollock (1997).

Gould and Pollock's (1997) model was modified to simultaneously estimate the population parameters for m populations such that the parameter vector is

$$\theta = \{x_1, x_2, \dots, x_m, k_1, k_2, \dots, k_m\}$$
 and the known data as

$$x = \{n_1, n_2, \dots, n_m, f_1, f_2, \dots, f_m\}.$$

Additional data were included into the estimator to test the hypotheses that 1) density and/or catchability was related to various physico-chemical variables, and 2) population size was related to the area fished.

In this modification, the probability of capture at pass i (p_i) was changed from

$$p_i = e^{-(kA)}$$

$$p_i = e^{-\left(k^j f_{ij}^j + A_j + T_j + pH_j + \mu S_j + DO_j + SAT_j + TDS_j + St_j + B_j\right)},$$

where f_{ij} is the effort expended in pool j during the i th sampling event, A_j is the percentage area fished of pool j , T_j is the water temperature of pool j , pH_j is the pH of pool j , μS_j is the conductivity of pool j , DO_j is the dissolved oxygen concentration in pool j , SAT_j is the oxygen saturation of pool j , TDS_j is the total dissolved solid concentration in pool j , St_j is the percentage of stone on the substrate of pool j and B_j is the percentage of boulders on the substrate of pool j .

The predictor variables (area, temperature, conductivity, pH, dissolved oxygen concentrations, oxygen saturation, total dissolved solids, percentage stones and percentage boulders) were stabilised using a z-transformation. Eleven different models were calculated (Table 5.1).

Akaike's information criterion (AIC) was calculated for each model to determine the most parsimonious model. Likelihood ratio tests were conducted between the reduced model and each of the other models to determine which of the parameters, if any, significantly affected the estimates of absolute abundance and catchability.

Table 5.1. Parameter combinations used in the Gould and Pollock (1997) depletion estimator calculated with depletion electrofishing data from four rivers in the GAENP. μS = conductivity; DO = dissolved oxygen concentrations; % O₂ Sat = oxygen saturation; TDS = total dissolved solids; Stones = percentage of stones in the substrate of the sampling site; Boulder = percentage of boulders in the substrate of the sampling site

Variation	Area	Temp	pH	μS	DO	% O ₂ Sat	TDS	Stones	Boulders
Null	-	-	-	-	-	-	-	-	-
Full	x	x	x	x	x	x	x	x	x
Area	x	-	-	-	-	-	-	-	-
Temp	-	x	-	-	-	-	-	-	-
pH	-	-	x	-	-	-	-	-	-
μS	-	-	-	x	-	-	-	-	-
DO	-	-	-	-	x	-	-	-	-
% O ₂ Sat	-	-	-	-	-	x	-	-	-
TDS	-	-	-	-	-	-	x	-	-
Stone	-	-	-	-	-	-	-	x	-
Boulder	-	-	-	-	-	-	-	-	x

Density calculations

Density was calculated as fish·m². In the calculation of fish·m², the Gould and Pollock (1997) MLE estimate of N was used as the numerator and the area of the water sampled was used as the denominator for all sites.

5.3. Results

No discernable trends of density were observed when the sites were viewed from the most upstream site through to the most downstream site (Figure 5.2).

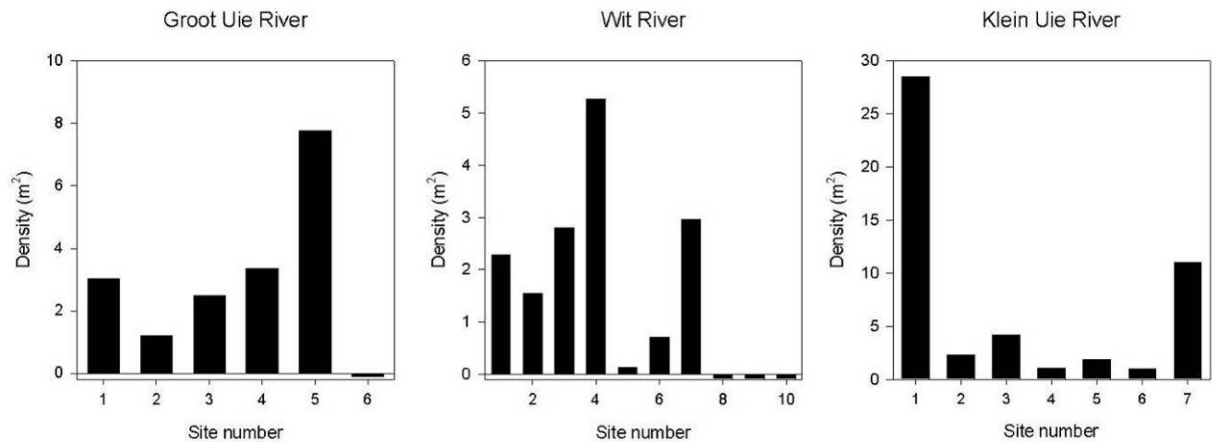


Figure 5.2. Densities of *Pseudobarbus afer* along a longitudinal gradient, indicating no discernable trend in the change of densities of fish from upstream to downstream environments. 1 refers to the highest sampling site on each respective river, 2 refers to the next most downstream sampling site, and so on. Negative values indicate sites where *Pseudobarbus afer* was absent

AIC and Likelihood ratio tests

The calculation of AIC for each of the variations indicated that the most parsimonious model included the percentage of boulders on the substrate of the pool (Table 5.3). The likelihood ratio test confirmed this result, with the only significant result coming from comparison between the ‘null’ model and the reduced ‘boulder’ model ($p = 0.026$). These results indicated that the presence of boulders on the substrate was associated with an increase in *P. afer* abundance.

Table 5.2. Information from 21 sites along four rivers where depletion electrofishing could be conducted.

Site	River	Pool size (Fished area)	Total catch	Abundance (N)	Catchability (k)	Density.m ²
GUIE-822-138	Groot Uie	250 (25)	66	77	0.51	3.07
GUIE-844-188	Groot Uie	350 (70)	81	84	0.64	1.21
GUIE-871-185	Groot Uie	40 (40)	82	104	0.43	2.59
GUIE-972-148	Groot Uie	50 (50)	120	181	0.34	3.62
GUIE-904-890	Groot Uie	80 (40)	267	311	0.48	7.76
KUIE-024-033	Klein Uie	210 (70)	140	164	0.47	2.34
KUIE-280-710	Klein Uie	75 (25)	61	67	0.53	2.70
KUIE-498-500	Klein Uie	800 (100)	100	104	0.65	1.04
KUIE-695-069	Klein Uie	7 (7)	182	200	0.56	28.58
KUIE-814-495	Klein Uie	30 (30)	54	56	0.69	1.87
KUIE-821-476	Klein Uie	60 (60)	49	61	0.47	1.01
KRO-167-761	Krom	30 (15)	45	50	0.85	3.33
KRO-550-520	Krom	28 (28)	44	49	0.51	1.76
WIT-213-958	Wit	6 (6)	22	31	0.36	5.15
WIT-221-898	Wit	28 (28)	75	75	0.83	2.68
WIT-225-944	Wit	90 (30)	3	3	0.60	0.10
WIT-228-920	Wit	84 (28)	17	20	0.46	0.71
WIT-903-359	Wit	38 (38)	57	74	0.77	1.98
WIT-884-925	Wit	63 (63)	54	61	0.50	0.97
WIT-898-683	Wit	90 (28)	32	42	0.57	2.80
WIT-926-349	Wit	200 (40)	80	119	0.34	2.98

Table 5.3. Statistics from AIC and likelihood ratio tests on 11 variations of the Gould and Pollock (1997) depletion estimator. lnL = log likelihood; df = degrees of freedom; AIC = Akaike's information criterion

Variation	lnL	df	AIC
Null	159.56	44	407.12
Full	157.60	62	439.19
Area	159.54	46	411.08
Temp	159.10	46	410.21
pH	156.95	46	405.90
µS	159.21	46	410.42
DO	159.42	46	410.83
% O ₂ Sat	159.33	46	410.65
TDS	159.04	46	410.08
Stone	159.46	46	410.92
Boulder	152.26	46	396.53

5.4. Discussion

For depletion estimators to be suitably applied, the estimator's assumptions need to be satisfied. The assumption that the population is closed can be explained with the method used to capture the fish. Block nets prevented the movement of fish into and out of the sampled area, when the sampled area was smaller than the size of the pool and thus there was no migration. Each multiple pass electrofishing event took less than an hour to complete, so recruitment was not possible. No natural mortality was observed, other than in a minority of sites along the Wit River, where crabs attacked and killed some of the stunned fish that were not collected during previous electrofisher passes. The assumption that all individuals have the same probability of capture can be violated by size-selectivity issues. Size-selectivity can affect the result of a depletion

experiment when the sampling method used is electrofishing , because, in general, larger fish are more susceptible to electrofishing than small fish (Mahon, 1980). If this trend is observed during a depletion experiment, the assumption that all individuals have the same probability of capture is violated. Mahon (1980) notes that size selectivity will present itself by a decrease in mean length over successive electrofishing passes. Mean lengths between the three passes during these experiments did not differ significantly (ANOVA: $F = 12.51$, $p > 0.05$), with the first pass' mean length being 31.61 ± 11.49 mm SL, the second pass 30.90 ± 11.32 mm SL, and the final pass 30.57 ± 10.02 mm SL, thus size-selectivity did not affect the results of these experiments.

Violation of the assumption regarding constant catchability can alter the result of the depletion experiment. A decrease in catchability over time will cause an overestimation of N , while an increase in catchability over time will result in an underestimation of N . There is no reason to believe that a change in catchability occurred during the electrofishing experiments, since the fish appeared to respond in the same way with each pass.

When used with multiple pass electrofishing as a sampling method, depletion methods have been shown to overestimate k by about 39%, and underestimate N by 88% (Mahon, 1980, Peterson et al., 2004). These biases stem from fish behaviour, stream characteristics, fish species and fish size (Peterson et al., 2004). Given that density is a standardised measure of abundance, it is a comparable metric across rivers in the same region and between pools in the same river. Therefore, if the underestimation fraction is constant throughout the sampling region, direct comparisons can still be made, if the objective of the comparison is simply to determine which river or section of river is more favourable for the species under investigation. For conservation

purposes, an underestimate of N is preferable to an overestimate of N , since a more conservative result will be more forgiving if management errors are made during the conservation process.

The positive relationship between absolute abundance and the presence of boulders can be explained by an increasing complexity of habitat with an increasing percentage of boulders in the pool. With a more complex habitat, the surface area of the substrate increases, thus providing more refuge, and an increased grazing area for the invertebrates on which *P. afer* feed.

In conclusion, multiple pass electrofishing is a suitable technique for determining the absolute abundance, catchability and density of fish in the GAENP. As a monitoring tool, multiple pass electrofishing appears to be an effective sampling strategy to determine the trends of absolute abundance over time.

Chapter 6: Movement of alien fish in the Greater Addo Elephant National Park

6.1. Introduction

An estimated 480 000 alien species have been introduced into a variety of different ecosystems globally (Pimentel et al., 2001). Although some of these species such as livestock and staple food crops are economically beneficial, others have caused major economic and ecological damage.

The control and eradication of these species costs an estimated US\$ 1.4 trillion per annum (Pimentel et al., 2001). The USA alone spends approximately US\$ 5.4 billion on the control and eradication of alien fish species (Pimentel et al., 2005). An estimated 160 fish species have been introduced into 120 different countries (Townsend and Winterbourn, 1992). The main motivations for these introductions are to promote recreational angling, to promote and enhance fisheries, to promote aquaculture and for aesthetic purposes (Pascual *et. al.*, 2002).

A total of 65 fish species have been introduced or translocated within southern Africa (Shumway, 1999). These fish have been responsible for the decline of at least 11 species of fish in South Africa and threaten a further 60% of endemic species (Pimentel et al., 2001). Despite this problem, the vast majority of introductions have not been investigated and the impacts of these alien species are unquantified (Townsend and Winterbourn, 1992). A total of eight alien species are present in the Sundays River System (Table 6.1). Four of these species are on the “100 of the world’s worst invasive species” list, namely *C. carpio*, *O. mossambicus*, *G. affinis* and *M. salmoides* (Lowe et al., 2000), and a fifth, *Clarias gariepinus*, is closely related to another listed species, *Clarias batrachus*.

Table 6.1. List of alien species present in the Sundays River System. * = listed as one of the world's worst alien species (Lowe et. al., 2000). I = introduced; T = translocated; ? = method of introduction unknown.

