CONSERVATION OF THE NATIVE FRESHWATER FISHES OF THE CAPE FLORISTIC REGION (SOUTH AFRICA): MANAGEMENT OF NON-NATIVE SPECIES

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DECLARATION

I declare that the work presented in this thesis is entirely my own and that where assistance from others has been accepted, this is fully acknowledged.

Two papers have been published during the preparation of this thesis, and a third has been accepted for publication. Text from these papers has been included in the text of this thesis. I was the lead author on each of these papers and took responsibility for the data analysis. The contribution of my co-authors to these papers is acknowledged.

Marr, S.M., L.M.E. Sutcliffe, J.A. Day, C.L. Griffiths and P.H. Skelton. (2009). Conserving the fishes of the Twee River, Cederberg: revisiting the issues. *African Journal of Aquatic Science* **34**:77-85.

Marr, S.M., M.P. Marchetti, J.D. Olden, D.L. Morgan, E. García-Berthou, I. Arismendi, J.A. Day, C.L. Griffiths and P.H. Skelton. (2010). Freshwater fish introductions in mediterranean-climate regions: are there commonalities in the conservation problem? *Diversity and Distributions* **16**:606-619.

Marr, S.M., N.D. Impson and D. Tweddle. (in press). Review of the proposal to eradicate non-native fish from priority river areas in the Cape Floristic Region of South Africa. *African Journal of Aquatic Science* **37**.

I was one of the supervisors for Owen Davies' B.Sc. Honours project. This work has been used as a reference in this thesis. Selected studies included in Davies' work were repeated for this thesis.

Davies, O.R. (2007). *Threats to the Successful Recovery of the Critically Endangered Twee River Redfin (Barbus erubescens) in the Suurvlei River, Western Cape, South Africa*. B. Sc. Honours (Unpublished) University of Cape Town, Cape Town.

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ABSTRACT

Freshwater fishes are among the most threatened taxa in the world. Increasing demand for freshwater, habitat degradation and the introduction of non-native species, will continue to place pressure on the remaining native freshwater fishes. A meta-analysis estimated that more than 90% of river habitat in three major catchments has been invaded by non-native fish and that catchments covering less than 1% of Cape Floristic Region have no recorded non-native fish introductions, the major rivers containing 10 or more non-native species. The majority of the native fishes continue to be threatened by the presence of non-native fish.

Profound taxonomic and functional changes to freshwater fish assemblages in Mediterranean-climate regions resulting from non-native fish introductions were identified. Phylogenetic preference was exhibited for species selected with more than 90% of introductions originating from five taxonomic orders. The pathways for introductions were consistent across all Mediterranean-climate regions. The results show strong evidence of on-going taxonomic and functional homogenization of freshwater fish faunas. Characteristics suitable for risk assessment databases in Mediterranean-climate regions were identified.

The difficulties associated with attempting to identify the reasons for the decline of a critically endangered fish, even for a relatively simple system, were demonstrated. Water quality, pesticide exposure, instream and riparian zone habitat, and dietary overlap between native and non-native species were explored. The results of the study were inconclusive. The results do indicate that native and non-native species can co-occur in the complex habitat of one tributary, but not in the simple habitat of the other.

A social survey of freshwater anglers showed that angling for non-native species is important to the anglers and that they are not likely to switch to angling for native species, with the possible exception of Clanwilliam yellowfish. The anglers considered the conservation of native fishes extremely important but less than half believed that the conservation authority were doing a good job in conserving native fishes. The results indicate that support for the conservation authority's proposed piscicide project is low but that it could increase support for conservation projects by using large cyprinids as flagship species. Many options are available to the conservation authority to improve their efforts to manage non-native fishes in the region. The most important of these are the delineation of roles and responsibilities and the compilation of a comprehensive conservation plan.

DEDICATION

I dedicate this work to my parents Mr Murray Stuart Marr (1936-1994) and Mrs Wendy Ann Marr (nee Stronach), future generations who will live in what we now call the Cape Floristic Region and the native freshwater fishes of the Cape Floristic Region.

Dad, I know that you are proud of me and, although you did not express it much while you were with us, I know that you have taken great pride in all my achievements. I was honoured to have you as a father. I know that you did not fail me in anything and gave to me, and my siblings, only your best. You have given me a picture of a man who I can aspire to be and have shown us the true meaning of a successful marriage. I love and respect you. We all miss the sunshine of your laughter in our lives, but we are also aware that you are not far away.

Mom, you are love personified. The love you have shown me is beyond my full recognition. I am grateful to have you as my mother. Thank you for your belief in me and the encouragement you have given me through the last six years. Yes, it has been tough on all of us and there were times I felt guilty about it, but your faith in me has helped me through. Thank you for staying with us. The dream you had of me is becoming a reality. Now you can see me in a red gown. I love you and want you to know that I do.

To the future generations, may you too have the pleasure of knowing the native freshwater fishes of the Cape Floristic Region and the organisms that they will evolve to. May we, the current generation, take responsibility for our actions, and those of previous generation,, so that our children's children's children's children may know of the native fishes of this region.

To the native freshwater fishes of the Cape Floristic Region, may you live long and prosper in clean water free from non-native fishes. I want to give back to you all the waters that once were yours; to evolve as a gift to future generations.

TABLE OF CONTENTS

Page

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To the Examiners who had the courage to reject my first submission, thank you. At the time, your decision was not appreciated, but the product that has resulted is far superior, and far more useful than my first effort. Your decision has been tough on me, but hopefully I have learned from the experience and am a better scientist as a result.

To my supervisors: Jenny Day, Charles Griffiths and Paul Skelton. Thank you for the support and encouragement in the process of creating this thesis. I know that it has taken longer than expected and that at times it must have been frustrating, but I am sure that you will agree that the end product has the potential to make the contribution you expected of me.

To my family: Mom, Jeni, James, Lindy and Sharon, this work would not have been possible without your love and support. Thank you for allowing me the space and time to pursue a dream. Thank you for your support through all those dark times when I knew no light in my life. Thank you for allowing me to be myself. I find really difficult to be anyone else.

CHAPTER 1. INTRODUCTION

1.1 Introduction

Human-mediated transformation of the biosphere threatens biodiversity worldwide (Siegfried and Davies 1982, Stiassny 1999, Sala *et al.* 2000) and is changing regional variations in the climatic cycles that shape evolutionary and ecological processes in aquatic and terrestrial ecosystems (Siegfried and Davies 1982, Millennium Ecosystem Assessment 2005b). Human habitation now extends to the majority of the biosphere and forms an integral part of the ecosystems that surround them (Siegfried and Davies 1982). Yet, there is growing evidence that human activities are detrimentally impacting many ecological systems, raising concerns that these changes may detrimentally impact human well-being (Siegfried and Davies 1982, Millennium Ecosystem Assessment 2005c). Direct drivers of ecosystem change include changes in land-use; species introduction or extirpation; external inputs (e.g. fertilizer use, pest control, and irrigation); resource consumption; and climate change (Millennium Ecosystem Assessment 2005a). Indirect drivers of change are primarily demographic; economic; socio-political; scientific or technological; and cultural or religious (Nelson 2005).

1.1.1 The Biotic Dimension of Conservation

There are several reasons why biodiversity is crucial for human well-being (Hoffmann 2004): biodiversity is the main wellspring of food and medicine; species diversity is essential for the functioning of many of the earth's ecosystems (biologically diverse ecosystems are more productive, stable, and resilient to disruption); preserving biodiversity is important for historical, cultural, and aesthetic reasons; and there are strong moral arguments that the rights of non-human animal species should be respected and protected. The overarching goals of conservation are to ensure that the biosphere can continue to renew itself and provide the means for all life; to ensure human survival and well-being; and to retain potential evolutionary pathways (Siegfried and Davies 1982). Therefore, conservation traditionally focuses on the indefinite retention of the structure, function and composition of biomes at landscape, community or ecosystem, species or population, and genetic levels (Noss 1990).

Most threatened species receive little or no conservation intervention. The small number that do are biased towards well-studied, charismatic species native to the developed world (Sitas *et al.* 2009). Neither the conservation status of a species, nor its evolutionary distinctiveness, appear to be important in conservation decision-making (Harrison and Stiassny 1999, Stiassny 1999, 2002, Sitas *et al.* 2009). For example, 40% of the annual endangered species budget in the United States of America (USA) is directed to seven bird species, four of which are regional sub-species, whereas very little funding is directed less charismatic species from monotypic genera (Metrick and Weitzman 1996). Bird species have received comparatively more conservation funding due to focused efforts of the Non-Governmental Organization (NGO) community and the especially favourable perception of birds as species worthy of protection (Czech *et al.* 1998).

1.1.2 The Human Dimension of Conservation

Nature conservation cannot be considered separately from human society (Berkes 2004). On the one hand conservation of natural systems is required for the well-bring and survival of humans and their environment while, on the other, many people are concerned about the survival of species of plants and animals *per se* (Siegfried and Davies 1982). Conservation is thus for and about people. Human societies drive the degradation of ecosystems. If there were no human societies, there would be no need for conservation. The generally accepted paradigm of human interaction with the environment is based on the separate existence of the Environmental, Economic and Social systems (Figure 1.1a), suggesting that these systems may be treated independently (Mebratu 1998). The intersection between the three systems is the interface where conservation is practiced, whereas the systems do not interact outside this interaction zone. In this model, the ultimate objective of conservation is the full integration of the Environmental, Economic and Social systems. Mebratu (1998) proposed the "Cosmic Interdependence" model representing the Economic Cosmos as a circle wholly within the Social Cosmos, itself a circle wholly within a circle representing the Biotic Cosmos, a circle wholly within the circle of the Abiotic Cosmos (Figure 1.1b). The Cosmic Interdependence model suggests that human society, and the economic and social sectors in particular, never have been, and never will be, separate systems independent of nature (Mebratu 1998). The environmental crises recorded throughout human history are an outcome of human neglect of the systematic interactions. Due to limitations of Newtonian thermodynamics for closed systems, the human cosmos cannot grow without displacing a portion of the biotic cosmos (Czech *et al.* 1998, Ehrlich and Pringle 2008). The conflict between economic growth and biodiversity conservation can therefore be described as a trade-off between humans and non-humans in competition for natural resources to survive (Czech *et al.* 1998, Ehrlich and Pringle 2008).

Human societies desire economic growth for a variety reasons: some seek to supply their basic needs, while others seek to attain higher levels of affluence (Rosales 2008). Because economic growth is the increase in production and consumption of goods and services, the ecological impact of economic growth increases with size of the economy (Czech *et al.* 1998). Human ecological impacts on freshwater fish and other threatened species have been shown to increase with increased human population size (Hoffmann 2004, Clausen and York 2008). Increased population size affects the freshwater habitat upon which fish rely through the loss of wetlands, modification of flow regimes, and discharge of pollution (Clausen and York 2008). Many economists and political leaders are adamant that perpetual economic growth and technological development will alleviate poverty, maximize human welfare and reduce societal environmental impacts (Rosales 2008). Technological advancement has not, however, reduced the societal environmental impact of economic growth, because it is based on perpetuating existing technology (Czech 2008). Further, when economic growth is placed in its biophysical context, such as the availability of fresh water or mineral resources, a paradox emerges - perpetual economic growth is impossible (Ehrlich and Pringle 2008, Limburg *et al.* 2011). Given that economic and social issues are more likely to inform policy than ecological issues and that the same ethics that justify conservation also demand that we are mindful of poverty and associated human suffering (Chan *et al.* 2007), we need to consider how we can achieve the conservation of biodiversity within the constraints of expanding Human cosmos.

1.1.3 Limitations to Conservation Interventions

While conservation of ecosystems and species is necessary to maintain the functioning of the biosphere and human well-being, there are limitations to the outcomes achievable through conservation programmes. These limitations can be financial, personnel, technical and societal. This topic appears to be poorly discussed in the literature and no references discussing these limitations could be sourced. The most common limitation to conservation action is inadequate financial support to initiate, implement, and maintain conservation programmes. Beyond financial constraints, there is also a lack of adequately trained personnel of manage and staff conservation programmes.

Scientific understanding of the functioning of ecological systems is incomplete and places limitations on conservation programmes. A technical limitation on what is currently technologically feasible further limits conservation programmes. Conservation programmes are currently unable to re-instate extinct species or restore severely compromised landscapes and ecosystems. While many stressors acting on ecosystems or individual species can be reduced, some systems have been heavily modified or denuded, such that restoration to a near-pristine state is no longer possible (Cowx and van Zyll de Jong 2004). Extinction of keystone species, or the disruption of essential ecological processes such as soil formation, may result in lifeless systems beyond restoration. Identifying effective and appropriate conservation actions remains a technical challenge for conservation programmes (Minns *et al.* 1996).

Beyond technical feasibility, conservation interventions need to be socially feasible. To be socially feasible, conservation projects require social and political support in addition to not impacting the livelihood of members of the society, or the infrastructure upon which their livelihood depends. For example, a freshwater fish conservation project to restore migratory fish species may propose removing barriers and water abstraction to restore migration pathways and re-instate the natural flow regime of the river system. If there are agricultural or industrial ventures in the catchment that rely on water abstraction and the dams to store water during low flow periods, the project is unlikely to be deemed socially feasible. Further, if a conservation project restricts access of low income communities to resources, areas or specific species, the project is unlikely to be deemed ethically or socially feasible. Political support is important when there is social resistance to conservation projects. For example, political support for the successful marine bird conservation programme to eradicate cats from Marion Island ensured completion of the project, despite opposition in South Africa (Huntley 1999). Some conservation programmes may encounter resistance from stakeholder groups that have a vested interest in maintaining the current status quo, or benefit through economic gain. For example, freshwater angling is an important economic component of New Zealand's tourism industry and many angling groups oppose native fish conservation projects for fear of losing the economic benefit provided by the current fisheries based on non-native salmonids (Chadderton 2003).

1.1.4 Freshwater Ecosystems

The *circa* 125,000 species of freshwater animals that have been described represent 9.5% of known animal species (including a third of the vertebrate species) (Dudgeon *et al.* 2006, Balian *et al.* 2008). Despite freshwater habitats making up such a small fraction of the Earth's water and only 0.8% of the Earth's surface area, exceptionally high concentrations of biodiversity are found in rivers, lakes, and coastal wetlands (Stiassny 1999). The insular nature of freshwater habitats has led to the evolution of many species with small geographic ranges, often encompassing a single lake or catchment (Dudgeon *et al.* 2006). These highly diverse freshwater ecosystems are vulnerable and once disturbed deteriorate faster than terrestrial systems (Stiassny 1999).