Species	Status
<i>Clarias gariepinus</i>	T
<i>Cyprinus carpio</i> *	I;T
<i>Gambusia affinis</i> *	T
<i>Labeo capensis</i>	T
<i>Labeobarbus aeneus</i>	T
<i>Micropterus salmoides</i> *	I
<i>Oreochromis mossambicus</i> *	?
<i>Tilapia sparrmanii</i>	?

The aim of this chapter is to examine the potential for the movement of alien species within the Sundays River System, and to determine what effects these fish would have should they penetrate critical conservation areas.

6.2. Materials and methods

General materials and methods for sampling the ichthyofauna of the Sundays River and its tributaries are described in Chapter 2. In addition, each river in the GAENP was examined by a fishway expert (Dr. A. Bok, Anton Bok Aquatic Consultants) to determine the presence of potential barriers to the upstream migration of alien species.

On the Wit River, the species composition upstream of the Upper Wit River (above the tail-end of Slagboom Dam), where alien species were absent, was compared to the species composition of the Lower Wit River (Slagboom dam and below), where alien species were present, using contingency table analysis.

6.3. Results

Alien fish were found to be widespread throughout the Sundays River system, and prolific in the Sundays River itself (see Chapter 3 for more details). Of the 3313 fish sampled, 723 were alien to the Sundays River system. This equates to 21.8% of the total catch. *Clarias gariepinus* was the most successful alien fish species in terms of the extent of their distribution, having penetrated the Kabouga, Klein Uie, Uie, Wit and Krom rivers. Alien fish were sampled in all tributaries, except the Groot Uie River (Figure 6.1).

A total of nine barriers to the upstream migration of fish were identified along the Sundays River (Figure 6.2), all of which were man-made in the form of two weirs (Figure 6.3), one dam wall (Figure 6.4) and six causeways (Figure 6.5). No natural barriers to upstream migration were found (Bok, 2008). On the Kabouga, Groot Uie and Klein Uie rivers, no man-made barriers were present, but a natural barrier in the form of a sloping bedrock shelf in a narrow gorge was found to be preventing the upstream movement of *C. gariepinus* under low-flow conditions (Figure 6.6). In flood conditions, however, fish will probably be able to negotiate this barrier (Bok, 2008). On the Wit River, the 10m high Slagboom Dam wall (Fig 6.7) is considered a complete barrier to migration and has effectively stopped *C. gariepinus* from extending its range (Bok,

2008). On the Krom River, there were two partial barriers to fish movement in the form of causeways (Bok, 2008).

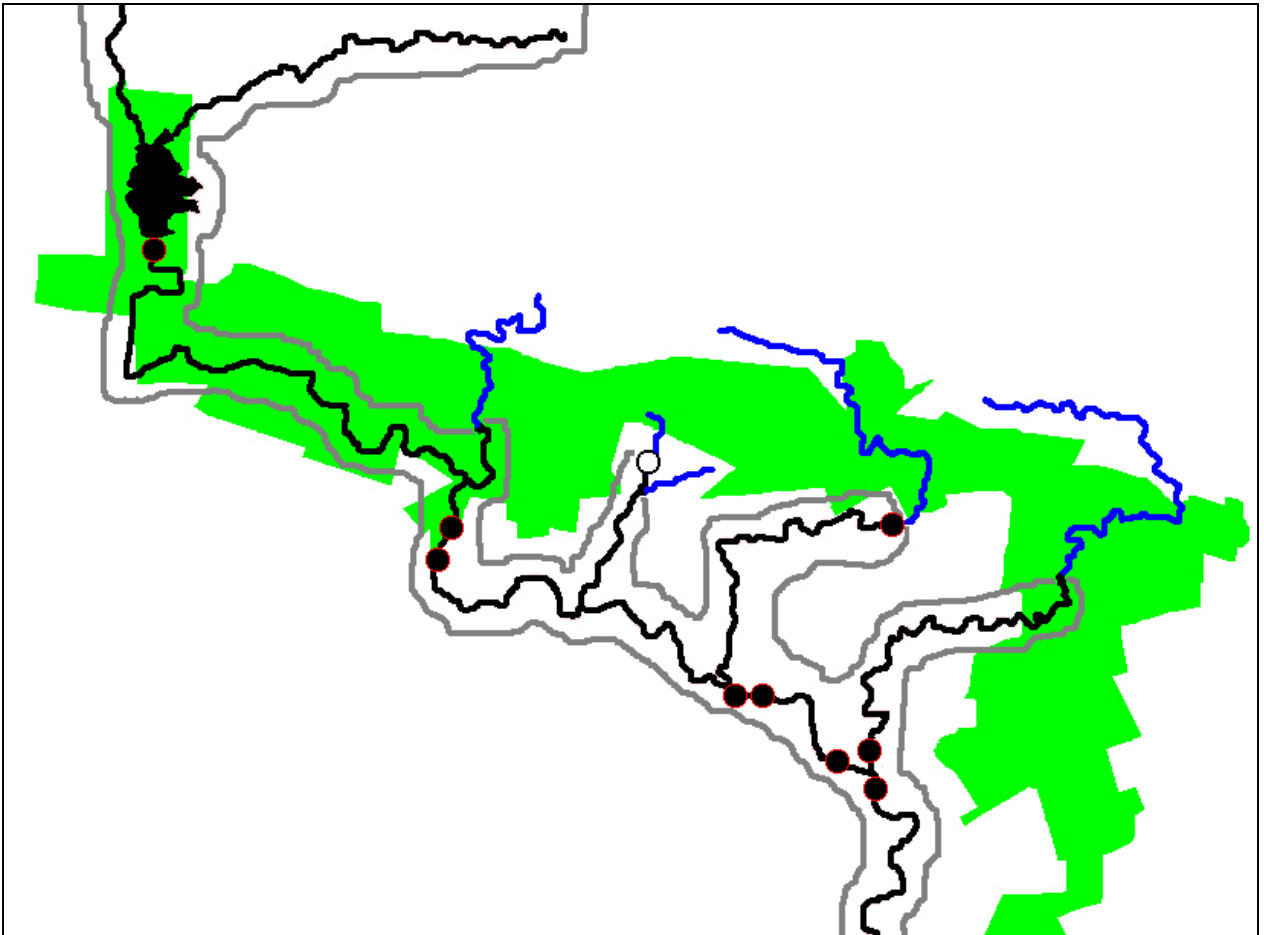


Figure 6.1. Extent of invasion by alien fish species within the Sundays River system. Areas inside the grey line indicate the areas where the presence of alien fish species was detected. Open circles denote natural barriers to upstream fish migration, closed circles denote man-made barriers to upstream fish migration.

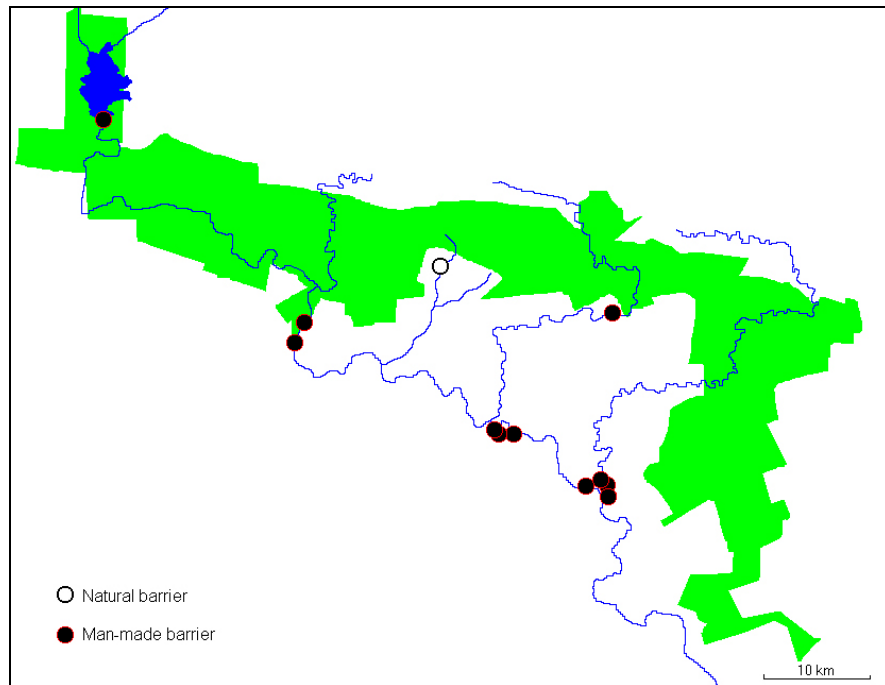


Figure 6.2. Distribution of natural and man-made barriers to fish migration in the Sundays River system.



Figure 6.3. Cleveland weir (left) and Korhaansdrift Dam (right) on the Sundays River (photographs courtesy of Olaf Weyl)



Figure 6.4. The 48 m high Darlington Dam wall, with 5 spillways (left) and six main outlet valves (right).
(right photograph courtesy of Olaf Weyl)



Figure 6.5. Two of the six causeways on the Sundays River that are partial barriers to the upstream migration of fish (photographs courtesy of Olaf Weyl)



Figure 6.6. Natural barrier of sloping bedrock limiting upstream fish movement on the Klein Uie River



Figure 6.7. Slagboom Dam wall on the Wit River, a partial barrier to upstream eel migration, and a complete barrier to the upstream migration of other fish species

Potential impacts of alien fish

The species composition of the Upper Wit River was found to be significantly different to the species composition of the Lower Wit River ($\chi^2 = 105.49$; $df = 2$; $p < 0.00001$). The Lower Wit River was populated by *M. salmoides*, whereas it was absent in the Upper Wit River. The differences could be attributed to the differential densities *P. afer*. There was a drastic reduction in the average density of *P. afer* from the Upper Wit River (1.2 ± 1.6 fish·m²) to the Lower Wit River (0 fish·m²). Similarly, *B. pallidus* was present where *M. salmoides* was absent, but absent where *M. salmoides* was present. In contrast, the density of *G. callidus* was not affected by the presence of *M. salmoides* (Figure 6.8). On the Klein Uie River, the average density (6.5 ± 10.9 fish·m²) of indigenous species above the presence of *C. gariepinus* was not significantly different to the average density (7.0 ± 7.8 fish·m²) of indigenous species below the presence of *C. gariepinus*. *P. afer* were found to co-exist with *C. gariepinus*.

On average, the population size for *P. afer* was 147 (± 97) fish per pool, 161 (± 133) fish per pool and 56 (± 31) fish per pool for the Groot Uie, Klein Uie and Wit rivers respectively (see Chapter 5).

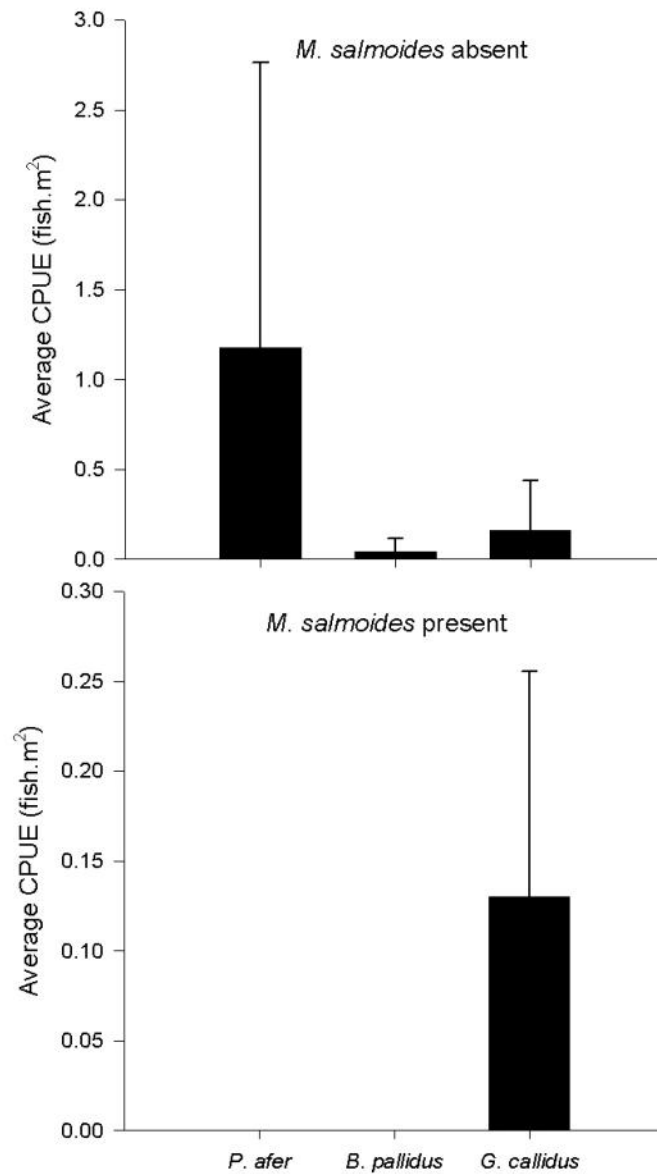


Figure 6.8. Average density in fish·m², with standard deviation, of small indigenous fishes from the Wit River separated by areas with *Micropterus salmoides* present and *Micropterus salmoides* absent.