The ecosystem services provided by freshwater ecosystems have been estimated to be worth about US$ 6.6 x 10^{12} per year globally, 20% of that provided by all the Earth's ecosystems (Costanza *et al.* 1987). Freshwater ecosystems have been modified abiotically (through flow regulation, water diversion and water abstraction) and biotically (through the introduction of non-native species and the extinction of native species) (Stiassny 1999, Cowx 2002, Dudgeon *et al.* 2006, Strayer and Dudgeon 2010). Human infrastructure now captures more than 50% of available fresh water runoff (Jackson *et al.* 2001); reservoirs retain 25% of the global sediment load (Vorosmarty and Sahagain 2000); and several major rivers no longer flow to the sea during dry periods (Wilcove and Master 2005). As a result, freshwater organisms are among the most threatened taxa worldwide (Bruton 1995, Stiassny 1999, Jenkins 2003, Strayer and Dudgeon 2010). The global consumption of fresh water will continue to increase and human impacts on freshwater ecosystems escalate as the human population continues to grow (Stiassny 1999, Suski and Cooke 2007, Ormerod *et al.* 2010, Strayer and Dudgeon 2010). There is thus an urgent need to assess the ecological status of freshwater ecosystems and determine how they are being affected by human transformations (Revenga and Kura 2003).

1.1.5 Freshwater Fish

Fish species constitute half of all known vertebrate species and occur in virtually every aquatic environment, but have received much less research and conservation attention than their terrestrial counterparts (Allendorf 1988, Harrison and Stiassny 1999, Stiassny 1999, 2002). The word '*fish*' describes a heterogeneous grouping of aquatic vertebrates comprised of hagfishes and lampreys, sharks, rays and chimaeras, and the finned bony fishes (Lévêque *et al.* 2008). Each continent has a distinctive freshwater fish fauna, as a result of physical barriers disrupting dispersal and differences in temperature adaptations (Berra 2007, Lévêque *et al.* 2008). Diversity is highest in the equatorial zone and decreases towards temperate and polar regions. Most oceanic islands are inhabited by species predominately of marine origin that have adapted to freshwaters (Lévêque *et al.* 2008).

At present, about 30,000 fish species are recognised, but some experts feel that the final number may be considerably higher (Nelson 2006, Berra 2007, Lévêque *et al.* 2008, Froese and Pauly 2010, Eschmeyer and Fricke 2011). Freshwater fishes (freshwater and strictly peripheral species) are currently estimated at almost 13,000 species (2,513 genera), or about 15,000 when all species occurring in fresh or brackish waters are included (Lévêque *et al.* 2008). The estimated 13,000 strictly freshwater species live in less than 1% of the earth's surface area, while the remainder live in salt water covering about 70% of the earth's surface.

The majority of species occur in five orders: Characiformes, Cypriniformes; Siluriformes; Perciformes; and Cyprinodontiformes (Nelson 2006, Berra 2007, Lévêque *et al.* 2008, Froese and Pauly 2010, Eschmeyer and Fricke 2011). Biogeographically, strictly freshwater species are distributed as follows: 4,035 species (705 genera) in the Neotropical region; 2,938 (390 genera) in the Afrotropical; 2,345 (440 genera) in the Oriental; 1,844 (380 genera) in the Palaearctic; 1,411 (298 genera) in the Nearctic; and 261 (94 genera) in the Australian (Lévêque *et al.* 2008).

The International Union for Conservation of Nature (IUCN) has estimated that 25% of evaluated freshwater fish species could be considered threatened with global extinction, a value exceeding that for all other vertebrate groups (Hinton-Taylor *et al.* 2009). The global extinction rate of freshwater fishes is also projected to rise to five times that of terrestrial vertebrates (Ricciardi and Rasmussen 1999). Traditional methods of conservation management (for example, regulation of exploitation, designation of protected areas, captive breeding programmes) are not as effective for freshwater fishes as they are for terrestrial animal groups (Cowx and Gerdeaux 2004). The migrations made by many fish species make them particularly challenging to conserve through area-based strategies (Schlosser and Angermeier 1995, Fausch *et al.* 2002). Many authors (Williams 1991, Lyle and Maitland 1992, Aparico *et al.* 2000, Keith 2000, Impson *et al.* 2002a) have concluded that the current systems of terrestrial protected areas do not offer adequate protection to threatened fish species or aquatic communities. Poor recruitment is one of the fundamental constraints on freshwater fish populations (Koehn 2003). Poor recruitment can arise from a lack of optimal spawning habitat, inadequate food resources, especially for larvae after they have absorbed the yolk sac, or high mortality of juvenile or other life stages (Cowx 2002). It is, therefore, important to understand the factors threatening freshwater fish populations.

1.1.6 Threats to Freshwater Fish

Freshwater fish populations are susceptible to extirpation through habitat degradation; water pollution; over-abstraction of water; habitat fragmentation by obstructions in the river (e.g. weirs and dams) and the presence of non-native fish species within the water body, particularly piscivores (Bruton 1995, Maitland 1995, Richter *et al.* 1997, Cowx 2002, Skelton 2002) - see Table 1.1. Human population and economic growth are the direct drivers of native fish declines (Limburg *et al.* 2011). Densely populated and rapidly expanding urban areas have resulted in aquatic habitat degradation (Lande 1998, Cowx 2002, Crivelli 2002). Leading drivers of native fish declines are also indicators of economic activity, such as surface water diversion, agriculture, invasive species, urbanization, and pollution (Limburg *et* *al.* 2011). Increased trade has resulted in the transfer of non-native species, leading to billions of US dollars of economic damages and declines in native biodiversity (Pimental *et al.* 2005). Socio-economic indicators (such as gross domestic product, human population density, and percentage urban area) have become better predictors of the diversity of non-native fish species than environmental factors commonly associated with native fish biodiversity (such as altitude, catchment area, or net primary productivity) (Leprieur *et al.* 2008a).

1.1.7 A Focus on Non-Native Fish Management

In the Cape Floristic Region, the primary study area for this thesis, there is a strong correlation between the presence of predatory non-native fish and the absence of native species (Impson 2007, Tweddle *et al.* 2009), a situation analogous with that in the Colorado River in the USA (Marsh and Pacey 2005). This immiscibility between native and non-native fishes is particularly evident in catchments where centrarchids (basses and sunfish), salmonids, and other piscivorous species have been introduced. Studies conducted in the Cape Floristic Region have shown that the presence of non-native fish affect the behaviour and composition of the native fish assemblages (Woodford and Impson 2004, Woodford *et al.* 2005, Shelton *et al.* 2008) and lower levels of the food web, including aquatic invertebrates and algae (Lowe *et al.* 2008).

Species rarely decline from one factor alone, but from a combination of several factors (Angermeier 1995, Moyle 1995, Schmutz *et al.* 2002). Habitat degradation and the presence of non-native fishes have been identified as the leading causes of native freshwater fish decline in the Cape Floristic Region (Impson 2007, Tweddle *et al.* 2009). The rivers of the Cape Floristic Region exhibit the predictable, seasonal progression typical of streams and rivers in Mediterranean-climate regions [see Sections 1.1.9 and 3.1]. As a result, aquatic biota are subjected to a rage of conditions varying from cold, highly oxygenated, low nutrient water winter floods to hot, oxygen depleted, high nutrient, stagnant water during the summer drought (Gasith and Resh 1999). The freshwater fishes of the Cape Floristic Region are typical of old, well established mountain faunas with a high degree of endemicity, geographically restricted ranges, relative inflexibility in life history styles, and lack anti-predator responses in the presence of piscivorous fish (Skelton 1987, 2002). Due to the nature of the rivers in the Cape Floristic Region and their native fishes, addressing habitat degradation without suitably addressing the presence of non-native fishes will not contribute to the conservation of the native fishes. The extent of non-native fish invasion is so large [see Section 3.3.2] and their presence so damaging to the native fish populations that the first step in any native fish conservation programme in the Cape Floristic Region has to be to adequately manage and reduce the populations of non-native fishes. Because the native fishes of the Cape Floristic Region cannot persist in the presence of predatory non-native fish, the remaining native fish populations can only be conserved, and their ranges increased, through the control, management or elimination of the non-native fish species (Impson 2007). This thesis, therefore, concentrates on the management of non-native freshwater fishes as a mechanism for the conservation of the native freshwater fishes in the Cape Floristic Region.

1.1.8 A Focus on Mediterranean-Climate Regions

The Cape Floristic Region, the primary study area, is one of five Mediterranean-climate regions: the Mediterranean Basin, California, central Chile, south-western and south-eastern Australia, and the south-western Cape of South Africa (Figure 1.2). Comparisons across climatically-similar regions remove large-scale drivers, such as regional climate (Pauchard *et al.* 2004), laying bare the ecological processes involved and elucidating the role of human activities (Jiménez *et al.* 2008). Habitat degradation, the impact of non-native species, and competition with humans for limited water resources are causing aquatic faunas in Mediterranean-climate regions to decline faster than in any other region in the world (Moyle 1995). Mediterranean freshwater fish species have thus evolved in harsh environments (e.g., facing severe droughts and floods) and have generally developed short lifespan, generalist habitat use, opportunistic feeding strategies, high fecundity and early sexual maturity (Hermoso *et al.* 2009). Non-native fish species currently represent more than a quarter of the total number of fish species per catchment in Mediterranean regions, which have been identified amongst the six global freshwater fish invasion hotspots (Leprieur *et al.* 2008a). Since the challenges associated with freshwater fish conservation are similar across Mediterranean-climate regions, this study has been limited to Mediterranean-climate regions to allow the lessons learned to be applied to the Cape Floristic Region. Since this work is couched so strongly in the context of Mediterranean-climate regions, a short introduction to this climatic region is provided next.

1.1.9 Mediterranean-Climate Regions

Mediterranean-climate regions are recognized hotspots of biodiversity and endemism (Cowling *et al.* 1996). Their isolation, by vast deserts or high mountains, have preserved the products of speciation events, resulting in relatively distinctive faunas that experienced little exchange of species with more temperate or equatorial areas (Di Castri 1991). With the exceptions of the south-western Cape and Australia, these regions have been biological refuges during glaciation events (Crivelli 1995). At the same time, these areas are among the most populated by humans, due to their mild climates and suitability for supporting highly-prized agricultural produce (such as deciduous and citrus fruit, winter wheat, table grapes, and wine). Human enterprise in Mediterranean regions has resulted in extensive habitat alteration and fragmentation and flow regulation using dams, in addition to the intentional and accidental introduction of many non-native species (Di Castri 1991). Mediterranean-climate regions, temperate grasslands, savannahs and shrub lands are the most transformed biomes (Millennium Ecosystem Assessment 2005a). By 1950, the native land cover of Mediterranean-climate regions had been reduced to 30% of the land area, with a further 2% being converted between 1950 and 1990 (Millennium Ecosystem Assessment 2005a). Mediterranean-climate regions also have some of the world's most skeletal terrestrial protected area networks (< 5% of the biome) and the remaining natural areas are under extremely intense development pressure (Underwood *et al.* 2009).

Mediterranean-type climates are young climates which were established in the Pleistocene (Di Castri 1991). They are governed by a symmetrical atmospheric circulation that produces a climate characterized by cool, wet winters and hot, dry summers (Gasith and Resh 1999). The moderating influence of the ocean keeps the winter temperatures mild and frosts are infrequent, except at higher elevations or inland. Seasonality and variability in rainfall are the principal attributes of the Mediterranean- climate. At least 65% and often 80% or more of the rain falls in the three months of winter, with most of the precipitation often falling during a few major storm events. Although the seasonal precipitation pattern is highly predictable, annual rainfall can vary markedly from year to year.

Rivers in Mediterranean-climate regions are physically, chemically, and biologically shaped by the unique, but predictable, natural seasonal sequence of abiotic (winter floods), biotic (declining flow in spring and summer), and abiotic (late summer drought) regulation that varies markedly in intensity between years (Gasith and Resh 1999). Although the biota is under abiotic pressure during floods, there are periods (spring-early summer) of moderate ecological conditions and high resource availability that allow the biota to recover from the floods and increase in densities. The increase in densities subsequently increases biotic pressures, such as competition and predation. If there is extreme drying, abiotic regulation returns. The seasonal sequence results in widely varying water chemistry variables including dissolved oxygen and nutrient levels. In late summer, flow can be reduced to such an extent that rivers, especially in the lower reaches, are reduced to stagnant isolated pools. The aquatic biota in Mediterranean-climate regions have adapted to the seasonal sequence through modified life histories or physiological tolerance (Gasith and Resh 1999).

1.2 Aims of this Thesis

This thesis seeks to evaluate the conservation of native freshwater fishes in the Cape Floristic Region through the management of non-native fish by engaging with the following questions:

- What is the status of the non-native fish invasion in the Cape Floristic Region, South Africa?
- Are there commonalities in the freshwater fish introductions in Mediterranean-climate regions and the outcomes of these introductions?
- Does a species introduction contribute more to the decline of an endemic-range restricted fish than habitat degradation?
- Can freshwater angling groups specifically targeting non-native species be convinced to target native fish?
- What mechanisms for the management of non-native fish could be adopted for the Cape Floristic Region?

1.3 Thesis Outline

The content of each chapter and the flow of ideas between chapters are presented below. In order to focus the discussion within this thesis, the core information is summarised within the chapters, with supplementary information supplied in the Appendices.

Chapter 1. Introduction
An introduction to the research topic is presented, outlining the need for the work and presenting the structure of the thesis.

Chapter 2. Introduction and Management of Non-Native Fish
This chapter provides a literature review of the socio-economic drivers for introductions, the vectors and pathways, factors influencing establishment success, the impacts of non-native fish introductions, landscape drivers of native species declines, and biotic homogenization of fish faunas and, the technical options available for the management of established non-native fish populations.

Chapter 3. The Cape Floristic Region and Its Freshwater Fish
This chapter introduces the primary study area, the Cape Floristic Region, and provides a summary of the native freshwater fish biodiversity of the region, the non-native fishes

introduced to the region, and an estimate of the current extent of non-native fish invasion. The history of freshwater fish conservation is summarised and current initiatives for the conservation of native fish populations and the management of non-native fish populations presented.

Chapter 4. Non-Native Fish in Mediterranean-Climate Regions

This chapter explores the hypothesis that non-native fish introductions in Mediterranean-climate regions have resulted in taxonomic and functional homogenization of the freshwater fish assemblages. The commonalities in the species successfully introduced, the outcomes of these introductions, and the vectors and pathways for the introductions were explored. The Mediterranean-climate regions included are: California, Chile, the northern Mediterranean Basin (the Iberian Peninsula to Turkey), south-western Australia and the south-western Cape (i.e. the Cape Floristic Region).

Chapter 5. Twee River Case Study – Non-Native Fish and Habitat Degradation

The threats to native fish populations are, in most cases, complex. Elucidating between the impacts of farming activities and those of non-native fishes can be challenging. In this chapter, the Twee River catchment in the Cape Floristic Region is explored as a case study to evaluate the hypothesis that the presence of non-native fish are the primary causes for the decline of the critically endangered Twee River redfin *Barbus erubescens* Skelton, 1984.

Chapter 6. Perceptions of Anglers

Angling groups are the primary beneficiaries of the non-native fish resources in the Cape Floristic Region. This chapter evaluates the hypothesis that freshwater anglers are prepared to repace their current non-native target species with native freshwater fishes of the region.

Chapter 7. Management of Non-Native Fish

This chapter discusses the management of non-native freshwater fish populations in the Cape Floristic Region in the context provided by the previous chapters. Recommendations are made for the management of non-native fish populations of the region.

Chapter 8. Summary and Conclusions

A summary of the findings of the thesis is presented and recommendations are made for further studies.

Figures

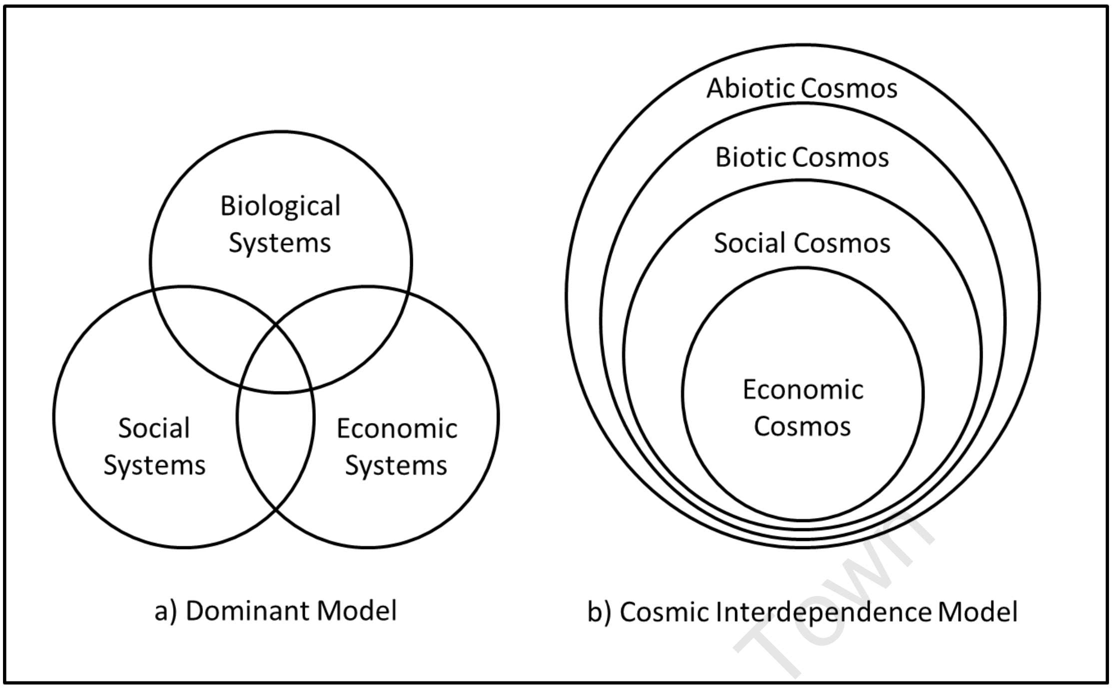


Figure 1.1: Conceptual models of the interaction between Society, Economy and Environment: a) the dominant model; b) the Cosmic Interdependence model (Mebratu 1998)

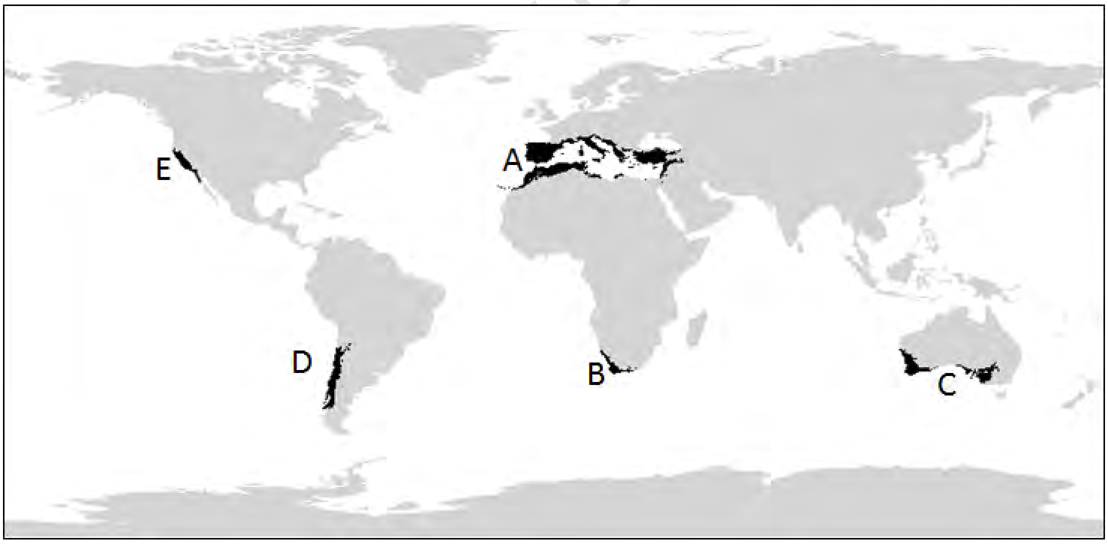


Figure 1.2: Location of Mediterranean-climate regions: the Mediterranean Basin (A), the south-western Cape in South Africa (B), south-western and south-eastern Australia (C), central Chile (D) and California in the USA (E).

Tables

Table1.1: Factors resulting in freshwater fishes becoming threatened as extracted from Bruton (1995), Crivelli and Maitland (1995), Maitland (1995), Moyle (1995), Richter *et al.* (1997), Cowx (2002), Crivelli (2002), Skelton (2002), and Limburg *et al.* (2011):

Factor	Examples
Impacts of introduced aquatic animals	Predation by non-native species, disruption of ecological processes, competition for food and space, physiological stress due to harassment or fin-nipping, infestation with non-native parasites and diseases, reductions in habitat quality, and genetic contamination through hybridisation
Water quality deterioration	Including organic, inorganic and thermal pollution from industrial, agricultural and domestic sources, eutrophication, acidification, salinisation, direct application of insecticides and herbicides in aquatic or riparian habitats, and atmospheric distribution of pesticides
Altered flow regimes and hydrological manipulations	Including the impacts of engineering works, impoundments, irrigation schemes, infilling, logging, water abstraction and drainage, stream diversion, interference with water flow regimes and fish migration paths, dredging and canalisation of rivers and streams, and flood control
Physical habitat degradation and fragmentation	Including water abstraction, dam building, channelisation, land use changes, deforestation, bank erosion, siltation, increased turbidity, de-oxygenation, sedimentation of spawning grounds, overgrazing and other forms of catchment mismanagement, urbanisation, disruption by human recreational activities such as boating and bait collection, removal of riparian vegetation, draining or salinisation of wetlands, and gravel extraction
Overexploitation	As a result of commercial, subsistence and recreational overfishing of target species and by-catches, poaching, collecting for the aquarium trade, destructive fishing practises, such as the use of dynamite and cyanide, and international trade in rare species
Global effects and climate change	Including the impact of increased UV radiation on plankton dynamics, altered hydrological and temperature regimes, and the effects of El Niño

CHAPTER 2. INTRODUCTION OF NON-NATIVE FISH – A LITERATURE REVIEW

The introduction of non-native fishes to natural water courses has been associated with the decline and extirpation of native fish species, resulting in the substitution of endemic species with widespread non-native species (Gido and Brown 1999, Olden *et al.* 2004, Light and Marchetti 2007). As discussed in the introduction, non-native fish are the single greatest threat to the native freshwater fishes of the Cape Floristic Region. In order to develop an effective strategy for the management of non-native freshwater fishes it is important to understand why and how they are introduced, the factors influencing the success of these introductions, and their consequences.

2.1 Introduction

Non-native fish introductions are strongly associated with numerous human activities, including international shipping, ornamental fish imports, aquaculture, biological control of mosquitoes and aquatic plants, development of new fisheries, irrigation schemes, shipping canals and locks connecting previously separate systems, and inter-basin transfer schemes (Allan and Flecker 1993, Maitland 1995, Ruesink 2005, Jeschke and Strayer 2006, Stohlgren *et al.* 2006). Socio-economic drivers determine why non-native fish are introduced into a new region. These drivers involve four main users of freshwater fishes: aquaculture, the ornamental fish trade, fisheries (commercial, subsistence, and recreational), and biological control programmes. Freshwater fish introductions are unique in that continued releases of non-native species often persist even after their invasive potential has been well documented (Lintermans 2004) indicating that the socio-economic drivers are greater than the motivation to conserve threatened native species.

For a species to successfully establish a self-sustaining population in a new environment, it must survive a series of events: capture in the native environment, transport to the new environment, release into the new environment, survival in the new environment, establishment of self-sustaining population in the new environment, and subsequently spread into areas beyond the initial establishment area (Copp *et al.* 2005a, Moyle and Marchetti 2006). The introduction-establishment process represents a formidable set of barriers that introduced species need to overcome (Moyle and Marchetti 2006) and consequently, most introduction attempts fail (Moyle and Light 1996, Williamson and Fitter 1996a, Moyle and Marchetti 2006). Williamson and Fitter (1996a) proposed the “Tens Rule” for species introductions, i.e. about 10% of species imported will escape; of those escaping about 10% establish self-sustaining populations in the wild; and of the species establishing in the wild 10% will become invasive or problem species. There are many reasons why species fail to establish self-sustaining populations [see Section 2.1.5] and it is important to understand these in order to inform the management of non-native fish populations and develop appropriate risk assessment protocols to evaluate proposed future introductions.

2.1.1 Defining Non-Native Species

A **non-native species** is generally defined as '*a species, [sub-species, race or variety] that does not occur naturally in a geographical area*' whereas a **native species** is generally defined as '*a species, [subspecies, race or variety] that occurs naturally in a geographical area, with dispersal occurring independent of human intervention, whether direct or indirect, intentional or unintentional*' (Copp *et al.* 2005a). However, Copp *et al.* (2005a) questioned whether strict biogeographical and ecological definition of a non-native species is appropriate. In many countries, non-native fish, for example common carp *Cyprinus carpio* Linnaeus, 1758 and goldfish *Carassius auratus* (Linnaeus, 1758), have inhabited natural water courses for a number of centuries. Could these be now considered 'honorary' native species? Does the length of time that a species has been present in a country come into consideration? Does the commercial or cultural (social-recreational) value of a non-native fish species affect its status? Socio-economic and political perspectives influence public perception of which species are beneficial, or should be considered part of natural systems (Copp *et al.* 2005a). A clear definition of what constitutes a non-native species is required when considering which species require active management. South Africa's National Environmental Management: Biodiversity Act (NEM:BA) defines a non-native species, described as an "alien" species, as:

- *"A species that is not a native species; or*
- *A native species translocated, or intended to be translocated, to a place outside its natural distribution range in nature, but not a native species that has extended its natural distribution range by natural means of migration or dispersal without human intervention."*

A non-native fish is therefore any fish species that has been introduced, or translocated, outside of its natural distribution range by human actions.

2.1.2 A Model for Non-Native Species Introductions

Colautti and MacIsaac (2004) developed a conceptual framework for the introduction-establishment-invasion process based on the concept of 'propagule pressure' (Williamson and Fitter 1996b, Richardson *et al.* 2000, Kolar and Lodge 2001). [See section 2.1.6 for discussion on propagule pressure.] Under the paradigm of 'propagule pressure' the number of individuals (propagules) for a given introduction attempt, the number of introduction attempts, and the frequency of introduction attempts, are considered to be the critical parameters that determine whether introduced species survive the filters that allow transition to subsequent stages (Figure 2.1). Species begin as native residents in a donor region (Stage 0). A portion of these native populations (propagules) are incorporated into transport vectors (Stage I), usually mediated by human activities. If these propagules survive the transport vector and released into the new environment outside their native range they become 'introduced' (Stage II), and have the potential to establish self-sustaining populations or become 'established' (Stage III) in the new environment. Following establishment, a combination of biotic and abiotic factors, in conjunction with propagule pressure, determine whether a species remains localized in low numbers (Stage III), becomes widespread yet in low numbers (rare) (Stage IVa), localized but dominant (Stage IVb), or widespread and dominant (Stage V). Local or regional dispersal of individuals determine which established species (Stage III) become widespread rare species (Stage IVa), or which locally dominant species (Stage IVb) become widespread and dominant (Stage V). Environmental and community-related factors determine which established species (Stage III) become locally dominant species (Stage IVb) or which widespread rare species (Stage IVa) become widespread and dominant (Stage V). Under this framework, a non-native species may be localized and numerically rare (Stage III), widespread but rare (Stage IVa), localized but dominant (Stage IVb), or widespread and dominant (Stage V). Some non-native species have an immediate impact on the ecosystem into which they have been introduced, even at low densities (lines A and B in Figure 2.2), while other non-native species only start exerting noticeable impacts on the ecosystem after they have exceeded a minimum threshold density (line C in Figure 2.2) (Bomford and Tilzey 1997). Management actions may be required to prevent unsustainable loss of native species even when the non-native species is at low densities in the recipient environment.

2.1.3 Non-Native Species Invasions

Biological invasions are complex processes that involve multiple interactions between the invading non-native species and the physical and biological characteristics of the recipient environment (Hayes and Barry 2008). Biological invasions are considered the second-most important global driver of species extinction, after habitat destruction (Vitousek *et al.* 1997), although they may be more important than habitat destructions in many regions. Invasive species are native or non-native species that establish a new range in which they proliferate, spread, and persist to the detriment of the environment, resulting in significant changes in the composition, structure, or ecosystem processes, or cause severe economic losses to human activities (Mack *et al.* 2000, Copp *et al.* 2005a, Strayer *et al.* 2006). Invasive species affect recipient systems at the organismal level by altering the native species' habitat use and foraging; at the population level by inducing changes in the abundance or distribution of native species; at the community level by altering both direct and indirect interactions among populations and inducing trophic cascades; and at the ecosystem level by changing the pathways and magnitude of movements of energy and nutrients (Simon and Townsend 2003). In reference to Colautti and MacIsaac (2004)'s model, invasive species are species at Stages IVa, IVb, or V and can cause significant changes in the recipient system at each stage.