6.4. Discussion

Records indicate that *C. carpio*, *M. salmoides* and *C. gariepinus* were introduced for angling purposes (Cambray and Jubb, 1977). Three of the other five alien species, *L. capensis*, *L. aeneus* and *G. affinis*, have most probably been passively translocated from the Orange and Great Fish rivers via the Orange-Fish-Sundays interbasin water transfer scheme (Cambray and Jubb, 1977). *Clarias gariepinus* populations are augmented by this IBWT scheme (Cambray and Jubb, 1977). The method of introduction of *T. sparrmanii* and *O. mossambicus* is unknown.

One of the major problems facing the eradication or control of alien species in the Sundays River system is that five of the eight alien species are being continually introduced into the catchment through the Orange-Fish-Sundays IBWT scheme. The populations of alien species therefore cannot be eradicated because of this continual recruitment from a source population. Control of these species through barrier weirs is also unlikely to be successful, since the invasion in the Sundays River is a downstream process, and barrier weirs are only effective when invasions are upstream processes (Adams et al., 2001). It is, therefore, recommended that no eradication or control measures are implemented on Darlington Dam or the Sundays River. Rather, the presence of certain alien species, such as *C. gariepinus*, *L. aeneus* and *C. carpio*, could be viewed as an opportunity to generate revenue for SANParks through the sale of fishing licences.

The lack of barriers to the upstream movement of fish in the critical conservation areas is a weakness in terms of the control of the alien species that are present in these areas. Since *C. gariepinus* is a potamodromous species, it will move upstream during high-flow conditions

(Gaigher, 1977, Clay, 1979). It is also present in all but one of the five tributaries. *Clarias gariepinus* is contained by barriers on the Wit River (Slagboom Dam wall) and on the Klein Uie River (sloping bedrock shelf). On the other tributaries where it is present, no barriers to its movements were found, implying that this species has complete freedom of movement. Bok (2008) notes that during high-flow conditions, *C. gariepinus* may be able to negotiate the barrier on the Klein Uie River by moving along the river margins, and penetrate the pristine *P. afer* habitat above this barrier. If *C. gariepinus* were to penetrate this environment, the populations of *P. afer* would be threatened (de Moor and Bruton, 1988). On the Wit River, illegal stocking of *C. gariepinus* into Slagboom Dam by anglers is entirely plausible and indeed likely, given that there is a fair amount of fishing effort on the dam and *C. gariepinus* are an angling species.

The decline of the *P. afer* population in the Wit River is similar to trends found in other parts of the world where *M. salmoides* have been introduced. In the western United States, the introduction of *M. salmoides* resulted in the decline of several minnow and pupfish species (Iguchi et al., 2004). In Guatemala, *M. salmoides* caused the local extinction of several fish species, and the complete extinction of an endemic waterbird, the Atitlan grebe *Podilymbus gigas* in Lake Atitlan (Shumway, 1999, Iguchi et al., 2004).

In Zimbabwe, *M. salmoides* has reduced the populations of small barb species where they co-exist, to such an extent that some populations have become locally extinct (Gratwicke & Marshall 2001). Gratwicke & Marshall (2001) found that when *M. salmoides* was absent, barbs were the most numerous in all the habitat types in which they sampled, but barbs were absent from 62% of the sites where *M. salmoides* was present. This is probably due to the body shape, colour and

behaviour of the barbs. Because barbs are elongate, *M. salmoides* selectively preys on these fish as they are easier to swallow (Gratwicke & Marshall 2001). In addition, the silvery colour of the barbs may attract predatory fish, especially since they tend to shoal in mid-water (Gratwicke & Marshall 2001). The vulnerability of these cyprinid fishes may be due to their evolution in the absence of predators (Godinho and Ferreira, 2000). The potential impact of *M. salmoides* can be illustrated using consumption data obtained from the literature. (Cochran and Adelman, 1982) suggest that a single 180 gram *M. salmoides*, on a maintenance diet of 0.45% of its body weight daily, would extirpate an average *P. afer* population on the Groot Uie, Klein Uie and Wit rivers in 75, 82 and 29 days respectively. This same fish, on a maximum consumption diet of 5.8% of its body weight daily, would extirpate the average *P. afer* population in the Groot Uie, Klein Uie and Wit rivers in 7, 7 and 3 days respectively.

In South Africa, the impacts of *M. salmoides* have been incompletely documented. In three examples provided by de Moor & Bruton (1988), *M. salmoides* have been implicated in the decline of indigenous ichthyofauna, but no quantitative data has been presented. In the Olifants River, on the Clanwilliam system, *M. salmoides* is thought to have been responsible for the decline in the populations of ten minnow species, while in the Swartkops river, they have been implicated in the decline of *P. afer* populations (de Moor & Bruton 1988). In the seasonal Blindekloof River, a tributary of the Swartkops River, Skelton (2000) found that, instead of a large population of minnow species, one or two large *M. salmoides* would occupy each pool. These individuals would then move further upstream when the Blindekloof River was in flow in search of new food sources (Skelton 2000).

The apparent lack of impact of *M. salmoides* on the *G. callidus* populations can be partially explained by *G. callidus*' coloration and behaviour. *G. callidus* is a cryptic, demersal species that hides amongst rocks and in vegetation (Skelton 2001). In addition, *G. callidus* have co-evolved with indigenous predators and their reproductive strategy may be suited to withstanding predation pressure. Townsend & Winterbourn (1992) found a similar trend in New Zealand streams. Introduced rainbow trout, *Oncorhynchus mykiss*, selectively preyed on koara, *Galaxias brevipinnis*, reducing their populations, while smelt, *Retropinna retropinna*, withstood this predation through their ability to hide in the substrate (Townsend & Winterbourn 1992).

The eradication of an established introduced species is both costly and impractical (Cambray 2003) and in many cases, complete eradication is unsuccessful, especially in larger systems (Pimentel *et. al.*, 2001; Lintermans 2004; Neilson *et. al.*, 2004). Several projects in Australia and New Zealand have used rotenone, a piscicide, to successfully eradicate alien species, such as *Gambusia* spp., common carp and rudd. An advantage with using rotenone is that complete eradication is achieved (Neilson *et. al.*, 2004). However, eradication projects making use of rotenone could unintentionally eradicate the indigenous ichthyofauna especially where they are confined to small rivers and have narrow distribution ranges (Cambray 2003).

In the case of the Wit River, eradication of *M. salmoides* is not feasible due to the size of Slagboom Dam, and the high likelihood of anglers re-introducing the species into the impoundment. For this reason, containment in the form of barrier weirs should be constructed in order to reduce the likelihood of *M. salmoides* migrating upstream during a period of high flow. The natural barrier on the Klein Uie River should be strengthened by the erection of a barrier

weir to constrain the upstream movement of *C. gariepinus*. The construction of barrier weirs have been successful for containing several alien species, including trout (Lintermans 2004).

There is a strong need for public education and awareness about the impacts of alien species on native fauna and flora. Unless the public is aware of these impacts, alien species will continue to be spread throughout the world (Townsend & Winterbourn 1992; Pimentel *et. al.*, 2001; Lintermans 2004). These education programmes need to be specifically tailored to each target group, because unless there is support and understanding from the community, management plans for alien species will, in all likelihood, fail (Koehn & MacKenzie 2004; Lintermans 2004). Fishing licenses, with an educational pamphlet attached, should be distributed to inform anglers of the dangers of moving species from one place to another.

Chapter 7. General discussion

7.1. Overview

The Sundays River system has been highly altered by anthropogenic actions. The presence of alien species in the system is a threat to the long-term survival of several indigenous species, through predation and inter-breeding. The IBWT scheme has affected the flow regime of the Skoenmakers and Sundays rivers to a large extent, creating conditions suitable for alien species. Darlington Dam was found to hold a source population for alien species. Alien fish were present in all but one of the tributaries.

Species composition in each of the seven rivers, with the exception of the Kabouga River, had changed significantly over time. Communities in Darlington Dam, Korhaansdrift Dam, Skoenmakers and Sundays rivers are now dominated by alien species. In the tributaries, communities are dominated by the endangered Eastern Cape redbfin *P. afer*. Indeed, these tributaries contain four of the last ten populations of *P. afer* (Swartz and Impson, 2007).

Clarias gariepinus was found to be the most successful invader having penetrated all but one of the tributaries. *Clarias gariepinus* are continually re-introduced through the IBWT into Darlington Dam, creating a source population for invasion into the lower Sundays River and its tributaries. *Micropterus salmoides* changed the species composition of the Wit River significantly through selective predation on *P. afer* and *Barbus pallidus*. Only one barrier to the upstream migration of alien fishes was present on the tributaries, and several additional barriers, on all

tributaries, are required to safeguard the populations of indigenous fish in the headwaters of these tributaries.

7.2. Conservation goals

The Sundays River and its tributaries are considered essential components of the GAENP, and improving and restoring its hydrological regimes and natural functioning is a high priority within the GAENP management plan (SANParks, 2006). Saunders et al. (2006) note that an ideal freshwater protected area should be implemented on an intact catchment with natural flow regimes and the absence of alien species. The Sundays River itself is far from an ideal catchment for conservation in that half of the fishes in its waters are alien, its hydrological regime is highly altered from an IBWT scheme and damming, and water extraction is a continual process. Some of the tributaries of the Sundays River, however, have the potential for successful conservation efforts to be applied.

The conservation of headwaters is considered by many researchers to be the optimal method by which aquatic freshwater ecosystems can be maintained and preserved (Angermeier and Winston, 1999, Saunders et al., 2002, Abell et al., 2007). The headwaters of the tributaries in the GAENP are relatively unaffected by anthropogenic activities but the presence of alien fishes in these headwaters is the principal threat to successful conservation efforts. Barrier weirs, preventing upstream movement of these alien fishes, would be the optimal method of restoring natural function to these tributaries, along with the education of the public with regard to the dangers posed by these alien fishes. A potential site for the construction of a barrier weir would be on the

upper Klein Uie River on a sloping bedrock shelf, which currently acts as a partial barrier to upstream fish migration (Bok, 2008). Other management options include the rehabilitation of certain stretches of river, eradication of alien species, translocation of *P. afer* to environments that are not immediately threatened by piscivorous alien species and captive breeding programmes.

7.3. Management options

River rehabilitation

Common rehabilitation techniques include narrowing and re-meandering of streams, re-profiling steep banks and creating specific features such as riffles and backwaters (Pretty et al., 2003). Large rehabilitation projects range from one to several kilometres in length and employ many different rehabilitation techniques and are generally economically expensive, while small rehabilitation projects are generally only a few hundred meters in extent, use few techniques and are cheap to implement (Pretty et al., 2003).

In the Ebbw Fawr River, most of the ichthyofauna was extirpated due to a century's worth of industrial pollution, but recently there were advances in waste treatment, such that the quality of the water improved significantly (Turnpenny and Williams, 1981). Despite the long period of pollution, fish rapidly re-colonized the river once these control measures were effective. However, a large amount of damage had been sustained to the river due to siltation and suspended solids pollution (Turnpenny and Williams, 1981). Although Turnpenny and Williams (1981) concluded that ichthyofauna responded positively to river rehabilitation, Pretty et al.

(2003) found that the value to conservation of implementing these schemes was negligible. This was due to the incorrect alterations, such as adding artificial riffles and flow deflectors to a lowland river, being made to the river under investigation, where instead, the creation of off-channel, marginal and floodplain habitats would have been a better option (Pretty et al., 2003). Pretty et al. (2003) state that the physical rehabilitation of a river is not necessarily equivalent to biological rehabilitation.

The main aim of the IBWT scheme was to provide irrigation for the Great Fish and Sundays River valleys, and the dependence of intensive citrus farming in the Sundays River valley implies that the IBWT scheme will remain in existence (Cambray and Jubb, 1977). The Skoenmakers and Sundays rivers are highly altered systems (Roux et al., 2002), and a rehabilitation project would in all likelihood fail due to the high dependence on these rivers of the local citrus industry and the high costs that would be associated with such a project.