Freshwater communities are particularly vulnerable to non-native species introductions (Saunders *et al.* 2002, Skelton 2002, Cambray 2003b). The establishment of non-native fishes into formerly fishless habitats has impacted the behaviour, distribution, and abundance of native species (micro-crustaceans, aquatic invertebrates, and amphibians), including the near-extirpation of large, active species, behavioural changes in other species to avoid microhabitats frequented by the non-native fish during daylight (Simon and Townsend 2003), and ecosystem functioning (Strayer 2010). Piscivorous fish introduced into systems containing native fish have reduced abundance or extirpated native species, especially in systems with no native piscivores (Strayer 2010). Non-native fishes affect their recipient environments by altering primary producers, zooplankton communities, stream invertebrates, and the composition of aquatic vegetation; reducing exchanges with neighbouring ecosystems; and altering light and nutrient availability by bioturbation and nutrient excretion (Simon and Townsend 2003, Baxter *et al.* 2004, Baxter *et al.* 2005, Baxter *et al.* 2007, Strayer 2010). The introduction of non-native freshwater fishes has resulted in impacts at genetic (gene transcription, hybridization); individual (behaviour, morphology, vital rates); population (transmission of parasites, invasive fish as parasites, demographic effects, distributional effects); community (species extinctions, compositional changes, alterations of food webs); and ecosystem (biochemical cycles, energy fluxes between ecosystems, ecological engineering) levels (Cucherousset and Olden 2011).

Biotic invasions also have economic impacts, through losses in potential economic output and the direct cost of combating invasions (Mack *et al.* 2000). The loss of potential income as a result of non-native fish species on the US$ 69 billion sports fishing industry in the USA was estimated to be US$ 5.4 billion per year (Pimental *et al.* 2005). However, this estimate does not include the cost of expensive control programmes for non-native fishes, such as those for the Colorado River Basin in the USA (Mueller 2005).

2.1.4 Lag-Times in Biological Invasions

Non-native species may exhibit a lag-phase between establishment and adverse effects becoming apparent (Crooks and Soule 1999, Crooks 2005). Some species may require a threshold population size before the invasion can proceed (Simberloff 2009). Allee effects – behavioural interactions that fail when population densities fall below threshold levels – may explain both the failure to establish and the lag time before spread or becoming dominant in the community (Leung *et al.* 2004, Simberloff 2009). Changes in the environment - such as flow, temperature, or nutrient regimes - could trigger an invasion (Simberloff 2009). Alternatively, the invasion may be facilitated by the presence of other non-native species, or the arrival of a new species, termed '*invasional meltdown*' (Simberloff and Von Holle 1999, Simberloff 2006).

2.1.5 Factors Influencing Establishment Success of Freshwater Fish Introductions

Considering that the environmental and economic costs of successful invasions are high (Pimental *et al.* 2000, 2005), it is important that the factors influencing the successful establishment and subsequent spread of non-native species are understood (García-Berthou 2007) in order to prevent future introductions and reduce the impacts of recently-introduced species. To identify the traits that determine establishment success, species established in a region should be compared to species that were introduced but failed to establish, rather than all species that could potentially be introduced (García-Berthou 2007). Unfortunately, many species introductions are neither authorised (illegal) nor adequately documented and, as a result, failed introductions are rarely recorded, limiting the ability to draw conclusions from the available data (Mack *et al.* 2000, Kolar and Lodge 2001, García-Berthou *et al.* 2005).

Several studies have evaluated the traits of successful invasive animals from both theoretical (Ehrlich 1989) and empirical data (Kolar and Lodge 2001, Hayes and Barry 2008). Although characteristics of the introduced species and characteristics of the recipient environment (local biotic community and environmental attributes) have been shown to be important in explaining introduction success (Simberloff 2009), it is now recognised that 'propagule pressure' is the most important determinant of introduction success (Cassey *et al.* 2005, Lockwood *et al.* 2005, Colautti *et al.* 2006, Simberloff 2009) [see Section 2.1.6].

There are many reasons why species fail to establish self-sustaining populations, including natural fluctuations of small populations (demographic stochasticity), impacts of extreme environmental events (environmental stochasticity), marginal or unsuitable habitat at introduction sites, and failure to disperse from the point of initial introduction (Lande 1993, Shea and Chesson 2002). Demographic stochasticity incorporates random fluctuations in birth rate, death rate, and sex ratio (Lande 1993, Lockwood *et al.* 2005, Simberloff 2009). In a small population, there is greater probability that the entire population will die in the next interval, or that all offspring will be of the same sex, than in a larger population (Lande 1993, Lockwood *et al.* 2005, Simberloff 2009). Environmental stochasticity, including unusual weather events - floods, fires, etc. - can totally eliminate founder populations (Lande 1993, Lockwood *et al.* 2005, Simberloff 2009). However, the arrival of further propagules minimizes the impact of such random environmental events on the establishment of a founder population (Lockwood *et al.* 2005, Simberloff 2009).

For freshwater fish, theoretical studies proposing factors influencing establishment success, e.g., Moyle and Light (1996), have been evaluated against empirical data to test the hypotheses proposed, e.g., Marchetti *et al*. (2004b, c). Studies to ascertain whether there are consistent predictors of successful freshwater fish introduction have been conducted at several scales: single system (e.g. the Laurentian Great Lakes of North America: (Kolar and Lodge 2002)), regional (e.g. California (Marchetti *et al.* 2004b, c), the Colorado River system USA (Olden *et al.* 2006), and the Iberian Peninsula (Alcaraz *et al.* 2005, Vila-Gispert *et al.* 2005, Ribeiro *et al.* 2008)) and global (Ruesink 2003, 2005). These studies have evaluated an extensive set of traits of the non-native organism and the ecosystem being invaded in an attempt to predict why particular species are more successful at establishing self-sustaining populations than others. Two studies reviewed this body of work: Moyle and Marchetti (2006) reviewed the Californian studies of Marchetti *et al*. (2004b, c), whereas García-Berthou (2007) reviewed the methodological approaches and conclusions of 12 quantitative studies.

Moyle and Marchetti (2006) showed that species are more likely to be successful if they: have a history of successful establishment, are highly desirable to humans, thrive in human-altered habitats, if there is a climatic match between source and recipient environments, and if more than 100 individuals are repeatedly introduced. García-Berthou (2007) found that the predictors of establishment success identified from most studies are reproductive variables (e.g. parental care), diet breadth, or environmental tolerances. The lack of data on propagule pressure and the use of sequential techniques for observational data sets were identified as limitations of these studies (García-Berthou 2007). All published studies are from temperate latitudes in the northern hemisphere. Comparative studies between different regions, or over multiple scales, are lacking (García-Berthou 2007). Both studies concluded that no universal set of characters predicts which introductions will succeed, or which non-native fish are likely to become problematic.

The results emerging from these studies are idiosyncratic, with different studies revealing different combinations of factors as the best predictors of establishment success (Ruesink 2005, Moyle and Marchetti 2006). The selective nature of fish introductions (Alcaraz *et al.* 2005) results in establishment rates that are much higher than those for other vertebrate taxa (García-Berthou *et al.* 2005, Jeschke and Strayer 2005). Non-native fishes are often introduced intentionally, with strong taxonomic biases towards game or forage fishes, species of human interest, or species that have been introduced elsewhere (Rahel 2000, 2002, Alcaraz *et al.* 2005, Clavero and García-Berthou 2006). Many species not expected to establish are simply not introduced. As a result, the importance of species-specific environment matching may be underestimated (Bomford *et al.* 2010). However, propagule pressure is now recognised as the confounding effect on establishment success (Simberloff 2009).

2.1.6 Propagule Pressure

Propagule pressure is a measure of the magnitude of the introduction effort of a species to a new region and incorporates the number of individuals per arrival event, the number of arrival events, and their frequency (Cassey *et al.* 2005, Lockwood *et al.* 2005, Colautti *et al.* 2006, Simberloff 2009). Propagule pressure is now considered the primary determinant of the successful establishment of introduced species (Cassey *et al.* 2005, Lockwood *et al.* 2005, Colautti *et al.* 2006, Simberloff 2009), particularly deliberate introductions (Lockwood *et al.* 2005), and is widely accepted as the '*null model*' for biological invasions (Cassey *et al.* 2005, Lockwood *et al.* 2005, Colautti *et al.* 2006, Simberloff 2009).

An increase in both the number of individuals per arrival event and number of arrival events decreases the likelihood that either demographic or environmental stochasticity will eliminate the founder population (Lockwood *et al.* 2005, Simberloff 2009). A particularly disastrous environmental event (e.g. fire, flood, or hurricane) is likely to extirpate most, if not all, established individuals in a single event, but is unlikely to affect subsequent arrivals (Lockwood *et al.* 2005, Simberloff 2009), thereby reducing the impact of environmental stochasticity. Similarly for demographic stochasticity. However, increasing the number of individuals per arrival event is likely to reduce the risk of demographic stochasticity more than the number of arrival events (Lockwood *et al.* 2005, Simberloff 2009). Moyle and Marchetti (2006) found that 70% of all of non-native fish introductions consisting of fewer than 100 individuals failed. However, introducing sufficiently large numbers of individuals frequently may not guarantee success (Simberloff 2009). For example, three species - Ayu *Plecoglossus altivelis* (Temminck and Schlegel, 1846), Atlantic salmon *Salmo salar* Linnaeus, 1758, and Bonneville cisco *Prosopium gemmifer* (Snyder, 1919) - failed to establish in California despite repeated large introductions (> 1000 individuals per introduction event) into seemingly appropriate habitats (Moyle and Marchetti 2006). Species that are introduced more frequently and those dispersing away from the introduction sites have a higher establishment success (Colautti 2005, Korsu and Huusko 2009). For ornamental fish, availability (frequency in shops, human population density, and the number of imports of ornamental fish into a region) is directly related to the frequency of introduction and establishment (Duggan *et al.* 2006, Copp *et al.* 2010). Further, easily-bred, hardy species (e.g. poeciliids) and larger-bodied species are over-represented among introductions relative to their frequency in the pet stores (Duggan *et al.* 2006).

Conservation genetics predicts that the risk of extirpation from inbreeding depression or genetic drift is increased in small populations (Allendorf and Lundquist 2003, Roman and Darling 2007). Yet, many widespread, invasive populations originated from a small number of individuals (Simberloff 2009). However, demographic and environmental stochasticity are more likely to extirpate founder populations than genetic factors (Lande 1993, Roman and Darling 2007, Simberloff 2009). Increased genetic variation from continued propagule pressure may negate inbreeding depression and genetic drift, leading to genotypes better adapted to the local or adjacent environment, enhancing the likelihood of persistence and future spread (Roman and Darling 2007, Simberloff 2009). Many non-native species have greater genetic variability in their introduced range than in their native range (Simberloff 2009). New genetic material from the arrival of new propagules is now thought to be a source of invasive genotypes; a likely explanation for the lag times in biological invasions [Section 2.1.4] (Simberloff 2009).

Propagule pressure is poorly recorded for most intentional introduction attempts and very rarely recorded for failed, illegal, or accidental introductions (Cassey *et al.* 2005, Lockwood *et al.* 2005, Puth and Post 2005, Colautti *et al.* 2006, Simberloff 2009). Because of its importance in establishment success, a metric defining the expected propagule pressure, such as the "*level of introduction effort*" variable used by Ribeiro *et al.* (2008), or the model developed by Gertzen *et al*. (2008), could be used as a surrogate for the expected propagule pressure.

2.2 Socio-Economic Drivers for Freshwater Fish Introductions

Freshwater fish are the most widely introduced aquatic animal group (Gozlan 2008) and continue to be introduced even though their detrimental impacts are well documented (Lintermans 2004). These introductions are driven by societal demands for fish products - food aquaculture (51% of species), ornamental fish (21%), recreational angling (12%), and fisheries (7%) (Gozlan 2008, Gozlan *et al.* 2010b). The motivation for non-native fish introductions is determined by socio-economic drivers that vary among the four main users of freshwater fishes: aquaculture, the ornamental fish trade, fisheries (commercial, subsistence and recreational), and biological control programmes and these are expanded on below.

2.2.1 Aquaculture

The increasing global demand for finfish as an affordable source of animal protein, or for their perceived health benefits, is driving the global increase in aquaculture (Delgado *et al.* 2003, De Silva *et al.* 2009, FAO 2010, Welcomme *et al.* 2010, Welcomme 2011). Declining marine stocks (Pauly *et al.* 2003, FAO 2010), increases in the cost of landed finfish (Pauly *et al.* 2003), and the rapidly expanding human population are all likely to increase the demand for freshwater aquaculture of finfish. Since the 1980s, aquaculture has been supported as the means of providing the shortfall between the marine fisheries supply and the global demand for finfish (De Silva *et al.* 2009). Globally, aquaculture is now the fastest growing primary industry (Tacon *et al.* 2010) and by 2009 produced 46% of all finfish consumed (FAO 2010). Aquaculture production has grown at an average rate of 9% per annum between 1950 and 2009 (De Silva *et al.* 2009, Diana 2009, FAO 2010). Asia dominates aquaculture production, contributing about 300 million tonnes of freshwater fish in 2004 (Gozlan 2008, FAO 2010), more than 90% of the global production (De Silva *et al.* 2009, Diana 2009, FAO 2010).

2.2.2 Fisheries – Commercial and Subsistence

Inland capture fisheries production has grown linearly at 3% per annum since 1950 (Welcomme *et al.* 2010, Welcomme 2011) producing more than 10 million tonnes of fish and crustaceans in 2008 (FAO 2010). Inland fisheries provide employment, nutrition and income to large numbers of rural households and are characterized by small-scale/household-based ventures. Participation in these fisheries is high; more than 56 million people were directly involved in inland fisheries in developing countries in 2009 (Welcomme *et al.* 2010), the bulk of the catch being consumed locally. Inland fisheries constitute one economic element of the livelihoods of traditional farming and fishing communities (Welcomme *et al.* 2010).