Eradication

The Sundays and Skoenmakers rivers and their associated impoundments are source populations for most of the alien species affecting the freshwater system of the GAENP. The size of the Sundays and Skoenmakers rivers, Darlington Dam, and Korhaansdrift Dam make eradication of these species practically impossible. In addition, the continuous re-introduction of alien species via the Orange-Fish-Sundays IBWT scheme (Cambray and Jubb, 1977) would render such a programme futile. The size of Slagboom Dam, on the Wit River, and the indication that *M. salmoides* has successfully colonized this area also negates the effectiveness of any eradication

scheme, unless the impoundment could be drained beforehand. Once an alien species has successfully colonised an area, it is virtually impossible to eradicate that species (Pimentel et al., 2001, Lintermans, 2004, Neilson et al., 2004).

In the tributaries, which are smaller and thus more manageable than the larger systems, eradication of alien species may be possible through a number of methods. Rotenone is a particularly effective piscicide, which has been used for removal of *G. affinis* and *C. carpio* in New Zealand and Australia (Neilson et al., 2004). This piscicide may prove to be effective in the tributaries of the GAENP, but care will have to be taken as rotenone will also eradicate any indigenous species in the same body of water (Cambray, 2003a). Other methods which have proved to be effective in reducing population sizes of alien invasive species are long-lining or seine-netting. In clearer waters, spear-fishing may be an option to eradicate individual fish.

Control of aliens

The control of alien freshwater fish species is done most effectively with the use of barrier weirs (Minckley, 1995, Lintermans, 2004). A total of nine barriers to fish migration were evaluated on the Sundays, Klein Uie and Wit rivers. Seven of these barriers were present on the Sundays River, four of which were partial barriers, and three were total barriers to upstream fish movement. These barriers will not be useful in the control of alien species as they are below the point of introduction, and barriers are only effective in limiting upstream movement (Minckley, 1995, Adams et al., 2001). Two barriers, one on the Wit River and one on the Klein Uie River, are currently effective in the control of the upstream migration of alien fish. However, the barrier on the Wit River, Slagboom Dam wall, is limiting the movement of *C. gariepinus* only, while *M.*

salmoides has established a large population in Slagboom Dam, and is not being restricted from movement into the upper catchment of the Wit River. The barrier on the Klein Uie River, a sloping bedrock shelf, is restricting the upstream movement of *C. gariepinus*. This barrier is in need of strengthening with the construction of a weir, since a single flood event would allow *C. gariepinus* to circumvent this barrier.

Special cases

Pseudobarbus afer

To protect the last 10 populations of *P. afer* (Swartz and Impson, 2007), several measures need to be considered, including translocation and a captive breeding programme. In Sri Lanka, four endangered endemic rain-forest fishes were translocated from their original threatened environment into a new area, which was pristine (Wikramanayake, 1990). These translocations proved successful, but Wikramanayake (1990) warned that the areas into which these endangered fish are translocated should meet the ecological requirements of the species. In addition, the indigenous community should be considered before any species are added to it (Wikramanayake, 1990).

The goal of any captive breeding programme is to increase the overall population size while maintaining genetic diversity (Minckley, 1995). Several issues need to be addressed before a captive breeding programme is undertaken. Firstly, the population from which the brood stock is selected needs to be sufficiently large to enable that population to continue to survive after a number of individuals have been removed from it (Minckley, 1995). Secondly, much

consideration on which genetic strain should be used should be given, especially when dealing with genetically isolated populations, such as *P. afer* (Minckley, 1995). If a captive breeding programme were to be implemented, the *P. afer* populations from the Klein Uie and Groot Uie rivers should be used, as these are the largest populations per pool (161 and 147 fish per pool respectively), and either the Groot Uie or the upper Krom rivers should be assessed for translocation destinations.

Labeo umbratus

The genetic purity of the Sundays River *L. umbratus* stock is threatened by the introduction of *L. capensis* and Orange River *L. umbratus* stock. Several hybrids between *L. capensis* and *L. umbratus* have already been found in Darlington Dam (Mpho, pers. comm., 2008, MSc Student, Rhodes University). Slagboom Dam, on the Wit River, was constructed in 1949 and therefore pre-dates the construction of the Orange-Fish-Sundays interbasin water transfer scheme, and the associated introduction of *L. capensis* and Orange River *L. umbratus*. The *L. umbratus* found in Slagboom Dam could then be considered one of the only genetically pure stocks along the Sundays River catchment. These fish share this body of water with a large population of *M. salmoides*, which would have some depression effect on their recruitment. Some form of protection for these fish should be investigated.

Education and awareness

There is a strong need for public education and awareness about the impacts of alien species on native fauna and flora. Unless the public is aware of these impacts, alien species will continue to

be spread throughout the world (Townsend & Winterbourn 1992; Pimentel *et. al.*, 2001; Lintermans 2004). Elvira & Almodovar (2001) recommend the use of fishing licenses, with an educational pamphlet attached, to inform anglers of the dangers of moving species from one place to another. These licenses could be issued at Darlington Dam, Korhaansdrift Dam and Slagboom Dam. Educational posters could also be erected at suitable places around the GAENP, such as the lodges and near accessible waterbodies. These education programmes need to be specifically tailored to each target group, because unless there is support and understanding from the community, management plans for alien species will, in all likelihood, fail (Koehn & MacKenzie 2004; Lintermans 2004).

7.4. Future monitoring and research

Monitoring

Monitoring the populations of threatened or endangered fishes should include two objectives; firstly, the size structure of the population should be accurately obtained, and secondly, any changes in the size of the population should be detected as early as possible (Hardie *et al.*, 2006). Multiple pass electrofishing, conducted at the same sites that were assessed in this study, will provide insights into population size changes over time. Population size structure can also be obtained from these data.

Monitoring efforts should also focus on the search for alien fish within the tributaries.

Electrofishing proved to be a useful technique in identifying the presence of *Claris gariepinus* in

the small pools on the tributaries. If *C. gariepinus* are present, long-lines can then be set overnight to remove them from these areas. Unless *C. gariepinus* is present, long-lines should not be set as they are likely to capture *A. mossambica*, which die on the longline. Continual monitoring of alien fish hotspots, such as Gwarrie Pan and upstream of Slagboom Dam, should be initiated to prevent, as much as possible, the upstream movement of these fishes.

Future research

The movement of alien fishes from Darlington into the downstream sections of the Sundays River, and hence the tributaries should be investigated, either by telemetry, or by tagging. This may enable conservation authorities to determine the mode of introduction of these fish into the tributaries. A tagging project is currently being run by the Department of Ichthyology and Fisheries Science in Darlington Dam, Korhaansdrift Dam and the Sundays River. This project should be continued, as it will provide valuable information on the movement patterns and population sizes of alien fishes.

A population estimate of *Labeo umbratus* from Slagboom Dam would be useful to determine the absolute abundance of the genetically pure strain in this impoundment. This information will be particularly useful for determining the risks associated with the eradication of *M. salmoides* from the impoundment.

An understanding of the biology of *P. afer*, in terms of breeding biology, diet, age and growth and physico-chemical tolerances, would be valuable for understanding the dynamics of the headwater environment. Similar studies should be conducted on *B. pallidus* and *G. callidus* to determine the level of interspecific competition that occurs between these three species in the upper catchments.

A dietary analysis of *A. mossambica* in the headwaters of the tributaries would provide useful information on the survival tactics of this species. The level of predation on *P. afer* by *A. mossambica* would be very valuable for determining the importance of *P. afer* to the survival of *A. mossambica* in these environments.

A study of seasonal changes to the community composition of the Sundays and Skoenmakers rivers and Darlington and Korhaansdrift dams would provide valuable insight into the role played by *O. mossambicus* in these environments.

The most important factor to determine in future is the ability of barrier weirs to prevent upstream movement of *M. salmoides* and *C. gariepinus*, since these two species are the greatest threat to the conservation goals of SANParks with regard to the GAENP's freshwater systems

Chapter 8. References

- Abell, R., Allan, J.D., Lehner, B. (2007) Unlocking the potential of protected areas for freshwater. *Biological Conservation* **134**, 48-63.
- Adams, S.B., Frissell, C.A., Rieman, B.E. (2001) Geography of invasion in mountain streams: consequences of headwater lake fish introductions. *Ecosystems* **2001**, 296-307.
- Agostinho, A.A., Thomaz, S.M., Gomes, L.C. (2005) Conservation of the biodiversity of Brazil's inland waters. *Conservation Biology* **19**, 646-652.
- Allendorf, F.W. (1988) Conservation biology of fishes. *Conservation Biology* **2**, 145-148.
- Almaca, C. (1995) Freshwater fish and their conservation in Portugal. *Biological Conservation* **72**, 125-127.
- Angermeier, P.L. (1995) Ecological attributes of extinction-prone species: loss of freshwater fishes of Virginia. *Conservation Biology* **9**, 143-158.
- Angermeier, P.L., Schlosser, I.J. (1989) Species-area relationship for stream fishes. *Ecology* **70**, 1450-1462.
- Angermeier, P.L., Winston, M.R. (1999) Characterizing fish community diversity across Virginia landscapes: prerequisite for conservation. *Ecological Applications* **9**, 335-349.
- Argent, D.G., Bishop, J.A., Stauffer, J.R., Carline, R.F., Myers, W.L. (2003) Predicting freshwater fish distributions using landscape-level variables. *Fisheries Research* **60**, 17-32.
- Baird, I.G., Flaherty, M.S. (2005) Mekong river fish conservation zones in southern Laos: assessing effectiveness using local ecological knowledge. *Environmental Management* **36**, 439-454.
- Balik, S. (1995) Freshwater fish in Anatolia, Turkey. *Biological Conservation* **72**, 213-223.
- Balon, E.K. (1975) The eels of Lake Kariba: distribution, taxonomic status, age, growth and density. *Journal of Fish Biology* **7**, 797-815.

- Baltz, D.M., Moyle, P.B. (1993) Invasion resistance to introduced species by a native assemblage of California stream fishes. *Ecological Applications* **3**.
- Begon, M., Townsend, C.R., Harper, J.L. (2006) *Ecology: from individuals to ecosystems*, Blackwell Publishing, Oxford, 738pp
- Benstead, J.P., De Rham, P.H., Gattolliat, J.L., *et al.* (2003) Conserving Madagascar's freshwater biodiversity. *Bioscience* **53**, 1101-1111.
- Bianco, P.G. (1995) Mediterranean endemic freshwater fishes of Italy. *Biological Conservation* **72**, 159-170.
- Bishir, J.W., Lancia, R.A. (1996) On catch-effort methods of estimating animal abundance. *Biometrics* **52**, 1457-1466.
- Bohlin, T., Sundstrom, B. (1977) Influence of unequal catchability on population estimates using the Lincoln index and the removal method applied to electro-fishing. *Oikos* **28**, 123-129.
- Bok, A.H. (1983) The demography, breeding biology and management of two mullet species (Pisces: Mugilidae) in the Eastern Cape, South Africa. PhD Thesis, Rhodes University, 268 pages.
- Bok, A.H. (2008) Obstacles to migration and proposed location and design of fishways and barriers to migration. Sundays River Freshwater Fish Research Project: Report prepared for SANParks, 61pp.
- Brower, A., Reedy, C., Yelin-Kefer, J. (2001) Consensus versus conservation in the upper Colorado River basin recovery program. *Conservation Biology* **15**, 1001-1007.
- Bruton, M.N., Bok, A.H., Davies, M.T.T. (1987) Life history styles of diadromous fishes in inland waters of southern Africa. *Transactions of the American Fisheries Society* **1**, 104-121.
- Caiola, N., DeSostoa, A. (2005) Possible reasons for the decline of two native toothcarps in the Iberian Peninsula: evidence of competition with the introduced Eastern mosquitofish. *Journal of Applied Ichthyology* **21**, 358-363.

- Cambray, J.A. (1990) Adaptive significance of a longitudinal migration by juvenile freshwater fish in the Gamtoos River System, South Africa. *South African Journal of Wildlife Research* **20**, 148-156.
- Cambray, J.A. (2003a) The global impact of alien trout species — a review; with reference to their impact in South Africa. *African Journal of Aquatic Science* **28**, 61-67.
- Cambray, J.A. (2003b) The need for research and monitoring on the impacts of translocated sharptooth catfish, *Clarias gariepinus*, in South Africa. *African Journal of Aquatic Science* **28**, 191-195.
- Cambray, J.A. (2005) Africa's *Clarias gariepinus* (Teleostei: Clariidae) appears in rivers in Brazil. *African Journal of Aquatic Science* **30**, 201-202.
- Cambray, J.A., Hecht, T. (1995) Comparison of the growth of two closely related redbfin minnows, *Pseudobarbus afer* (Peters, 1864) and *Pseudobarbus asper* (Boulenger, 1911) (Pisces, Cyprinidae), in the Gamtoos River System, South Africa. *Journal of African Zoology* **109**, 349-376.
- Cambray, J.A., Jubb, R.A. (1977) Dispersal of Fishes via the Orange-Fish Tunnel, South Africa. *Journal of the Limnological Society of South Africa* **3**, 33-35.
- Castelo, R. (1994) Biogeographical considerations of fish diversity in Bioko. *Biodiversity and Conservation* **3**, 808-827.
- Caswell, H. (1976) Community structure: a neutral model analysis. *Ecological Monographs* **46**, 327-354.
- Changeux, T., Pont, D. (1995) Current status of the riverine fishes of the French Mediterranean basin. *Biological Conservation* **72**, 137-158.
- Clay, D. (1979) Sexual maturity and fecundity of the African catfish (*Clarias gariepinus*) with an observation on the spawning behaviour of the Nile catfish (*Clarias lazera*). *Zoological Journal of the Linnaean Society* **65**, 351-365.
- Coad, B. (1981) Environmental change and its impact on the freshwater fishes of Iran. *Biological Conservation* **19**, 51-80.

- Cochran, P.A., Adelman, I.R. (1982) Seasonal aspects of daily ration and diet of largemouth bass, *Micropterus salmoides*, with an evaluation of gastric evacuation. *Environmental Biology of Fishes* **7**, 265-275.
- Contreras-B, S., Lozano-V, M.L. (1994) Water, endangered fishes, and development perspectives in arid lands of Mexico. *Conservation Biology* **8**, 379-387.
- Cooke, S.J., Bunt, C.M., Hamilton, S.J., *et al.* (2005) Threats, conservation strategies, and prognosis for suckers (Catostomidae) in North America: insights from regional case studies of a diverse family of non-game species. *Biological Conservation* **121**, 317-331.
- Cousins, S.H. (1991) Species diversity measurement: choosing the right index. *Trends in Ecology and Evolution* **6**, 190-192.
- Cowling, R.M., Pressey, R.L., Rouget, M., Lombard, A.T. (2003) A conservation plan for a global biodiversity hotspot - the Cape Floristic Region, South Africa. *Biological Conservation* **112**, 191-216.
- Coyle, K.J. (1993) The new advocacy for aquatic species conservation. *Journal of the North American Benthological Society* **12**, 185-188.
- Craig, J.F., Halls, A.S., Barr, J.J.F., Bean, C.W. (2004) The Bangladesh floodplain fisheries. *Fisheries Research* **66**, 271-286.
- Crivelli, A.J. (1981) The biology of the common carp, *Cyprinus carpio* L. in the Camargue, southern France. *Journal of Fish Biology* **18**, 271-290.
- Curtis, B., Roberts, K.S., Griffin, M., Bethune, S., Hay, C.J., Kolberg, H. (1998) Species richness and conservation of Namibian freshwater macro-invertebrates, fish and amphibians. *Biodiversity and Conservation* **7**, 447-466.
- de Moor, F.C., Wilkinson, R.C., Herbst, H.M. (1986) Food and feeding habits of *Oreochromis mossambicus* (Peters) in hypertrophic Hartbeespoort Dam, South Africa. *South African Journal of Zoology* **21**, 170-176.
- de Moor, I.J., Bruton, M.N. (1988) Atlas of alien and translocated indigenous aquatic animals in southern Africa. South African National Scientific Programmes No. 144, 310pp.

- DeJong, T.M. (1975) A comparison of three diversity indices based on their components of richness and evenness. *Oikos* **26**, 222-227.
- Dudgeon, D. (2000) Conservation of freshwater biodiversity in Oriental Asia: constraints, conflicts, and challenges to science and sustainability. *Limnology* **1**, 237-243.
- Dudgeon, D. (2005) River rehabilitation for conservation of fish biodiversity in monsoonal Asia. *Ecology and Society* **10**, 15-34.
- Duncan, J.R., Lockwood, J.L. (2001) Extinction in a field of bullets: a search for causes in the decline of the world's freshwater fishes. *Biological Conservation* **102**, 97-105.
- Eberle, M.E., Hargett, E.G., Wenke, T.L., Mandrak, N.E. (2002) Changes in fish assemblages, Solomon River basin, Kansas: habitat alterations, extirpations and introductions. *Transactions of the Kansas Academy of Science* **105**, 178-192.
- Economidis, P.S. (1995) Endangered freshwater fishes of Greece. *Biological Conservation* **72**, 201-211.
- Elvira, B. (1995) Conservation status of endemic freshwater fish in Spain. *Biological Conservation* **72**, 129-136.
- Fesl, C. (2002) Niche-oriented species-abundance models: Different approaches of their application to larval chironomid (Diptera) assemblages in a large river. *The Journal of Animal Ecology* **71**, 1085-1094.
- Fievet, E., Bonnet-Arnaud, P., Mallet, J.P. (1999) Efficiency and sampling bias of electrofishing for freshwater shrimp and fish in two Caribbean streams, Guadeloupe Island. *Fisheries Research* **44**, 149-166.
- Fu, C., Wu, J., Chen, J., Wu, Q., Lei, G. (2003) Freshwater fish biodiversity in the Yangtze River basin of China: patterns, threats and conservation. *Biodiversity and Conservation* **12**, 1649-1685.
- Gaigher, I.G. (1977) Reproduction of the catfish (*Clarias gariepinus*) in the Hardap Dam, South West Africa. *Madoqua* **10**, 55-59.
- Garcia-Berthou, A.B.C. (2002) Ontogenetic diet shifts and interrupted piscivory in introduced largemouth bass (*Micropterus salmoides*). *International Review of Hydrobiology* **87**, 353-363.

- Godinho, F.N., Ferreira, M.T. (2000) Composition of endemic fish assemblages in relation to exotic species and river regulation in a temperate stream. *Biological Invasions* **2**, 231-244.
- Goren, M., Ortal, R. (1999) Biogeography, diversity and conservation of the inland water fish communities in Israel. *Biological Conservation* **89**, 1-9.
- Gould, W.R., Pollock, K.H. (1997) Catch-effort estimation of population parameters under the robust design. *Biometrics* **53**, 207-216.
- Gratwicke, B., Marshall, B.E. (2001) The relationship between the exotic predators *Micropterus salmoides* and *Serranochromis robustus* and native stream fishes in Zimbabwe. *Journal of Fish Biology* **58**, 68-75.
- Habit, E., Belk, M.C., Tuckfield, R.C., Parra, O. (2006) Response of the fish community to human-induced changes in the Biobio River in Chile. *Freshwater Biology* **51**, 1-11.
- Hardie, S.A., Barmuta, L.A., White, R.W.G. (2006) Comparison of day and night fyke netting, electrofishing and snorkelling for monitoring a population of the threatened golden galaxias (*Galaxius auratus*). *Hydrobiologia* **560**, 145-158.
- Hill, M.O. (1973) Diversity and evenness: a unifying notation and its consequences. *Ecology* **54**, 427-432.
- Hitt, N.P., Angermeier, P.L. (2006) Effects of adjacent streams to local fish assemblage structure in western Virginia: implications for biomonitoring. *Transactions of the American Fisheries Society* **48**, 75-86.
- Hurlbert, S.H. (1971) The nonconcept of species diversity: a critique and alternative parameters. *Ecology* **52**, 577-586.
- Ibanez, C., Oberdorff, T., Teugels, G., *et al.* (2007) Fish assemblage structure and function along environmental gradients in rivers of Gabon (Africa). *Ecology of Freshwater Fish* **16**, 315-334.
- Iguchi, K., Matsuura, K., McNyset, K.M., *et al.* (2004) Predicting invasions of North American basses in Japan using native range data and a genetic algorithm. *Transactions of the American Fisheries Society* **133**, 845-854.

- Inogwabini, B.I. (2005) Fishes of the Salonga National Park, Democratic Republic of Congo: survey and conservation issues. *Oryx* **39**, 78-81.
- Jang, M.H., Lucas, M.C., Joo, G.J. (2003) The fish fauna of mountain streams in South Korean national parks and its significance to conservation of regional freshwater fish biodiversity. *Biological Conservation* **114**, 115-126.
- Jordan, F., Jelks, H.L., Bortone, S.A., Dorazio, R.M. (2008) Comparison of visual survey and seining methods for estimating abundance of an endangered, benthic stream fish. *Environmental Biology of Fishes* **81**, 313-319.
- Jowett, I.G., Richardson, J. (1996) Distribution and abundance of freshwater fish in New Zealand rivers. *New Zealand Journal of Marine and Freshwater Research* **30**, 239-255.
- Jowett, I.G., Richardson, J., McDowall, R.M. (1996) Relative effects of in-stream habitat and land use on fish distribution and abundance in tributaries of the Grey River, New Zealand. *New Zealand Journal of Marine and Freshwater Research* **30**, 463-475.
- Keith, P. (2000) The part played by protected areas in the conservation of threatened French freshwater fish. *Biological Conservation* **92**, 265-273.
- King, C.E. (1964) Relative abundance of species and MacArthur's model. *Ecology* **45**, 716-727.
- Koehn, J.D., MacKenzie, R.F. (2004) Priority management actions for alien freshwater fish species in Australia. *New Zealand Journal of Marine and Freshwater Research* **38**, 457-472.
- Laurance, W.F. (2005) When bigger is better: the need for Amazonian mega-reserves. *Trends in Ecology and Evolution* **20**, 645-648.
- Ling, N. (2004) *Gambusia* in New Zealand: really bad or just misunderstood? *New Zealand Journal of Marine and Freshwater Research* **38**, 473-480.
- Ling, N., Willis, K. (2005) Impacts of mosquitofish, *Gambusia affinis*, on black mudfish, *Neochanna diversus*. *New Zealand Journal of Marine and Freshwater Research* **39**, 1215-1223.

- Lintermans, M. (2004) Human-assisted dispersal of alien freshwater fish in Australia. *New Zealand Journal of Marine and Freshwater Research* **38**, 481-501.
- Lloyd, L.N., Walker, K.F. (1986) Distribution and conservation status of small freshwater fish in the River Murray, South Australia. *Transactions of the Royal Society of South Australia* **110**, 49-57.
- Lombard, A.T., Johnson, C.F., Cowling, R.M., Pressey, R.L. (2001) Protecting plants from elephants: botanical reserve scenarios within the Addo Elephant National Park, South Africa. *Biological Conservation* **102**, 191-203.
- Lopez-Rojas, H., Bonilla-Rivero, A.L. (2000) Anthropogenically induced fish diversity reduction in Lake Valencia Basin, Venezuela. *Biodiversity and Conservation* **9**, 757-765.
- Lowe, S., Browne, M., Boudjelas, S., DePoorter, M. (2000) 100 of the world's worst invasive alien species. A selection from the global invasive species database. The Invasive Species Specialist Group (ISSG), a specialist group of the Species Survival Commission (SSC) of the World Conservation Union (IUCN), 12pp.
- Mahon, R. (1980) Accuracy of catch-effort methods for estimating fish density and biomass in streams. *Environmental Biology of Fishes* **5**, 343-360.
- Maitland, P.S. (1985) Criteria for the selection of important sites for freshwater fish in the British Isles. *Biological Conservation* **31**, 335-353.
- Maitland, P.S. (1995) The conservation of freshwater fish: past and present experience. *Biological Conservation* **72**, 259-270.
- Maitland, P.S. (2004) Ireland's most threatened and rare freshwater fish: an international perspective on fish conservation. *Biology and Environment: Proceedings of the Royal Irish Academy* **104B**, 5-16.
- Maitland, P.S., Lyle, A.A. (1991) Conservation of freshwater fish in the British Isles: the current status and biology of threatened species. *Aquatic Conservation: Marine and Freshwater Ecosystems* **1**, 25-54.
- Meffe, G.K. (1984) Effects of abiotic disturbance on coexistence of predator-prey fish species. *Ecology* **65**, 1525-1534.