2.2.3 Fisheries – Recreational

Recreational fisheries are the capture, and possibly harvesting, of aquatic animals as recreation, without intentional economic gain (Cowx *et al.* 2010). Recreational fisheries have a high socio-economic and socio-cultural importance, but these benefits are difficult to group, quantify or evaluate (Arlinghaus *et al.* 2002), especially in economic terms. The economic and social benefits derived from recreational fisheries include classical economic benefits of angling-related industries (such as tackle shops, angling venues, travel, and hospitality establishments) in addition to the social and psychological welfare of anglers (Cowx *et al.* 2010). A larger number of people participate in recreational fisheries, an estimated 10.6% of the population in industrialized countries (Arlinghaus and Cooke 2009). A recent survey estimated the commercial value of South Africa's recreational fisheries (deep sea, rock and surf and inland combined) to be three times that of rugby and cricket combined (Leibold and van Zyl 2007).

Protection of the environment and aquatic biodiversity are generally strongly supported among recreational anglers (Granek *et al.* 2008, Cowx *et al.* 2010) because anglers have a vested interest in preserving or enhancing the resources they utilise (Cowx *et al.* 2010). The recreational fishing sector is a powerful lobby for the conservation or rehabilitation of damaged aquatic ecosystems (Cowx *et al.* 2010). However, recreational anglers explicitly targeting non-native species can become powerful opponents of native fish conservation programmes, such as in the Colorado River basin (USA) (Clarkson *et al.* 2005, Marsh and Pacey 2005) and in the south-western Cape (South Africa) (Cambray 1997a, 2003a, b). Widespread introduction of non-native fish for recreational angling has frequently occurred without adequate environmental impact assessments or monitoring (Cambray 2003b), resulting in subsequent losses of biodiversity in many systems (Cambray 1997a, 2003a, b, Clarkson *et al.* 2005, Marsh and Pacey 2005).

2.2.4 Ornamental Fish Trade

The ornamental fish trade is now recognized as a major pathway for the introduction of freshwater organisms (Copp *et al.* 2010, Strecker *et al.* 2011). Ornamental fish are a worldwide business (Chapman *et al.* 1997, Pelicice and Agostinho 2005), estimated to be worth US$ 25 billion-per-year worldwide (Padilla and Williams 2004) and growing at 14% annually (Padilla and Williams 2004, Cohen *et al.* 2007, Whittington and Chong 2007, Strecker *et al.* 2011). An estimated 4 000–5 000 freshwater fish species have been kept in aquaria (Pelicice and Agostinho 2005, Whittington and Chong 2007) and about 350 million fish are traded each year (Pelicice and Agostinho 2005). Although the majority of freshwater fish traded are produced by fish farms in Singapore, Malaysia, Japan, Israel and the USA, large numbers are still removed from the wild (7-8% of market), especially in South America (Brazil, Peru and Colombia) and Africa (Nigeria, Congo, Democratic Republic of Congo) (Andrews 1990, Gerstner *et al.* 2006, Whittington and Chong 2007), where collection from wild populations is cheaper than captive breeding (Ng and Tan 1997).

The ornamental fish trade and its associated industries are a major source of overseas income for areas of Africa, South America and South East Asia (Andrews 1990), employing thousands of people (Pelicice and Agostinho 2005). It is estimated that some 3 000 families make a living from and 100 000 persons benefit economically from the trade, often in villages where few other economic opportunities are available (Gerstner *et al.* 2006). Wild-caught aquarium fishes are potentially one of the few sustainable resources in the Amazon basin (Gerstner *et al.* 2006). Unlike some other vertebrates, freshwater fish can be sustainably harvested provided that sufficient tracts of their natural habitat (including breeding grounds) are conserved and collection takes place during the dry season when many fish populations suffer natural die-backs (Ng and Tan 1997).

Ornamental fish are kept in 14% of British homes, 12–14% of Australian homes, and 8-10% of US homes (Andrews 1990, Lintermans 2004, Rixon *et al.* 2005, Gerstner *et al.* 2006). Because of the widespread dispersal of ornamental fish to homes and businesses, these fish can be released into all freshwater habitats (Padilla and Williams 2004).

2.2.5 Biological Control

Many freshwater fish species have been introduced for the biological control of mosquitoes e.g. mosquitofish *Gambusia affinis* (Baird and Girard, 1853) and *G. holbrooki* Girard, 1859), aquatic weeds e.g. grass carp *Ctenopharyngodon idella* (Valenciennes, 1844), Asian carp

(*Aristichthys nobilis* (Richardson, 1845) and *Hypophthalmichthys molitrix* (Valenciennes, 1844) and various cichlid species e.g. red breasted tilapia *Tilapia zilli* (Gervais, 1849), and nematodes in catfish ponds in the United States e.g. Asian carp (Pípalová 2006, Conover *et al.* 2007, Pyke 2008, Whittier and Aitkin 2008, Froese and Pauly 2010). However, most fish species released as biological control agents do not meet the basic requirements for biological control – target specificity. The target organism may be included in their diet, but fish diets are usually broader than a single organism.

2.3 Vectors and Pathways for Freshwater Fish Introductions

The dispersal pathways for non-native species have been identified as areas where management actions could reduce the introduction of potentially harmful species (Hulme *et al.* 2008), but the dispersal pathways have been poorly studied (Puth and Post 2005). Hulme *et al.* (2008) identify three broad mechanisms by which a species may be introduced, directly or indirectly, as a result of human activity: importation of a commodity (the non-native species itself may be the commodity), the arrival of a transport vector (ship, train, aircraft or motor vehicle), and/or natural spread from a neighbouring region. These three mechanisms result in six principal pathways:

- Release - intentional introduction as a commodity for release (game animals, bio-control agents and landscape plants).
- Escape - intentional introduction as a commodity, but unintentionally escapes (gardens, fur farms, aquaculture and zoos).
- Contaminant - unintentional introduction with a specific commodity (parasites, pest, and commensals of traded plants or animals),
- Stowaway - unintentional introduction attached to, or within, a transport vector (hull fouling, ballast water, soil/sediment organism).
- Corridor - unintentional introduction via infrastructure linking previously unconnected regions (for example canals and inter-basin transfer schemes).
- Unaided - unintentional introduction through natural dispersal of introduced species across political borders.

The majority of freshwater fish introductions have been intentionally released into new systems (Stohlgren *et al.* 2006). Vectors are required to move species along the pathways. The vectors for the introduction of non-native freshwater fishes have historically included nature conservation authorities (or government agencies that develop freshwater fisheries), recreational and subsistence anglers, the aquaculture industry, the ornamental fish trade, canals and inter-basin transfer schemes (Richardson *et al*. 2004), ballast water transfers (MacIsaac *et al.* 2001, Kolar and Lodge 2002), natural dispersion of species introduced into neighbouring countries, and bio-control programmes. The vectors operate along a combination of pathways (Table 2.1). Although the vectors and pathways for freshwater fish introductions operate globally, the following discussion focuses on South Africa, supplemented by selected examples from elsewhere.

2.3.1 Ballast Water Introductions

Ballast water has been one of the major vectors for the introduction of Ponto-Caspian fishes into the Laurentian Great Lakes in North America (Mills *et al.* 1993, MacIsaac *et al.* 2001, MacIsaac *et al.* 2002, Grigorovich *et al.* 2003). Ruffe *Gymnocephalus cernuus* (Linnaeus, 1758), tubenose goby *Proterorhinus marmoratus* (Pallas, 1814) and round goby *Neogobius melanostomus* (Pallas, 1814), zebra and quagga mussels (*Dresina* sp) and aquatic invertebrates from the Caspian Sea have caused major changes to the Great Lakes faunal assemblages (MacIsaac *et al.* 2001, Kolar and Lodge 2002). Ballast water introductions operate through the pathway of stowaways and threaten harbours situated on estuaries and navigable rivers. None of South Africa's rivers are navigable by ocean going vessels, but two harbours, Richards Bay and East London, are situated on estuaries and are therefore at risk to ballast water transfers. To date, no freshwater fish introductions attributable to ballast water releases have been recorded at either Richards Bay or East London harbours. Marine species in other taxonomic groups attributed to ballast water releases or hull fouling have been recorded at Richards Bay (Mead *et al.* 2011b, Mead *et al.* 2011a). Ballast water is not considered an important vector for freshwater fish introductions is South Africa

2.3.2 Natural Dispersal from Neighbouring Countries or Provinces

Non-native fishes introduced into neighbouring countries or provinces could disperse unaided via shared waterways. South Africa shares river systems with Namibia (Orange-Vaal River), Botswana (Orange-Vaal and Limpopo Rivers), Zimbabwe (Limpopo River), Mozambique (east flowing rivers from Limpopo to Kosi Rivers), Swaziland (headwaters of east flowing rivers) and Lesotho (headwaters of Orange River and many east coast rivers). The majority of South Africa's neighbouring countries are severely water-stressed (Namibia, Botswana and southern Zimbabwe). Only Mozambique poses a significant risk of introductions of freshwater fishes to South Africa through this vector.

2.3.3 Inter-Basin Transfer Schemes and Canals

Man-made inter-catchment connections represent a major vector for native and non-native fishes and aquatic invertebrates to be transferred to new catchments (Cambray and Jubb 1977, Bruton and van As 1986, Lintermans 2004, Copp *et al.* 2005a, Panov *et al.* 2009). The connection of catchments through canals, locks, and inter-basin transfers creates corridors through which species can move, or be moved, to adjacent catchments, or to areas of catchments formerly protected by impassable barriers such as major waterfalls. Canals and locks have resulted in freshwater fish transfers in Europe (Fruget *et al.* 1998, Panov *et al.* 2009) and the Laurentian Great Lakes (Ricciardi 2006), including the invasion of the Great Lakes by the sea lamprey *Petromyzon marinus* Linnaeus, 1758 (Ricciardi and Rasmussen 1998). Almost all major river catchments in South Africa are connected to adjacent catchments through tunnels, canals or pipes, and in some cases, such as the Tugela-Vaal and the Orange-Great Fish River transfer schemes, entirely different aquatic ecoregions have been connected (Cambray and Jubb 1977, Bruton and van As 1986). Inter-basin transfer schemes result in the creation of corridors for the dispersal of fish, and other aquatic organisms, to the recipient systems.

2.3.4 Government Agencies

In many countries the spread of non-native freshwater fishes has been performed by governmental agencies (Lintermans 2004, Marchetti *et al.* 2004b, c, Copp *et al.* 2005a, Iriarte *et al.* 2005). The introduction of trout *Salmo trutta* Linnaeus 1758 and *Oncorhynchus mykiss* (Walbaum, 1792) into South Africa was primarily to provide European settlers with familiar freshwater angling species (de Moor and Bruton 1988). The establishment of trout and warm-water fisheries was driven by the Department of Inland Fisheries in the early 20th century (Coke 1988, Skelton 2000). Fodder fish, such as bluegill sunfish *Lepomis macrochirus* Rafinesque, 1819 and banded tilapia *Tilapia sparrmani* A. Smith, 1840 were also introduced to provide food for angling species after they had consumed the native fishes (Harrison 1950, 1951a, b, 1952a, b). Historically, nature conservation authorities in South Africa have introduced numerous non-native freshwater fish in order to develop inland fisheries (Coke 1988, Richardson *et al.* 2004) and native fish outside their natural range as conservation measures (Impson and Tharme 1998, Tharme and Anderson 1999), although these practices have largely ceased (Richardson *et al.* 2004).

2.3.5 Ornamental Fish Trade

Ornamental fish enter natural waterways through the dumping of unwanted organisms, escape from garden ponds or breeding farms (e.g. during storms), and the ritualistic release of species during religious practices (Severinghaus and Chi 1999, Padilla and Williams 2004, Copp *et al.* 2005d, Duggan *et al.* 2006). Healthy ornamental fish are most commonly released when owners tire of them, or the fish become too large, aggressive, or prolific for their aquaria (Duggan *et al.* 2006, Gertzen *et al.* 2008). Ornamental fish releases are positively related to human population density, the ornamental trade (density of pet shops), and human access routes (Copp *et al.* 2005d, Copp *et al.* 2010).

The extent of ornamental fish releases is frequently underestimated (Welcomme 1992). Forty of about 100 species of ornamental fishes frequently introduced into North American waters have established populations (Rixon *et al.* 2005). More than 50% of non-native fish introductions in Australia are ornamental fish (Lintermans 2004).Aquarists, between 5 and 6.5%, have admitted to having released fish (Gertzen *et al.* 2008, Strecker *et al.* 2011). Further, many species are available for purchase through the Internet, even though they are regionally banned (Padilla and Williams 2004). In Australia, smuggling of ornamental fish is estimated to be equivalent to about 5–10% of the legal imports to Australia and is increasing (Lintermans 2004). For South Africa, the aquarium trade poses a medium risk of future introductions of freshwater fish in South Africa (Richardson *et al.* 2004), but more so in the sub-tropical coastal areas of Kwazulu-Natal where several species including guppy *Poecilia reticulata* Peters, 1859 and swordtail *Xiphophorus helleri* (Heckel, 1848).

2.3.6 Aquaculture

Aquaculture is the leading vector of aquatic non-native species worldwide (Casal 2006), with more than 50% non-native species having been intentionally introduced for aquaculture (Casal 2006, Cook *et al.* 2008, Gozlan 2008). Aquaculture has led to introduction of seaweeds, fish, invertebrates, fish parasites, and pathogens, but the rapid expansion of this sector could result in the introduction of many more species (Naylor *et al.* 2001, De Silva *et al.* 2009). The proportion of non-native species in Asian freshwater finfish production has increased from about 20% prior to 1995 to more than 35% in 2005 (De Silva *et al.* 2009). Half of the freshwater fish species introduced for aquaculture have established in the wild, while 35% have failed to establish (Casal 2006). The fate of the remaining 15% is not known (Casal 2006).

The release of fish to natural water bodies by the aquaculture industry can occur through legal and illegal release of non-native species and escape of fish from culture facilities. There is also a risk that imported fish stocks are contaminated by undesirable fish species. The spread of the stone maroko *Pseudorasbora parva* (Temminck and Schlegel, 1846) and gibel carp *Carassius gibelio* (Bloch, 1782) across Europe has been linked to contamination of hatchery stocks (Copp *et al.* 2005a, Gozlan *et al.* 2010a). The aquaculture industry is a medium-risk pathway for future introductions of freshwater fishes in South Africa (Richardson *et al.* 2004).