- Minckley, W.L. (1995) Translocation as a tool for conserving imperiled fishes: experiences in western United States. *Biological Conservation* **72**, 297-309.
- Morgan, D.L., Gill, H.S., Maddern, M.G., Beatty, S.J. (2004) Distribution and impacts of introduced freshwater fishes in Western Australia. *New Zealand Journal of Marine and Freshwater Research* **38**, 511-523.
- Moyle, P.B. (1995) Conservation of the native freshwater fishes in the Mediterranean-type climate of California, USA: a review. *Biological Conservation* **72**, 271-279.
- Moyle, P.B., Crain, P.K., Whitener, K., Mount, J.F. (2003) Alien fishes in natural streams: fish distribution, assemblage structure, and conservation in the Cosumnes River, California, U.S.A. *Environmental Biology of Fishes* **68**, 143-162.
- Moyle, P.B., Light, T. (1996a) Biological invasions of fresh water: empirical rules and assembly theory. *Biological Conservation* **78**, 149-161.
- Moyle, P.B., Light, T. (1996b) Fish invasions in California: do abiotic factors determine success? *Ecology* **77**, 1666-1670.
- Neilson, K., Kelleher, R., Barnes, G., Speirs, D., Kelly, J. (2004) Use of fine-mesh monofilament gill nets for the removal of rudd (*Scardinius erythrophthalmus*) from a small lake complex in Waikato, New Zealand. *New Zealand Journal of Marine and Freshwater Research* **38**, 525-539.
- Nishizawa, E., Kurokawa, T., Yabe, M. (2006) Policies and resident's willingness to pay for restoring the ecosystem damaged by alien fish in Lake Biwa, Japan. *Environmental Science and Policy* **9**, 448-456.
- Oberdorff, T., Guegan, J.F., Hugueny, B. (1995) Global scale patterns of fish species richness in rivers. *Ecography* **18**, 345-352.
- Pandit, A.K. (1991) Conservation of wildlife resources in wetland ecosystems of Kashmir, India. *Journal of Environmental Management* **33**, 143-154.
- Pascual, M., Macchi, P., Urbanski, J., *et al.* (2002) Evaluating potential effects of exotic freshwater fish from incomplete species presence-absence data. *Biological Invasions* **4**, 101-113.
- Peet, R.K. (1975) Relative diversity indices. *Ecology* **56**, 496-498.

- Pegg, M.A., McClelland, M.A. (2004) Spatial and temporal patterns in fish communities along the Illinois River. *Ecology of Freshwater Fish* **13**, 125-135.
- Pelicice, F.M., Agostinho, A.A. (2005) Perspectives on ornamental fisheries in the upper Parana River floodplains, Brazil. *Fisheries Research* **72**, 109-119.
- Peterson, J.T., Thurow, R.F., Guzevich, J.W. (2004) An evaluation of multipass electrofishing for estimating the abundance of stream-dwelling salmonids. *Transactions of the American Fisheries Society* **133**, 462-475.
- Pimentel, D., McNair, S., Janecka, J., *et al.* (2001) Economic and environmental threats of alien plant, animal, and microbe invasions. *Agriculture, Ecosystems and Environment* **84**, 1-20.
- Pimentel, D., Zuniga, R., Morrison, D. (2005) Update on the environmental and economic costs associated with alien-invasive species in the United States. *Ecological Economics* **2005**, 273-288.
- Platania, S.P. (1991) Fishes of the Rio Chama and upper Rio Grande, New Mexico, with preliminary comments on their longitudinal distribution. *The Southwestern Naturalist* **36**, 186-193.
- Potts, W.M. (2003) Towards the development of species-specific fish production models for small reservoirs in southern Africa. PhD Thesis, Rhodes University, 161 pages.
- Potts, W.M., Booth, A.J., Hecht, T., Andrew, T.G. (2005) Reproductive biology of a riverine cyprinid, *Labeo umbratus* (Teleostei: Cyprinidae), in small South African reservoirs. *African Journal of Aquatic Science* **30**, 147-155.
- Pretty, J.L., Harrison, S.S.C., Sheperd, D.J., Smith, C., Hildrew, A.G., Hey, R.D. (2003) River rehabilitation and fish populations: assessing the benefit of instream structures. *The Journal of Applied Ecology* **40**, 251-265.
- Putman, R.J. (1994) *Community Ecology*, Chapman & Hall, London, 180pp
- Rakaj, N., Flloko, A. (1995) Conservation status of freshwater fish of Albania. *Biological Conservation* **72**, 195-199.

- Rehage, J.S., Barnett, B.K., Sih, A. (2005) Foraging behaviour and invasiveness: do invasive *Gambusia* exhibit higher feeding rates and broader diets than their non-invasive relatives? *Ecology of Freshwater Fish* **14**, 352-360.
- Reinthal, P.N., Stiassny, M.L.J. (1991) The freshwater fishes of Madagascar: a study of an endangered fauna with recommendations for a conservation strategy. *Conservation Biology* **5**, 231-242.
- Ricciardi, A., Rasmussen, J.B. (1999) Extinction rates of North American freshwater fauna. *Conservation Biology* **13**, 1220-1222.
- Roux, D., deMoor, F., Cambray, J.A., Barber-James, H. (2002) Use of landscape-level river signatures in conservation planning: a South African case study. *Conservation Ecology* **6**, 6-20.
- Russell, I.A. (1999) Freshwater Fish of the Addo Elephant National Park. *South African Journal of Aquatic Science* **24**, 57-70.
- SANParks (2006) Addo Elephant National Park, park management plan. Internal Report, 39pp.
- Saunders, D.L., Meeuwig, J.J., Vincent, A.C.J. (2002) Freshwater protected areas: strategies for conservation. *Conservation Biology* **16**, 30-41.
- Scott, L.E.P., Skelton, P.H., Booth, A.J., Verheust, L., Harris, R., Dooley, J. (2006) Atlas of southern African freshwater fishes. *Smithiana Monograph* **2**, 1-303.
- Seber, G.A.F. (1973) *The estimation of animal abundance and related parameters*, Charles Griffin & Company, London, 506pp
- Seber, G.A.F., LeCren, E.D. (1967) Estimating population parameters from catches large relative to the population. *Journal of Animal Ecology* **36**, 631-643.
- Shumway, C.A. (1999) *Forgotten waters: freshwater and marine ecosystems in Africa. Strategies for biodiversity conservation and sustainable development*, New England Aquarium, Boston, 181pp
- Skelton, P.H. (1983) Perspectives on the conservation of threatened fishes in southern Africa. *The Naturalist* **27**, 3-12.

- Skelton, P.H. (1987) South African red data book – fishes. *South African National Scientific Programmes Report* **137**, 1-199.
- Skelton, P.H. (1990) The conservation and status of threatened fishes in southern Africa. *Journal of Fish Biology* **37 (Suppl. A)**, 87-95.
- Skelton, P.H. (1993) *A complete guide to the freshwater fishes of southern Africa*, Southern Book Publishers, Halfway House, 388pp
- Skelton, P.H. (2000) Flagships and fragments – perspectives on the conservation of freshwater fishes in southern Africa. *African Journal of Aquatic Science* **25**, 37-42.
- Skelton, P.H. (2002) An overview of the challenges of conserving freshwater fish in South Africa. In: *Conservation of freshwater fishes: options for the future*. (Eds. M.J. Collares-Pereira, M.M. Coelho, I.G. Cowx), Fishing News Books, Oxford, pp. 221-236.
- Skelton, P.H., Cambray, J.A., Lombard, A.T., Benn, G.A. (1995) Patterns of distribution and conservation status of freshwater fishes in South Africa. *South African Journal of Zoology* **30**, 71-81.
- Spataru, P., Viveen, W.J.A.R., Gophen, M. (1987) Food composition of *Clarias gariepinus* (= *C. lazera*) (Cypriniformes, Clariidae) in Lake Kinneret (Israel). *Hydrobiologia* **144**, 77-82.
- Stadtlander, T. (2006) Comparative biology of the smallmouth yellowfish *Labeobarbus aeneus* in Glen Melville Reservoir and the Great Fish River, Eastern Cape, South Africa. MSc Thesis, Rhodes University, 77 pages.
- Swartz, E., Impson, N.D. (2007) *Pseudobarbus afer*. In: *2007 Red List of Threatened Species*. Vol. 2008. IUCN 2007.
- Tokeshi, M. (1990) Niche apportionment or random assortment: Species abundance patterns revisited. *The Journal of Animal Ecology* **59**, 1129-1146.
- Tokeshi, M. (1993) Species abundance patterns and community structure. *Advances in Ecological Research* **24**, 112-186.
- Townsend, C.R. (1996) Invasion biology and ecological impacts of brown trout *Salmo trutta* in New Zealand. *Biological Conservation* **78**, 13-22.

- Townsend, C.R. (2003) Individual, population, community, and ecosystem consequences of a fish invader in New Zealand streams. *Conservation Biology* **17**, 38-47.
- Townsend, C.R., Crowl, T.A. (1991) Fragmented population structure in a native New Zealand fish: an effect of introduced brown trout? *Oikos* **61**, 347-354.
- Townsend, C.R., Winterbourn, M.J. (1992) Assessment of the environmental risk posed by an exotic fish: the proposed introduction of channel catfish (*Ictalurus punctatus*) to New Zealand. *Conservation Biology* **6**, 273-282.
- Turnpenny, A.W.H., Williams, R. (1981) Factors affecting the recovery of fish populations in an industrial river. *Environmental Pollution* **26**, 39-58.
- Tweddle, D., Ter Morshuizen, L. (1996) A fish survey of the Little Fish, Skoenmakers and Sundays river systems, 4th – 11th June 1996. JLB Smith Institute of Ichthyology: Investigational Report, No. 55, 14pp.
- Udoidiong, O.M. (1988) A comparative survey of the fish communities of two Nigerian headwater streams in relation to man-made perturbations. *Biological Conservation* **45**, 93-108.
- Vanicek, C.D. (1970) Distribution of Green River fishes in Utah and Colorado following closure of Flaming Gorge Dam. *The Southwestern Naturalist* **14**, 297-315.
- Vitousek, P.M., D'Antonio, C.M., Loope, L.L., Rejmanek, M., Westbrooks, R. (1997) Introduced species: a significant component of human-caused global change. *New Zealand Journal of Ecology* **21**, 1-16.
- Wei, Q., Ke, F., Zhang, J., *et al.* (1997) Biology, fisheries, and conservation of sturgeons and paddlefish in China. *Environmental Biology of Fishes* **48**, 241-255.
- Whitfield, A.K. (1998) Biology and ecology of fishes in southern African estuaries. *Ichthyological Monographs of the J.L.B. Smith Institute of Ichthyology* **2**, 1-223.
- Wikramanayake, E.D. (1990) Conservation of endemic rain forest fishes of Sri Lanka: results of a translocation experiment. *Conservation Biology* **4**, 32-37.
- Wildman, T.L., Neumann, R.M. (2003) Comparison of snorkeling and electrofishing for estimating abundance and size structure of brook trout and brown trout in two southern New England streams. *Fisheries Research* **60**, 131-139.

- Winemiller, K.O., Kelso-Winemiller, L.C. (2003) Food habits of tilapiine cichlids of the Upper Zambezi River and floodplain during the descending phase of the hydrologic cycle. *Journal of Fish Biology* **63**, 120-128.
- Woodford, D.J., Impson, N.D. (2004) A preliminary assessment of the impact of alien rainbow trout (*Oncorhynchus mykiss*) on indigenous fishes of the upper Berg River, Western Cape Province, South Africa. *African Journal of Aquatic Science* **29**, 107-111.