2.3.7 Biological Control

The release of fish to natural water bodies for bio-control can occur through legal and illegal release of non-native species. There is a risk of fish stocks released for bio-control being contaminated with undesirable species. The release of non-native fish to control mosquitoes has had devastating impacts on native fish assemblages (García-Berthou 1999, Lawler *et al.* 1999, Ayala *et al.* 2007, Laha and Mattingly 2007, Alemandi and Jenkins 2008, Pyke 2008) and has ceased in many countries, although the illegal spread of these species continues (Morgan *et al.* 2004, Marr *et al.* 2010). Non-native fish are frequently introduced for the control of aquatic vegetation. In South Africa, grass carp has been widely introduced into farm dams to control aquatic vegetation (de Moor and Bruton 1988). It was initially believed that South Africa's rivers were not suitable for grass carp (de Moor and Bruton 1988) but they have established recruiting populations in the Orange-Vaal and Pongola systems (Skelton 2001). Today only certified triploid grass may be introduced. Although these fish are sterile, they are long-lived and grow to be very large (over 1m) and escapees into natural watercourses might have an impact on both aquatic vegetation and invertebrates (Pípalová 2006).

2.3.8 Fisheries – Recreational and Subsistence

Non-native fish have been widely introduced to establish recreational fisheries (Cambray 2003b, Johnson *et al.* 2009, Gozlan *et al.* 2010b). The release of fish to natural water bodies for angling occurs through legal and illegal release of non-native species. There is a risk that fish consignments legally released for angling are contaminated by undesirable species. The illegal movement of fish between water bodies can seriously compromise both recreational fisheries and conservation programmes (Lintermans 2004, Impson 2006, Johnson *et al.* 2009, Gozlan *et al.* 2010b). Once introduced, non-native species are subsequently illegally spread by anglers to neighbouring catchments (García-Berthou *et al.* 2005, Clavero and García-Berthou 2006). Despite strict laws prohibiting introductions, spread of these species continues in defiance of the law (Lintermans 2004, McDowall 2004). Unauthorized introductions in seven U.S. regions accounted for 90% of new introductions between 1981–1999, but only 15–43% prior to 1981 (Rahel 2004).

The US$1 billion bait-fish industry in North America has resulted in bait-bucket transfers becoming a major introduction vector (Litvak and Mandrak 1993, Ludwig and Leitch 1996). Anglers discarding bait fish where they are fishing, or into local dams and ponds to provide bait for future trips (Lintermans 2004). Because the practice is illegal, it is difficult to quantify (Lintermans 2004). The use of live bait fish in South Africa has not been quantified.

In South Africa, most recreational anglers adhere to the catch-and-release ethic, whereas subsistence anglers eat the fish they catch. Many recreational anglers belong to clubs or societies that adhere to strong conservation ethics that discourage the spread of non-native fish. Of greater concern are subsistence anglers who move large fish, such as sharptooth catfish, and land-owners who stock their dams to provide fishing for their friends and family. Recreational and subsistence anglers pose a high risk for future introductions of non-native fishes in South Africa (Richardson *et al.* 2004).

2.4 Landscape Factors Threatening Native Fish Populations

The extirpation of native freshwater fishes has been attributed to an array of factors, including physical habitat alteration (including flow alterations), introduction of non-native species, changes in water quality, and pollution (pesticides, excessive nutrients) (Ross *et al.* 2001). Environmental disturbances have occurred simultaneously with non-native fish introductions (Light and Marchetti 2007) and some authors (Gurevitch and Padilla 2004, Didham *et al.* 2005, Gozlan 2008) have argued that the distinction between the effects of these two factors (habitat loss and non-native species introduction) have not been adequately separated in the scientific literature. Are non-native species "drivers" of the decline of native species, or are non-native species "passengers" riding on the impacts of human modification of natural ecosystems and native species declines a result of habitat loss? It is therefore important for studies of non-native fish introductions to rigorously consider all the factors that could contribute to the decline in native fish assemblages before implicating non-native species in native species decline (Gurevitch and Padilla 2004). Distinguishing whether non-native species are the drivers of native species declines, or passengers on habitat degradation, is not trivial (Moyle 1976, 1999, Moyle and Marchetti 2006). Therefore land-use change, in-stream habitat availability, and changes in water quality (pesticides, excessive nutrients) should be incorporated in studies of the impacts of non-native species (Gurevitch and Padilla 2004).

2.4.1 Land use change

Conversion of a catchment from one land use to another may influence stream ecosystems via changes to nutrient loading, solar energy flux, hydrology, sediment inputs, organic matter inputs and decomposition rates (Townsend *et al.* 2004). Changes in land use are known to influence water quality, hydrology, sediment transport and riparian zones, resulting in declines in native fish populations (Richards *et al.* 1996, Johnson and Gage 1997, Johnson *et al.* 1997, Ross *et al.* 2001, Argent *et al.* 2003, Argent and Carline 2004). Streams draining agricultural lands support fewer sensitive insect and fish taxa than streams draining forested catchments (Allan 2004b). Streams in highly agricultural landscapes tend to have poor habitat quality, reflected in declines in habitat indexes and bank stability, as well as greater deposition of sediments on and within the streambed (Allan 2004b). Agricultural land use degrades streams by increasing non-point inputs of pollutants, impacting riparian and stream channel habitat, and altering flows (Allan 2004a, b), increasing inputs of sediments, nutrients, and pesticides (Allan 2004b). Similarly, deforestation, urbanization and river channelization result in similar impacts on river systems (Kamdem Toham and Teugels 1999, Wang *et al.* 2000, Wang *et al.* 2001, Bojsen and Barriga 2002, Morgan and Cushman 2005). Appropriate management programmes can improve stream conditions within the constraints of the existing land use and agricultural activities (Allan 2004a).

2.4.2 Habitat Availability

The association between fish and their habitat is complex (Collares-Pereira and Cowx 2004) and translating the habitat requirements of individuals to population-level effects is challenging (Rosenfeld 2003). Understanding the relationship between instream habitat and the health of fish populations is the first step towards developing efficient and effective conservation or restoration strategies (Rabeni and Sowa 1996, Bond and Lake 2003, Rosenfeld 2003). Water depth and velocity, in conjunction with habitat heterogeneity, are good predictors of species richness (Marsh-Matthews and Matthews 2000). Habitat connectivity is critical for successful fish dispersal. Connectivity between suitable habitats within river systems has been severely restricted as a result of dam building, water diversion and introduction of non-native species (Fagan *et al.* 2002, Leprieur *et al.* 2008a). Fagan *et al.* (2002) found that the level of fragmentation of fish populations was consistently associated with an elevated risk of species extinction.

2.4.3 Water Quality

The physico-chemical characteristics of river water are important in determining the survival of fish and other aquatic species, particularly pH (Bhatt *et al.* 2002, Sutela *et al.* 2010), dissolved oxygen saturation (Eklöv *et al.* 1999), and temperature (Tonn 1990, Bhatt *et al.* 2002). Temperature influences many physiological processes in fish, including spawning, development, growth, and metabolism (Tonn 1990). Temperature affects oxygen dissolution and saturation in water, and haemoglobin uptake and delivery of oxygen from the gills to respiring tissues (Bhatt *et al.* 2002, Dallas and Day 2004). Shifts in the average temperature or temperature ranges alter aquatic communities (Tonn 1990), including increasing the distribution range of invasive species towards the poles and upstream in river systems (Rahel and Olden 2008). Changes in water quality or temperature during the breeding season may account for large-scale fish embryo mortality (Cambray and Stuart 1985). Changes in pH influence the toxicity of many dissolved compounds (Dallas and Day 2004), particularly aluminium. Increased nutrient inputs increase primary productivity (Dallas and Day 2004), but may decrease oxygen concentrations during the night (Eklöv *et al.* 1999). Ammonia is toxic to many fish (Dallas and Day 2004) and can decrease oxygen concentrations during oxidation to nitrate (Eklöv *et al.* 1999). Suspended solids reduce light penetration, photosynthetic growth, and visibility (Dallas and Day 2004). Suspended solids also smother bethic habitat impacting upon aquatic invertebrates and reducing spawning success (Eklöv *et al.* 1999, Dallas and Day 2004, Sutela *et al.* 2010). Organic pollution increases nutrient availability and chemical oxygen demand (Dallas and Day 2004). See Table 2.2 for a summary of the impact of selected water quality variables on aquatic ecosystems.

2.4.4 Flow Modifications

Water quantity impacts fish by disrupting the cues for reproduction or migration in conjunction with knock-on effects on many critical physicochemical characteristics, such as water temperature, channel geomorphology and physical habitat diversity (including water temperature, oxygen content, water chemistry, and substrate particle sizes). Flow is considered a "master variable" limiting the distribution and abundance of riverine species and regulating the ecological integrity of river systems (Richter *et al.* 1996, Poff *et al.* 1997, Bunn and Arthington 2002). In streams of similar size, physical habitat characteristics (water depth, current velocity, substrate) are important factors influencing fish community composition (Gorman and Karr 1978, Bain *et al.* 1988). The maintenance or restoring of natural flow patterns (timing and magnitude) has been proposed as a means of enhancing native fish recruitment and controlling the populations of non-native fishes, particularly fecund, rapidly growing, small-bodied species (Propst and Gido 2004). However, a natural flow regime alone is unlikely to ensure persistence of native fish assemblages (Propst *et al.* 2008).

2.4.5 Pesticides

Agricultural insecticide and herbicide runoff is likely responsible for some of the association between agricultural land use and stream biota (Allan 2004b). Non-point-source agricultural pollution, mainly pesticide contamination of runoff or spray drift, is regarded as the greatest threat to aquatic systems in rural areas (Thiere and Schulz 2004). Organophosphorus, carbamate, and synthetic pyrethroid compounds are the most commonly used insecticides (Fulton and Key 2001). Because of their rapid degradation in the environment, organophosphorus pesticides have become one of the most widely-used insecticides. However, these compounds generally lack target specificity and have high acute toxicity toward many non-target vertebrate and invertebrate species (Fulton and Key 2001). The effects of pesticides include direct toxic effects, delayed effects (where sub-lethal doses may alter the behaviour of an organism, or may accumulate within the organism), reductions in food supply, reduction in habitat (e.g. the destruction of reed beds), and altering competitive relationships among aquatic organisms (Moore 1967).

In agriculture, organophosphorus pesticides are used to control insects on fruits, vegetables, grain crops, and stored seeds (Fulton and Key 2001). In agricultural landscapes, stream and river faunas are most sensitive to pesticide impacts and should be protected by a buffer zone in areas where pesticides are applied in close proximity to these ecosystems (Biggs *et al.* 2007). Organophosphorus insecticides inhibit cholinesterase enzymes in both vertebrate and invertebrate organisms. These enzymes are responsible for the removal of the neurotransmitter acetylcholine from the synaptic cleft through hydrolysis (Fulton and Key 2001). When choline esterases are inhibited, an accumulation of acetylcholine occurs, interfering with the normal nervous system function. Once bound, organophosphorus compounds are considered irreversible inhibitors, as recovery usually depends on new enzyme synthesis (Fulton and Key 2001). Most organophosphorus insecticides degrade rapidly in the environment, falling to below detection levels within hours to days, whereas acetylcholine esterase inhibition persists for much longer in many species, days to weeks (Fulton and Key 2001).

2.5 Biotic Homogenisation

Biotic homogenization describes the process whereby geographically-restricted species are replaced by widespread species, ultimately reducing regional biodiversity (McKinney and Lockwood 1999). Environmental modification, species introductions and extirpations of endemic species have been identified as the main drivers of biotic homogenization (Rahel 2002). Increasing global transport promotes the spread of non-native species (McKinney and Lockwood 1999), increasing the risk that non-native species will be introduced to, and establish self-sustaining populations in, natural systems. Environmental modification (such as altered flow regimes, increased silt loading, clearing of riparian vegetation) promotes the loss of range-limited specialist species which, in turn, are replaced by widespread generalist native or non-native species (Scott and Helfman 2001). Although non-native species sometimes thrive in disturbed environments, many invade relatively undisturbed natural areas (McKinney and Lockwood 1999, Rahel 2002). The prediction that geographically-restricted native species with sensitive requirements (stenotopes) will continue to be extirpated while widespread, tolerant species (eurytopes) will spread and become increasingly dominant (Brown 1989, Scott and Helfman 2001), will increase the degree of homogenization at continental scales (McKinney and Lockwood 1999).

2.5.1 Defining Biotic Homogenization

Biotic homogenization can be defined as '*the process by which species similarity across space increases over time due to species invasions and extinctions*' (Olden and Rooney 2006). Most published studies evaluate biotic homogenization at species level (Olden *et al.* 2004, Olden and Rooney 2006). Olden and Rooney (2006) argue that imposing a narrow phylogenetically based definition does not accurately reflect the multidimensional nature of biotic homogenization, recommending that the broader, overarching ecological processes that result in formerly unique biota losing biological distinctiveness be included in the definition of biotic homogenization. The terms Taxonomic, Genetic and Functional Homogenization have been proposed to describe the homogenization at the species, genetic and functional levels (Olden *et al.* 2004, Olden and Rooney 2006).

2.5.2 Ecological and Evolutionary Impacts of Biotic Homogenization

The ecological consequences of biotic homogenization are reflected at five levels: effects on individuals (demographic rates such as mortality and growth), genetic effects (hybridization), population dynamic effects (abundance, population growth), community effects (species richness, diversity, trophic structure), and effects on ecosystem processes (nutrient availability, primary productivity, energy flow) (Parker *et al.* 1999). Biotic homogenization may simply decrease the abundance of all members of a native community, or may have differential impacts on different species, resulting in a fundamental change in community composition (e.g. native species extirpations, impacts on lower and higher trophic levels) and function (e.g. trophic links, energy flow). The greatest impacts often occur when a non-native species performs an entirely novel function in the recipient community (Parker *et al.* 1999), or when predators and prey do not share an evolutionary history (Rodda *et al.* 1997).

Biotic homogenization results in species compositions and relative abundances that have not previously occurred within any biome; termed 'novel' or 'emerging ecosystems' (Milton 2003, Hobbs *et al.* 2006). Speciation is a result of numerous environmental and evolutionary processes acting on a biological template (Olden *et al.* 2004) and changing the ecological processes of the biological template will have evolutionary consequences that are difficult to predict (Olden 2006). The evolutionary impacts of species introductions include the altering of evolutionary pathways of native species by competitive exclusion, niche displacement, hybridization, introgression, predation, and ultimately extinction (Mooney and Cleland 2001). Further, non-native species have been found to evolve in response to their interactions with native species and the new abiotic environment though flexibility in behaviour and mutualistic interactions in their new environment (Mooney and Cleland 2001). The source of future biodiversity has potentially been compromised by the combining of diverging evolutionary lineages through hybridization and introgression (Olden *et al.* 2004) and the loss of evolutionary potential through the extinction of range-limited species. Biotic homogenization could result in the origin and diversification of new species as non-native species evolve in new environments, or hybridization takes place (Olden *et al.* 2004).