Appendix 1. Raw Data (Dates and GPS co-ordinates)

Site ID	Watercourse	Date	Time	South	East
DAR-040-630	Darlington	15/11/2007	12:00	33 12.040	25 09.630
DAR-349-348	Darlington	15/11/2007	11:00	33 12.349	25 09.348
DAR-509-132	Darlington	23/02/2008	11:15	33 11.509	25 08.132
DAR-676-130	Darlington	23/02/2008	10:50	33 11.676	25 08.130
DAR-784-175	Darlington	23/02/2008	10:30	33 11.784	25 08.175
DAR-919-092	Darlington	15/11/2007	14:00	33 11.919	25 08.092
DAR-930-111	Darlington	23/02/2008	09:00	33 11.930	25 08.111
DAR-963-099	Darlington	15/11/2007	13:00	33 08.963	25 09.099
DAR-988-115	Darlington	15/11/2007	15:00	33 11.988	25 08.115
DAR-989-137	Darlington	23/02/2008	09:30	33 11.989	25 08.137
GUIE-822-138	Groot Uie	05/09/2007	08:22	33 20.822	25 32.138
GUIE-844-188	Groot Uie	05/09/2007	10:15	33 20.844	25 32.188
GUIE-871-185	Groot Uie	05/09/2007	11:23	33 20.871	25 32.185
GUIE-899-864	Groot Uie	05/09/2007	15:15	33 20.899	25 31.864
GUIE-904-890	Groot Uie	05/09/2007	13:56	33 20.904	25 31.890
GUIE-972-148	Groot Uie	05/09/2007	12:26	33 20.972	25 32.148
KAB-351-765	Kabouga	05/09/2007	18:12	33 18.351	25 22.765
KAB-855-777	Kabouga	27/09/2007	07:00	33 19.855	25 25.777
KAB-893-780	Kabouga	27/09/2007	08:00	33 19.893	25 25.780
KOR-566-362	Sundays	27/03/2008	08:00	33 22.566	25 21.362
KRO-166-771	Krom	06/11/2007	13:03	33 23.166	25 47.771
KRO-167-761	Krom	06/11/2007	13:35	33 23.167	25 47.761
KRO-300-708	Krom	06/11/2007	14:21	33 23.300	25 47.708
KRO-310-710	Krom	07/11/2007	09:57	33 23.310	25 47.710
KRO-550-520	Krom	07/11/2007	12:33	33 23.550	25 47.520
KRO-787-578	Krom	06/11/2007	15:00	33 25.787	25 46.578
KRO-981-261	Krom	07/11/2007	10:42	33 24.981	25 47.261
KUIE-024-033	Klein Uie	06/09/2007	10:00	33 19.024	25 30.033
KUIE-269-588	Klein Uie	06/09/2007	12:20	33 19.269	25 29.588
KUIE-280-710	Klein Uie	06/09/2007	12:08	33 19.280	25 29.710
KUIE-375-518	Klein Uie	26/09/2007	09:40	33 18.375	25 29.518
KUIE-389-438	Klein Uie	06/09/2007	12:33	33 19.389	25 29.438
KUIE-498-500	Klein Uie	06/09/2007	13:21	33 19.498	25 29.500
KUIE-634-498	Klein Uie	06/09/2007	17:11	33 20.634	25 29.498
KUIE-695-069	Klein Uie	26/09/2007	11:00	33 18.695	25 30.069
KUIE-814-495	Klein Uie	25/09/2007	16:33	33 19.814	25 29.495
KUIE-821-476	Klein Uie	06/09/2007	14:13	33 19.821	25 29.476
KUIE-835-444	Klein Uie	06/09/2007	15:04	33 19.835	25 29.444

KUIE-881-301	Klein Uie	06/09/2007	16:30	33 19.881	25 29.301
SKO-239-421	Skoenmakers	16/01/2008	16:00	33 08.239	25 09.421
SKO-437-534	Skoenmakers	16/01/2008	15:00	33 08.437	25 09.534
SKO-440-851	Skoenmakers	16/01/2008	08:00	33 06.440	25 13.851
SKO-463-727	Skoenmakers	17/01/2008	06:50	33 06.463	25 13.727
SKO-464-718	Skoenmakers	17/01/2008	06:45	33 06.464	25 13.718
SKO-464-723	Skoenmakers	17/01/2008	07:00	33 06.464	25 13.723
SKO-492-541	Skoenmakers	17/01/2008	09:30	33 06.492	25 13.541
SUN-439-974	Sundays	23/11/2007	13:00	33 12.439	25 08.974
SUN-502-601	Sundays	03/09/2007	17:00	33 20.502	25 21.601
SUN-502-601	Sundays	26/09/2007	21:00	33 20.502	25 21.601
SUN-502-601	Sundays	26/03/2008	10:30	33 20.502	25 21.601
SUN-519-999	Sundays	23/11/2007	15:00	33 12.519	25 08.999
SUN-904-001	Sundays	27/03/2008	10:00	33 22.904	25 21.001
SUN-974-961	Sundays	27/03/2008	10:00	33 22.974	25 20.961
UIE-747-545	Uie	05/09/2007	17:03	33 22.747	25 27.545
WIT-027-032	Wit	12/09/2007	13:00	33 22.027	25 41.032
WIT-096-676	Wit	13/09/2007	08:00	33 22.096	25 39.676
WIT-213-958	Wit	12/09/2007	08:27	33 22.213	25 41.958
WIT-221-898	Wit	12/09/2007	10:23	33 22.221	25 41.898
WIT-225-944	Wit	12/09/2007	09:28	33 22.225	25 41.944
WIT-228-920	Wit	12/09/2007	09:54	33 22.228	25 41.920
WIT-285-612	Wit	08/02/2008	08:30	33 22.285	25 40.612
WIT-311-454	Wit	11/09/2007	16:33	33 22.311	25 40.454
WIT-317-443	Wit	11/09/2007	18:06	33 22.317	25 40.443
WIT-317-443	Wit	07/02/2008	08:00	33 22.317	25 40.443
WIT-317-443	Wit	08/02/2008	08:00	33 22.317	25 40.443
WIT-348-331	Wit	16/09/2007	12:00	33 22.348	25 40.331
WIT-348-331	Wit	08/02/2008	12:00	33 22.348	25 40.331
WIT-767-662	Wit	30/03/2008	16:00	33 19.767	25 40.662
WIT-790-321	Wit	29/03/2008	12:00	33 19.790	25 40.321
WIT-798-435	Wit	30/03/2008	07:00	33 19.798	25 40.435
WIT-803-591	Wit	30/03/2008	08:11	33 19.803	25 40.591
WIT-816-533	Wit	12/09/2007	15:56	33 19.816	25 40.533
WIT-855-123	Wit	30/03/2008	14:00	33 19.855	25 40.123
WIT-858-812	Wit	31/03/2008	08:00	33 19.858	25 39.812
WIT-884-925	Wit	11/09/2007	14:35	33 21.884	25 40.925
WIT-898-683	Wit	12/09/2007	15:33	33 19.898	25 40.683
WIT-898-683	Wit	06/02/2008	14:00	33 19.898	25 40.683
WIT-903-359	Wit	11/09/2007	12:20	33 20.903	25 41.359
WIT-903-359	Wit	06/02/2008	15:40	33 20.903	25 41.359

WIT-926-349	Wit	11/09/2007	10:10	33 20.926	25 41.349
WIT-926-349	Wit	06/02/2008	16:18	33 20.926	25 41.349

Appendix 2. Raw data (Water quality data)

Site ID	Temp	pH	uS	DO	% O2 Sat	TDS
DAR-040-630	24.6	8.7	677	11.5	143	349
DAR-349-348	23.6	8.82	720	11.1	135.6	359
DAR-509-132	25.9	8.9	562	9.2	117.3	286
DAR-676-130	25.6	9.01	560	10.3	129.6	285
DAR-784-175	25.4	9	560	10.7	135.4	286
DAR-919-092	20.3	8.7	699	9.8	111.5	356
DAR-930-111	24.6	8.9	567	9.7	123.8	289
DAR-963-099	24.2	9.27	712	12.9	158.8	364
DAR-988-115	20.1	8.7	700	9.6	106.2	357
DAR-989-137	24.4	8.89	572	9.6	120	291
GUIE-822-138	13.4	6.9	283	6.2	62.2	139
GUIE-844-188	16.2	7.05	225	6.5	62.1	113
GUIE-871-185	14.2	6.92	250	3.2	32.5	122
GUIE-899-864	15.7	6.91	273	6.7	69	135
GUIE-904-890	18.5	7.13	270	6.4	67.5	135
GUIE-972-148	17.6	6.92	292	8.4	91.6	153
KAB-351-765	17.6	7.45	1089	10.8	116.4	545
KAB-855-777	17.4	6.96	533	11	122	266
KAB-893-780	15	6.92	540	2.7	27.7	269
KOR-566-362	19	8.4	553	0	0	289
KRO-166-771	16.7	7.25	834	6.7	69.7	425
KRO-167-761	15.9	7.11	809	3.3	34.2	408
KRO-300-708	16.9	6.65	680	1.8	18.7	347
KRO-310-710	16.9	6.73	694	8.3	101	350
KRO-550-520	16.5	7.58	982	10	117.5	492
KRO-787-578	16.7	7.7	3999	6.1	63.6	2000
KRO-981-261	27.6	8.38	1016	16.2	207	518
KUIE-024-033	13.7	7.21	343	6.4	62	171
KUIE-269-588	14.9	7.4	306	6.9	69	154
KUIE-280-710	14.2	7.21	282	7.4	73.2	138
KUIE-375-518	13.7	6.37	392	5.4	53.9	225
KUIE-389-438	16.6	7.28	371	8	83.5	184
KUIE-498-500	16.7	7.36	322	6.4	66	160
KUIE-634-498	15.2	7.25	600	8.8	89.2	300
KUIE-695-069	22.2	6.99	406	6.9	73.2	199
KUIE-814-495	18	6.73	249	9.8	106.9	125
KUIE-821-476	17.2	7.21	250	9.7	102.8	124
KUIE-835-444	17.9	7.31	253	10.1	107.8	127

KUIE-881-301	15.1	7.4	256	8.7	82.9	131
SKO-239-421	0	0	0	0	0	0
SKO-437-534	29.2	8.05	714	0	0	352
SKO-440-851	25.5	7.43	683	0	0	342
SKO-463-727	0	0	0	0	0	0
SKO-464-718	0	0	0	0	0	0
SKO-464-723	0	7.33	1504	0	0	734
SKO-492-541	0	0	0	0	0	0
SUN-439-974	26.7	8.42	692	12.2	155.4	353
SUN-502-601	19	7.06	718	9.6	105	357
SUN-502-601	0	0	0	0	0	0
SUN-502-601	21.5	8.75	591	0	0	300
SUN-519-999	26.5	8.52	765	12.9	164.3	390
SUN-904-001	20.9	8.52	584	0	0	296
SUN-974-961	20.9	8.52	584	0	0	296
UIE-747-545	17	7.3	681	2.5	25.3	342
WIT-027-032	18.9	7.33	600	9.1	100	298
WIT-096-676	17.2	6.59	633	7.4	77.6	316
WIT-213-958	11.6	7.26	394	9	83.1	196
WIT-221-898	14.5	7.26	392	9.3	93	196
WIT-225-944	12.4	7.5	396	8.7	83.4	197
WIT-228-920	14.7	7.51	392	7.7	77.5	195
WIT-285-612	21.2	7.79	386	8.1	93	192
WIT-311-454	21.8	8.07	468	6.6	75.4	234
WIT-317-443	16.6	6.04	686	7.5	79	341
WIT-317-443	21.4	8.03	446	5.4	63.2	222
WIT-317-443	21.4	8.03	446	5.4	63.2	222
WIT-348-331	0	0	0	0	0	0
WIT-348-331	0	0	0	0	0	0
WIT-767-662	18.7	7.73	326	0	0	167
WIT-790-321	21.6	7.35	271	0	0	139
WIT-798-435	22.3	7.91	257	0	0	133
WIT-803-591	17.6	7.25	288	0	0	145
WIT-816-533	14.7	7.29	260	9.5	96.8	131
WIT-855-123	25.1	7.4	280	0	0	128
WIT-858-812	19	6.99	287	0	0	145
WIT-884-925	21.6	7.05	307	11.6	141.2	155
WIT-898-683	20.9	7.32	311	6.7	78.2	155
WIT-898-683	29.2	6.25	314	5.5	76.3	156
WIT-903-359	18.6	7.16	305	5.7	62.6	153
WIT-903-359	25.5	7.83	339	7.7	108	168

WIT-926-349	16.4	7.5	317	4.2	44.1	162
WIT-926-349	27.5	7.74	335	7.1	90.8	166

Appendix 3. Raw data (Substrate & vegetation type)

Site ID	Watercourse	Width	Length	Depth	Area fished	Substrate type						Vegetation type		
						Mud	Sand	Gravel	Stones	Boulder	Bedrock	Emergent	Floating	Submerged
DAR-040-630	Darlington	0	0	0	0	20	0	60	20	0	0	0	0	20
DAR-349-348	Darlington	0	0	0	0	80	0	5	5	10	0	0	0	0
DAR-509-132	Darlington	0	0	0	0	0	0	100	0	0	0	0	0	0
DAR-676-130	Darlington	0	0	0	0	0	0	100	0	0	0	0	0	0
DAR-784-175	Darlington	0	0	0	0	100	0	0	0	0	0	0	0	0
DAR-919-092	Darlington	0	0	0	0	0	0	100	0	0	0	0	0	70
DAR-930-111	Darlington	0	0	0	0	0	0	100	0	0	0	0	0	0
DAR-963-099	Darlington	0	0	0	0	100	0	0	0	0	0	50	0	0
DAR-988-115	Darlington	0	0	0	0	0	0	10	10	0	80	0	0	0
DAR-989-137	Darlington	0	0	0	0	0	0	25	50	0	25	0	0	0
GUIE-822-138	Groot Uie	5	50	40	25	10	0	0	45	45	0	0	0	0
GUIE-844-188	Groot Uie	7	50	100	70	5	0	0	25	70	0	0	0	0
GUIE-871-185	Groot Uie	4	10	80	40	10	0	0	40	50	0	0	0	0
GUIE-899-864	Groot Uie	6	25	30	150	5	0	0	5	90	0	0	0	0
GUIE-904-890	Groot Uie	4	20	30	40	0	0	0	10	90	0	0	0	0
GUIE-972-148	Groot Uie	5	10	30	50	5	0	0	20	75	0	0	0	20
KAB-351-765	Kabouga	1	7	30	7	100	0	0	0	0	0	0	0	0
KAB-855-777	Kabouga	50	100	2000	0	80	0	20	0	0	0	0	0	0
KAB-893-780	Kabouga	3	10	50	30	0	0	0	0	100	0	0	0	0
KOR-566-362	Sundays	0	0	0	90	100	0	0	0	0	0	15	0	0
KRO-166-771	Krom	3	5	15	15	0	0	0	75	25	0	2	0	0