2.5.3 Taxonomic Homogenization

The majority of the studies published on biotic homogenization have investigated taxonomic homogenization based on historical and current species lists, mostly based on presence/absence data. Taxonomic homogenization studies have been published for a number of taxonomic groups including plants (Kühn and Klotz 2006, McKinney 2006, Schwartz *et al.* 2006, Vellend *et al.* 2007, Castro and Jaksic 2008a, Castro and Jaksic 2008b, Lambdon *et al.* 2008, Abelleira Martínez 2010, Qian and Guo 2010), birds (Lockwood *et al.* 2000, Lockwood and McKinney 2001, Cassey *et al.* 2007, La Sorte and McKinney 2007, Cassey *et al.* 2008, van Rensburg *et al.* 2009), freshwater fish (Radomski and Goeman 1995, McKinney and Lockwood 1999, Rahel 2000, Duncan and Lockwood 2001, Marchetti *et al.* 2001, Rahel

2002, Marchetti *et al.* 2004a, Taylor 2004, Clavero and García-Berthou 2006, Eberle and Channell 2006, Marchetti *et al.* 2006, Rahel 2007, Leprieur *et al.* 2008b, Olden *et al.* 2008, Taylor 2010), and large ungulates (Spear and Chown 2008).

The introduction of fish species differ across temporal and spatial scales have random biases in human selection and propagule pressure, resulting in each catchment within a region receiving a unique set of species (Marchetti *et al.* 2001, 2006, Leprieur *et al.* 2008b, Olden *et al.* 2008) Establishment rates differ between regions due to the environmental and biological attributes of the recipient region (Leprieur *et al.* 2008b, Olden *et al.* 2008). Many catchments retain of a limited numbers of native or endemic species, thereby retaining their uniqueness (Marchetti *et al.* 2006). The dispersal abilities and environmental tolerances of introduced species are usually limited and many non-native species have difficulties in spreading autonomously beyond the catchment of their introduction (Marchetti *et al.* 2001, 2006, Leprieur *et al.* 2008b). However, neighbouring catchments show similar rates of homogenization/differentiation in comparison to more distant catchments (Leprieur *et al.* 2008b), with non-native species likely to be moved to neighbouring catchments over time (Clavero and García-Berthou 2006). Species translocated within a region homogenize fish communities, whereas non-native species differentiate fish communities (Leprieur *et al.* 2008b). Urbanization is an important driver of freshwater fish homogenization (Marchetti *et al.* 2001, 2006, Olden *et al.* 2008).

Patterns of species introductions are dynamic and vary over time in response to natural and human-related factors (Olden *et al.* 2008). At smaller spatial scales, the introduction of different species is more likely, contributing to greater differences in compositional similarity (Marchetti *et al.* 2001, 2006, Olden *et al.* 2008). Further homogenization is likely to occur and threatened species are extirpated and additional non-native species establish self-sustaining populations (Olden *et al.* 2008). The rate of homogenization is predicted to accelerate under the pressure of urban and agricultural expansion and global climate change, resulting in species extirpations and a few species being found almost everywhere (Olden *et al.* 2008).

2.5.4 Functional Homogenization

The impact of human activities on ecosystems functioning is resulting in a greater interest in the functional aspect of biotic homogenization, although very few studies of functional homogenization, all of bird communities, have been published (Devictor *et al.* 2007, 2008, Clavero *et al.* 2009, Clavero and Brotons 2010, Baiser and Lockwood 2011). As with functional diversity studies, the definition of the study objective is important in determining the functional traits used (Petchey and Gaston 2006). The majority of published studies have focussed on the composition of generalists versus specialists in the bird communities (Devictor *et al.* 2007, 2008, Clavero *et al.* 2009, Clavero and Brotons 2010). While this is one of the important findings of most biotic homogenization studies – the replacement of range-restricted specialist species by wide-spread generalist species, functional homogenization studies should incorporate aspects of ecosystem functioning, furthering the knowledge of communities and ecosystems based on what organisms do (Petchey and Gaston 2006), becoming a tool to predict the consequences of biotic changes resulting from human activities (Chapin *et al.* 1998). It is therefore important to know how a change in functional diversity influences ecosystem processes, the dynamics of ecosystems, and their stability (Petchey and Gaston 2006).

The selection of functional traits to represent the changes in the ecosystem under investigation requires knowledge about the particular organisms' interactions with their environment, with each other, and how these traits vary across environmental gradients (Petchey and Gaston 2006). Many studies investigating the functional traits of estuarine and freshwater fish communities have used morphometric measures (such as standard body length to body depth ratio, pectoral fin aspect ratio, caudal fin aspect ratio, eye diameter, mouth protrusion, oral gape, height of gill raker, gut length) to determine the resource use of the fish species (Winston 1995, Dumay *et al.* 2004, Peres-Neto 2004, Mouillot *et al.* 2005, Mason *et al.* 2007, Mouillot *et al.* 2007, Villéger *et al.* 2008, 2010). The ecological guild concept (Simberloff and Dayan 1991) has been used to classify freshwater fish by reproductive strategies (Balon 1975, 1981), locomotion morphology (Webb 1984, Webb and Weihs 1986, Webb 1988), body morphology (Gatz 1979a, b, 1981), position in the river and migration (Welcomme *et al.* 2006), trophic level and feeding mechanisms (Matthews 1998, Aarts and Nienhuis 2003), life history strategies (Winemiller and Rose 1992), flow and depth preferences (Aadland 1993), substrate preferences (Goldstein and Meador 2004), or combinations of these (Poff 1997, Angermeier and Winston 1999, Goldstein and Simon 1999, Goldstein and Meador 2004, 2005, Olden *et al.* 2006, Blanck and Lamouroux 2007, Blanck *et al.* 2007, Olden and Kennard 2010). Fish functional groups have been proposed as indices of biotic integrity (Karr 1981, 1987, 1991, Welcomme *et al.* 2006). Many of the classification systems and functional groups in literature require measurements or data not freely available for the majority of species, especially species from regions where fish faunas have been poorly studied (e.g. Africa, Asia, South America), or where taxonomies are in a state of flux. Macroecological studies, in most cases, have to rely on data repositories such as FishBase (Froese and Pauly 2010) to compile data for these species.

This leads to the question: *What is a suitable set of functional traits that can be used to summarise the function of fish species in freshwater ecosystems?* Firstly, a measure of fish function should summarise the species role in energy and nutrient cycling by identifying the position in the food web (trophic level), level of energy/nutrient accumulation/turn-over in the individuals/species (longevity or body size), and role in the transfer of energy/nutrients to other systems (migration). Secondly, a measure of the species specialization of habitat preferences (flow, depth, substrate, turbidity), distribution (size of native range), sensitivity to change (salinity, temperature, oxygen levels), feeding style *sensu* Matthews (1998), and reproductive strategy (parental care, reproductive guild *sensu* Balon (1975, 1981)) is required to communicate the unique function that the species provides.

Figures

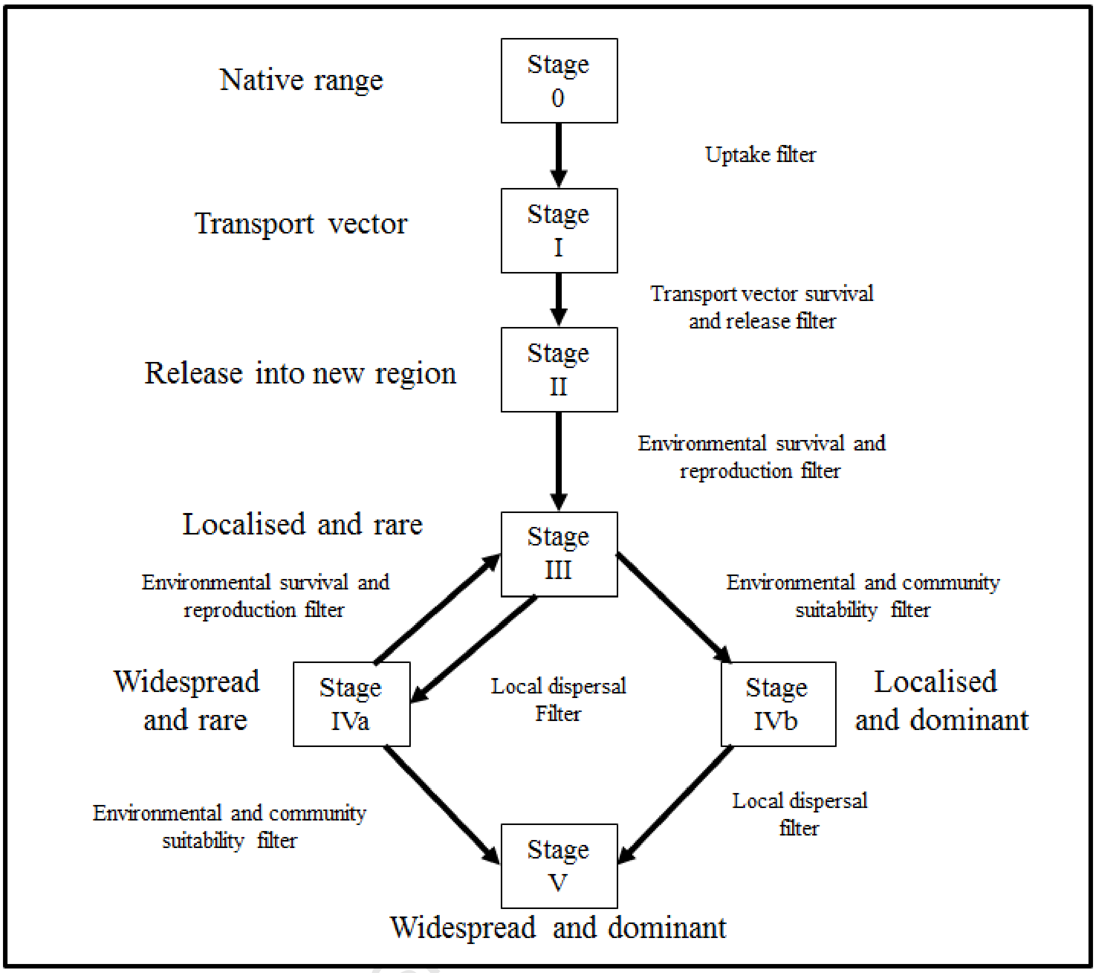


Figure 2.1: Framework for biological invasion. Potential invaders begin as propagules in the donor region (Stage 0) and are subjected to a series of filters that may preclude progression to subsequent stages. Stages III through V are divided based on non-native species abundance and distribution. A non-native species may be localized and numerically rare (Stage III), widespread but rare (Stage IVa), localized but dominant (Stage IVb) or widespread and dominant (Stage V). (Adapted from Colautti and MacIsaac (2004))

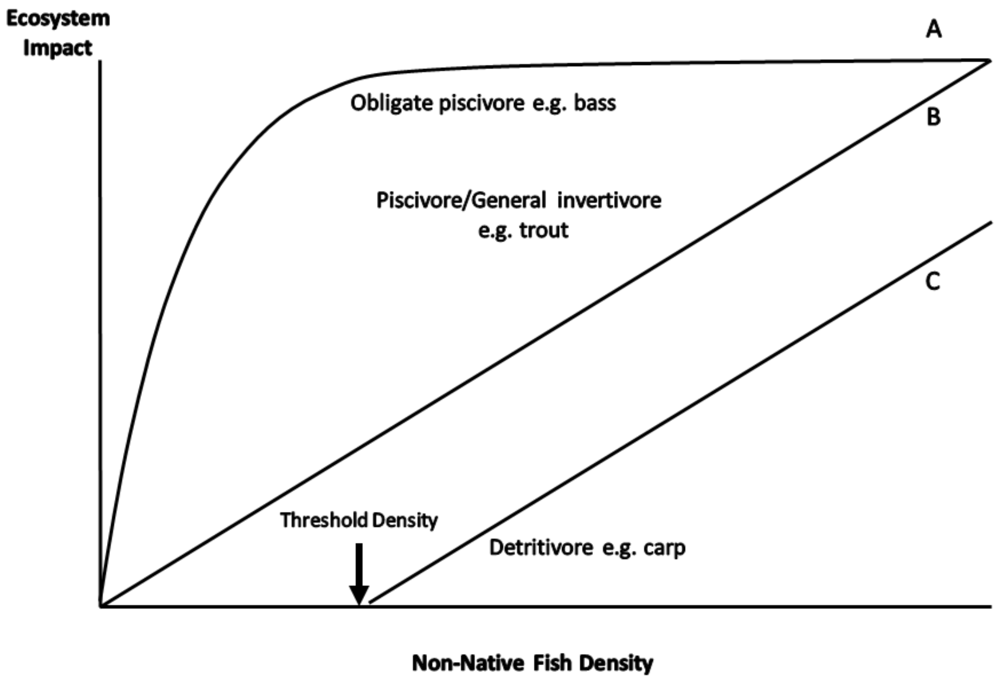


Figure 2.2: Hypothetical model reflecting the impact of a non-native species on the recipient ecosystem with increasing population density. Line A represents a non-native species that has an immediate drastic impact on the recipient assemblage, even at low population densities. Line B represents a species having a linear impact with increasing population density, whereas Line C represents a species that requires a threshold population density before its impact becomes evident in the recipient environment followed by a linear increase in impact with increasing population density. (Adapted from Bomford and Tilzey (1997))

Tables

Table2.1: Pathways used by the respective vectors for the introduction and/or spread of non-native freshwater fishes

		Pathways						
		Release		Escape	Contaminant	Corridors	Stowaway	Unaided
		Legal	Illegal					
Vectors	Conservation authorities	X			X			
	Recreational anglers	X	X		X			
	Subsistence anglers		X		X			
	Bio-control	X	X	X				
	Aquaculture industry	X	X	X	X			
	Ornamental fish trade		X	X	X			
	Inter-basin transfer schemes					X		
	International shipping						X	
	Neighbouring countries							X

Table2.2: The effect of selected water quality variables on aquatic ecosystems (adapted from Dallas and Day (2004)).