KRO-167-761	Krom	3	10	30	15	90	0	0	10	0	0	0	0	10
KRO-300-708	Krom	2	12	20	12	90	0	0	10	0	0	5	20	0
KRO-310-710	Krom	2	10	25	20	50	0	0	25	25	0	5	0	0
KRO-550-520	Krom	4	10	15	28	89	0	0	10	1	0	0	0	0
KRO-787-578	Krom	4	12	20	48	5	0	0	95	0	0	0	0	0
KRO-981-261	Krom	10	30	400	0	90	0	0	10	0	0	0	5	0
KUIE-024-033	Klein Uie	7	30	40	70	10	0	0	0	90	0	15	0	0
KUIE-269-588	Klein Uie	3	10	60	0	20	0	0	5	75	0	0	0	0
KUIE-280-710	Klein Uie	5	15	25	25	5	0	0	15	80	0	10	0	0
KUIE-375-518	Klein Uie	5	10	100	0	0	0	10	10	40	40	0	0	0
KUIE-389-438	Klein Uie	2	5	15	0	20	0	0	10	70	0	0	0	0
KUIE-498-500	Klein Uie	8	100	80	100	0	0	0	10	90	0	5	0	0
KUIE-634-498	Klein Uie	3	10	10	30	0	0	0	0	100	0	0	0	0
KUIE-695-069	Klein Uie	1	7	60	7	20	0	0	0	45	35	0	0	0
KUIE-814-495	Klein Uie	2	15	50	30	20	0	0	5	30	45	40	0	0
KUIE-821-476	Klein Uie	3	20	50	60	5	0	0	5	0	90	5	0	0
KUIE-835-444	Klein Uie	5	30	150	10	0	0	0	0	0	0	20	0	0
KUIE-881-301	Klein Uie	8	100	100	40	5	0	0	5	90	0	5	0	0
SKO-239-421	Skoenmakers	0	0	0	0	75	0	25	0	0	0	0	0	0
SKO-437-534	Skoenmakers	0	0	0	0	75	0	25	0	0	0	50	0	0
SKO-440-851	Skoenmakers	0	0	0	0	25	0	75	0	0	0	0	0	0
SKO-463-727	Skoenmakers	0	0	0	0	15	0	85	0	0	0	0	0	0
SKO-464-718	Skoenmakers	0	0	0	0	15	0	85	0	0	0	0	0	0
SKO-464-723	Skoenmakers	3	10	150	15	10	0	20	0	0	70	0	0	0
SKO-492-	Skoenmakers	0	0	50	0	0	0	100	0	0	0	0	0	0

541														
SUN-439-974	Sundays	10	100	200	50	40	0	20	10	10	0	5	0	0
SUN-502-601	Sundays	50	120	0	0	0	0	0	0	0	0	40	0	5
SUN-502-601	Sundays	0	0	0	0	0	0	0	0	0	0	0	0	0
SUN-502-601	Sundays	0	0	0	0	0	0	0	0	0	0	0	0	0
SUN-519-999	Sundays	6	150	150	20	20	0	40	20	20	0	10	0	0
SUN-904-001	Sundays	20	200	300	100	40	0	10	0	40	0	10	0	0
SUN-974-961	Sundays	20	200	300	0	40	0	10	0	40	0	10	0	0
UIE-747-545	Uie	2	5	50	10	0	0	0	10	90	0	0	0	0
WIT-027-032	Wit	10	15	200	25	0	0	5	70	15	5	0	0	0
WIT-096-676	Wit	12	12	150	144	40	5	0	5	50	0	20	0	10
WIT-213-958	Wit	1.2	5	15	6	25	0	40	25	10	0	0	0	0
WIT-221-898	Wit	3.5	6.5	30	28	10	0	25	45	10	10	0	0	0
WIT-225-944	Wit	3	30	20	30	60	0	10	10	10	10	0	0	0
WIT-228-920	Wit	4	21	70	28	20	0	0	40	20	20	10	0	0
WIT-285-612	Wit	15	150	300	0	0	0	70	10	10	10	15	0	15
WIT-311-454	Wit	1.5	4	40	6	0	0	0	30	30	40	0	0	0
WIT-317-443	Wit	10	5	50	50	10	0	10	40	40	0	10	5	0
WIT-317-443	Wit	0	0	0	0	0	0	0	0	0	0	0	0	0
WIT-317-443	Wit	0	0	0	0	0	0	0	0	0	0	0	0	0
WIT-348-331	Wit	0	0	0	0	0	0	0	0	0	0	0	0	0
WIT-348-331	Wit	0	0	0	0	0	0	0	0	0	0	0	0	0
WIT-767-662	Wit	8	4	50	0	0	0	10	40	40	0	0	0	0
WIT-790-321	Wit	5	20	80	0	0	0	10	20	60	10	0	0	0
WIT-798-435	Wit	1	15	50	0	5	0	5	0	45	45	0	0	0

WIT-803-591	Wit	15	200	100	0	0	15	15	10	60	0	5	0	0
WIT-816-533	Wit	5	20	120	0	0	20	10	20	10	40	0	0	0
WIT-855-123	Wit	2	30	80	0	0	0	10	40	50	0	0	0	0
WIT-858-812	Wit	20	50	200	0	0	0	0	40	60	0	15	1	0
WIT-884-925	Wit	7	9	100	63	0	0	0	25	25	50	0	0	0
WIT-898-683	Wit	4.5	20	20	28	0	0	0	50	50	0	0	0	0
WIT-898-683	Wit	5	20	20	15	0	0	0	20	80	0	0	0	100
WIT-903-359	Wit	2.5	15	15	37.5	0	0	0	90	10	0	0	0	0
WIT-903-359	Wit	5	45	20	0	0	0	0	80	20	0	0	0	0
WIT-926-349	Wit	4	50	30	40	0	0	80	10	10	0	0	0	0
WIT-926-349	Wit	10	50	40	0	0	0	80	10	10	0	0	0	0

Appendix 4. Raw Data (number of fish sampled per site)

AM = *Anguilla mossambica*; BA = *Barbus anoplus*; BP = *Barbus pallidus*; CC = *Cyprinus carpio*; CG = *Clarias gariepinus*; GAE = *Gilchristella aestuaria*; GAF = *Gambusia affinis*; GC = *Glossogobius callidus*; LA = *Labeobarbus aeneus*; LC = *Labeo capensis*; LU = *Labeo umbratus*; MS = *Micropterus salmoides*; OM = *Oreochromis mossambicus*; PA = *Pseudobarbus afer*; TS = *Tilapia sparrmanii*.

Site ID	AM	BA	BP	CC	CG	GAE	GAF	GC	LA	LC	LU	MS	OM	PA	TS
DAR-040-630	0	0	0	0	0	9	15	14	0	0	1	0	15	0	0
DAR-349-348	0	0	0	0	0	0	3	13	0	0	0	0	8	0	0
DAR-509-132	0	0	0	0	0	25	0	0	0	0	4	0	12	0	0
DAR-676-130	0	0	0	0	0	4	0	0	0	0	4	0	21	0	0
DAR-784-175	0	0	0	0	0	3	0	0	0	0	0	0	10	0	0
DAR-919-092	0	0	0	0	0	13	0	1	0	0	0	0	0	0	0
DAR-930-111	0	0	0	0	0	5	0	0	0	0	3	0	8	0	0
DAR-963-099	0	0	0	0	0	0	0	0	0	0	0	0	5	0	0
DAR-988-115	0	0	0	0	0	9	1	8	0	0	0	0	0	0	0
DAR-989-137	0	0	0	0	0	3	0	1	0	0	16	0	50	0	0
GUIE-822-138	0	0	2	0	0	0	0	0	0	0	0	0	0	66	0
GUIE-844-188	0	0	0	0	0	0	0	0	0	0	0	0	0	81	0
GUIE-871-185	0	0	3	0	0	0	0	0	0	0	0	0	0	82	0
GUIE-904-890	0	0	2	0	0	0	0	0	0	0	0	0	0	273	0
GUIE-972-148	0	0	9	0	0	0	0	0	0	0	0	0	0	124	0
KAB-351-765	1	100	0	0	0	0	6	0	1	0	0	0	0	0	0
KOR-566-362	7	0	0	1	19	14	83	27	2	0	5	0	7	0	0
KRO-166-771	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
KRO-167-761	0	0	5	0	0	0	0	0	0	0	0	0	0	45	0
KRO-300-708	0	0	5	0	0	0	0	0	0	0	0	0	0	0	0
KRO-310-710	0	0	10	0	0	0	0	1	0	0	0	0	0	0	0
KRO-550-520	0	0	43	0	0	0	0	0	0	0	0	0	0	1	0
KRO-981-261	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0

KUIE-024-033	0	0	0	0	0	0	0	0	0	0	0	0	0	140	0
KUIE-280-710	0	0	0	0	0	0	0	0	0	0	0	0	0	61	0
KUIE-375-518	0	0	0	0	0	0	0	0	0	0	0	0	0	50	0
KUIE-389-438	0	0	0	0	0	0	0	0	0	0	0	0	0	23	0
KUIE-498-500	0	0	0	0	0	0	0	0	0	0	0	0	0	100	0
KUIE-634-498	0	0	5	0	0	0	0	0	0	0	0	0	0	38	0
KUIE-695-069	1	0	0	0	0	0	0	0	0	0	0	0	0	182	0
KUIE-814-495	0	0	1	0	0	0	0	0	0	0	0	0	0	54	0
KUIE-821-476	0	0	3	0	0	0	0	0	0	0	0	0	0	53	0
KUIE-835-444	0	0	0	0	2	0	0	0	0	0	0	0	0	1	0
KUIE-881-301	0	0	0	0	0	0	0	0	0	0	0	0	0	33	0
SKO-239-421	0	0	0	0	0	0	5	0	1	0	0	0	9	0	0
SKO-437-534	0	0	0	0	0	0	6	3	0	0	0	0	21	0	0
SKO-440-851	0	0	0	1	0	0	2	0	8	0	2	0	0	0	5
SKO-463-727	0	0	0	1	1	0	0	0	0	0	0	0	15	0	0
SKO-464-718	0	0	0	0	44	0	0	0	5	0	0	0	0	0	0
SKO-464-723	0	0	0	0	1	0	0	0	0	0	0	0	2	0	0
SKO-492-541	0	0	0	0	3	0	0	0	1	1	0	0	5	0	0
SUN-439-974	0	0	0	0	9	0	0	4	0	0	6	0	0	0	0
SUN-502-601	0	0	0	0	4	13	17	0	0	0	13	0	0	0	0
SUN-519-999	0	0	0	0	0	64	0	44	3	1	3	0	1	0	0
SUN-904-001	0	0	0	0	0	0	45	55	0	0	0	0	11	0	0
SUN-974-961	8	0	0	0	12	0	0	0	2	3	11	0	189	0	0
UIE-747-545	3	0	0	0	1	0	0	0	0	0	0	0	0	0	0
WIT-027-032	0	0	6	0	0	0	0	24	0	0	0	0	0	1	0
WIT-096-676	0	0	0	0	0	0	0	36	0	0	0	0	0	0	0
WIT-213-958	0	0	0	0	0	0	0	1	0	0	0	0	0	22	0
WIT-221-898	0	0	0	0	0	0	0	0	0	0	0	0	0	78	0
WIT-225-944	0	0	0	0	0	0	0	0	0	0	0	0	0	3	0
WIT-228-920	0	0	0	0	0	0	0	0	0	0	0	0	0	17	0
WIT-285-612	1	0	0	0	0	0	0	0	0	0	0	4	0	0	0
WIT-311-454	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0
WIT-317-443	9	0	0	0	0	0	0	7	0	0	3	8	0	0	0
WIT-348-331	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0

WIT-884-925	0	0	0	0	0	0	0	53	0	0	0	0	0	1	0
WIT-898-683	0	0	10	0	0	0	0	0	0	0	0	0	0	56	0
WIT-903-359	0	0	8	0	0	0	0	10	0	0	0	0	0	60	0
WIT-926-349	0	0	5	0	0	0	0	4	0	0	0	0	0	85	0