Water Quality Variables	**Major Effects**
	Physical Factors
Temperature	Determines metabolic rate Determines availability of nutrients and toxins Determines oxygen saturation level Changes provide cues for breeding, migration, etc.
Turbidity and suspended solids	Turbidity determines the degree of light penetration, photosynthesis, and visibility Suspended solids reduce light penetration, smother and clog surfaces (e.g. gills) and adsorb nutrients, toxins, etc.
	Chemical Factors
pH	Determines ionic balances Affects chemical species solubility Affects gill functioning
Conductivity, salinity, TDS, individual ions	Affects osmotic, ionic and water balance
Dissolved oxygen	Required for aerobic respiration
Organic enrichment	Reduces oxygen concentration Increases nutrient levels
Nutrient enrichment	Not toxic per se: cause eutrophication and thus impacts community structure Can increase daytime oxygen concentration and decrease night-time oxygen concentration Eutrophication can reduce light penetration and visibility
Biocides	Usually target specific groups (e.g. molluscs, insects, plants) and thus alter community structure Can impact non-target organisms
Trace metals	Many usually at low concentrations Some mutagenic, tetragenic, carcinogenic Some metabolic inhibitors

CHAPTER 3. THE CAPE FLORISTIC REGION AND ITS FRESHWATER FISH

The Cape Floristic Region is the primary study area for this thesis. This chapter seeks to introduce the study area and its freshwater fish fauna, present a historical account of the introduction of non-native fishes in the region, discuss the development of conservation initiatives, and outline current and proposed measures for the conservation of native freshwater fish and the management of non-native fish.

Although the majority of this chapter constitutes a review of available literature, original calculations have been performed using Spatial Analyst extension packages for Geographical Information System (GIS) programmes. The figures cited in sections 3.1.6, 3.1.7, and 3.3.2 are the results of original calculations.

3.1 The Cape Floristic Region

The Cape Floristic Region is a 87,892 km^2 Mediterranean-climate region at the south-western tip of Africa. It is recognised for its diversity and endemism of plants and animals (Cowling and Pressey 2003), having the highest recorded plant species densities of any temperate or tropical region of equivalent size (Cowling and Holmes 1992). Sixty-eight per cent of flowering plant species, 19.5% of genera, and six plant families are endemic to the region (Cowling and Holmes 1992). The Cape Floristic Region contains the Cape Floral Kingdom, the smallest plant kingdom and the only one found within a single country (Wishart and Day 2002, Younge and Fowkes 2003). Two biomes are included in the area: the Fynbos and Succulent Karoo Biomes (Cowling and Holmes 1992); both are recognised biodiversity hotspots (Myers *et al.* 2000). Although the Cape Floristic Region is recognised for its plant biodiversity, it is also a centre of endemism for freshwater fauna, including aquatic invertebrates, amphibians and fish (Skelton *et al.* 1995, Wishart and Day 2002, Younge and Fowkes 2003). The degree of endemism of the aquatic invertebrates, amphibians and fishes is proportionally higher than that of the plant diversity, even though the numbers of species in each are significantly lower (Wishart and Day 2002), as indeed is the surface area occupied by freshwater ecosystems.

3.1.1 Terrestrial Habitat Transformation and Protected Areas

As with other Mediterranean-climate regions, the Cape Floristic Region has been modified by human activities (Groves and Di Castri 1991). Major threats to biodiversity include a rapidly expanding human population, increasing demands for fresh water (Impson *et al.* 2002a), loss of habitat to agriculture, rapid and insensitive development of housing, overexploitation of marine resources and wild flowers, spread of non-native species, and inappropriate fire regimes (Younge and Fowkes 2003). The on-going loss of biodiversity is driven by a lack of capacity within, and poor co-ordination between, natural resources management and conservation bodies, lack of awareness of the importance of biodiversity in government and amongst the general public, and a short-term focus on meeting the needs of a large, previously disadvantaged populace (Gelderblom *et al.* 2003, Lochner *et al.* 2003, Younge and Fowkes 2003).

An estimated 30% of the surface area has been transformed to agriculture (< 30% including cultivated land, forestry and pasture lands), alien woody plants (1.6% with > 20% alien infestation), and urban (1.6%) land-use (Rouget *et al.* 2003b). It is predicted that at least 30% of the remaining natural vegetation might be transformed within the next 20 years (Rouget *et al.* 2003b). Statutory and non-statutory protected areas constitute about 22% of the region (Rouget *et al.* 2003a), which is considerably higher than the global average of 5% for Mediterranean-climate regions (Underwood *et al.* 2009).

Following the political transition of South Africa in 1994, opportunities to access international funding for conservation action arose (Younge and Fowkes 2003). Funding was received from the Global Environmental Faculty for the Cape Peninsula Biodiversity Project. This was later expanded to incorporate the entire Cape Floristic Region under the Cape Action-plan for People and the Environment (CAPE) Project (Younge and Fowkes 2003). [See the special issue of *Biological Conservation* (2003) volume 112.]

3.1.2 Icthyological Provinces

Roberts (1975) divided the freshwater fish fauna of southern Africa into two ichthyological provinces, the Zambezian and the Southern Temperate (or Cape). The Zambezian Province stretches from the northern watershed of the Kunene-Okavango and Zambezi River catchment south to the Orange River in the west and the Bushmans River in the east (Skelton 1994). The Southern Temperate Province is entirely within South Africa, occurring in the coastal streams of the Cape Floristic Region, Eastern Cape and KwaZulu-Natal, the Orange River system and the Highveld-Middleveld reaches of the Limpopo, Inkomati and Pongola systems (Skelton

1994). The Cape Floristic Region has always been considered a distinct biogeographic province within the Southern Temperate Province (Skelton 2001).

Recently southern Africa was divided into 21 freshwater ecoregions based on freshwater fish distributions (Abell *et al.* 2008). In Abell's classification, the Cape Floristic Region falls entirely within the Cape Fold Ecoregion. Although the "Cape Fold Ecoregion" is the more appropriate term to use for the study area from a freshwater fish perspective, this term has only recently been proposed and most researchers are more familiar with the term "Cape Floristic Region". Therefore, the term "Cape Floristic Region" is used in this thesis.

3.1.3 Evolution of the Drainage Basins

The Cape Floristic Region is situated on a passive continental margin that developed during the break-up of Gondwana (McCarthy and Rubidge 2005). The mountains of the Cape Floristic Region (the Cape Fold Mountains and outliers such as the Cape Peninsula and Piketberg) have probably changed very little throughout the Cenozoic (Deacon *et al.* 1983). Globally, the sea-level has risen (marine transgression) and fallen (regression) as a result of plate tectonics and polar ice caps. Fluctuations in sea-level, ranging from > 200 m above the present level to 120 m below, have resulted in complete inundation of the coastal lowlands of the Cape Floristic Region to exposure of wide areas of the continental shelf, particularly on the Agulhas Plain (Dingle and Rogers 1972, Hendey 1983).

The drainage history of South Africa has been described by Dingle and Hendey (1984), Partridge and Maud (1987), and Goudie (2005). Skelton (1994) provides a summary of how the drainage history has resulted in the current distribution of freshwater fish in the Zambezian and the Southern Temperate Ichthyofaunal Provinces. The rivers of the Cape Floristic Region have been largely isolated from the river systems to the north, the exception being the Olifants-Doring system, the headwaters of which were joined to the headwaters of the Orange-Vaal system during the late to middle Cretaceous (McCarthy and Rubidge 2005). By the early Cenozoic, the Kalahari River draining southern Botswana and Namibia had captured the headwaters of the Orange-Vaal system, following uplift of the southern and eastern sub-continent margins (Goudie 2005). The Berg River was connected to the Orange-Olifants system during periods of lower sea-levels in the Miocene and Pliocene epochs of the late Tertiary (Hendey 1983). The other large rivers of the Cape Floristic Region, the Breede, Gouritz and Gamtoos Rivers, have never been linked to each other during periods of low sea-level (Hendey 1983). Smaller coastal rivers have been connected during periods of lower sea levels (Swartz *et al.* 2007), some to larger rivers such as the Breede and Gamtoos (Skelton

1984). [See Appendix A.1 for a review of the geological history and Appendix A.2 for a brief review of the drainage development of the Cape Floristic Region.]

The relative stability of the subcontinent has resulted in the rivers of the Cape Floristic Region being largely unchanged since the beginning of the Cenozoic (65 Mya), with many river systems likely to have been confined to their present valleys through the Cenozoic (Hendey 1983). River captures have occurred including the Berg capturing the headwaters of the Breede, the Breede capturing tributaries of the Gouritz, and distinct river captures in the Gamtoos system (Skelton 1984).

3.1.4 Rainfall and River Water Characteristics

The Cape Floristic Region has a typical Mediterranean type climate of cool, wet winters, grading eastwards along the south coast into spring and autumn rainfall peaks (Davies *et al.* 1993). Rain falls almost entirely in winter in the western part of the region, but becomes increasingly non-seasonal towards the east. In addition, orographic rainfall, driven by strong onshore winds over the mountainous coastal areas, results in increased runoff. The highest rainfall occurs in the mountains of the South-Western Cape (2500 mm/a). The mountain ranges of the south coast divide the cloudy coastal belt (rainfall 750 mm/a) from the dry Karoo interior with rainfall less than 400 mm/a (Davies *et al.* 1993).

The majority of the rivers of the Cape Floristic Region drain the coastal slope between the Cape Fold Mountains and the coast, while the inland areas are drained by three large river systems, the Olifants-Doring, Gouritz and Gamtoos Rivers (Figure 3.1). The rivers draining the Cape Fold Mountains are oligotrophic, have characteristic low pH and many have high humic concentrations derived from the nutrient poor soils and the tannin-rich fynbos vegetation (King *et al.* 1979, Lambrechts 1979, Raubenheimer and Day 1991, Davies *et al.* 1993). Generally the upper reaches of the rivers are steep, fast flowing mountain streams running through Mountain fynbos or Afromontane forest, whilst their lower reaches pass through land disturbed by human activities that affect the quality and quantity of water (King *et al.* 1979, Davies *et al.* 1993). The rivers are usually divided into those with clear, white, slightly acid waters, such as the Olifants, Berg, Eerste and Breede Rivers and those with dark, very acid waters, such as the Palmiet, Keurbooms and Southern Cape Rivers. Generally, the white rivers are longer with well-developed zones (mountain source, mountain stream, foothills, lower river and estuary) whilst the dark rivers change abruptly from mountain stream to estuary (King *et al.* 1979). The porous nature of the soils of the region results in most of the rainfall over mountainous areas percolating into streams, with standing waters being confined to seepage areas (King *et al.* 1979). Most rivers change markedly in their physico-chemical character after leaving the mountains. Generally, nutrients, turbidity and pH increase, while flow may be poor in the lower reaches, especially in summer in the western rivers, where the majority of the water is extracted for human use.

3.1.5 Ecological Status of the Rivers of the Cape Floristic Region

The Cape Floristic Region contains the most threatened river ecosystems in South Africa (Nel *et al.* 2007). Nel *et al*. (2007) describe 21% of the rivers of the Cape Floristic Region as being "*Critically Endangered*", 64%, as "*Endangered*", and 14% as "*Vulnerable*". A river ecosystem is described as *critically endangered* and inadequately represented in the country when < 20% of its total original length remains intact. *Endangered* rivers have 20-40% of their total length intact, having lost significant amounts of their natural habitat, thus compromising their ability to support ecological and evolutionary processes. *Vulnerable* rivers have 40-60% of their total length intact and have lost some of their original natural habitat and in danger of compromising their ability to support ecological and evolutionary processes with continued loss of habitat (Nel *et al.* 2007).

3.1.6 Aquatic Ecoregions of the Cape Floristic Region

Aquatic ecoregions have been established for South Africa based on physiography (relief, slope form, drainage density, stream frequency, area with slopes < 5%, and altitude), climate (mean annual precipitation, coefficient of variation (%) of annual precipitation, rainfall concentration index, and rainfall seasonality), geology and soils, and potential natural vegetation (Kleynhans *et al.* 2005). Of the 31 aquatic ecoregions identified for South Africa, ten are expressed in the Cape Floristic Region. Three of these are only in a single catchment (Namaqua Highlands and Western Coast Belt in the Olifants-Doring and the Drought Corridor in the Gamtoos), while a further two are expressed in the same three catchments (Great Karoo and Nama Karoo in the Olifants-Doring, Gouritz and Gamtoos). The Great Karoo and Southern Fold Mountains ecoregions account for more than 55% of the area of the Cape Floristic Region (Great Karoo 30% and Southern Foul Mountains 25%). [Note: The figures presented in the following sections regarding the proportion of each aquatic ecoregion, ecological status classification, and areas transformed were calculated using Spatial Analyst package in the Arc View 3.3 GIS package.]

The Southern Folded Mountains and South Western Coastal Belt have the highest proportion of endangered and critically endangered rivers (94% and 90% respectively). The Western

Folded Mountains (48%), Western Coastal Belt (30%), Great Karoo (27%), Southern Coastal Belt (27%) and Nama Karoo (26%) have more than 25% of their river length endangered or critically endangered.

3.1.7 Major Catchments of the Cape Floristic Region

Three river systems dominate the Cape Floristic Region. The Olifants-Doring system covers 27% of the region, the Gouritz 25%, and the Gamtoos 19%. The remaining major systems are the Breede (7%), Berg (5%), Verlorenvlei (1%) and Heuningnes (1%). In this section, the characteristics of the river systems form the focus of the discussion. The majority of the discussion in this section is based on Skelton (1984), supplemented with original calculations. The ichthyofauna of the river systems is discussed in Section 3.2.

<u>Olifants-Doring</u>

The system is formed by two major tributaries, the Olifants and the Doring Rivers. The Olifants tributary is perennial and has a higher rainfall than the annual Doring tributary, which is rain-shadowed by the Cederberg Mountains. The tributaries of the Olifants are rocky streams with steep gradients, while those of the Doring are longer, with shallower gradients. The Cederberg mountain tributaries of both the Olifants and Doring are generally clear streams with rocky substrates. Water quality is characterised by low pH and mineral concentrations. The water of the Olifants tributary is similar to that of its tributaries until the confluence with the Doring. The Doring passes over Bokkeveld and Dwyka formations, resulting in alkaline and highly mineralised waters. The Olifants-Doring system contains eight aquatic ecoregions and is the largest catchment in the region. The major aquatic ecoregions drained are the Great Karoo (30% of catchment), Nama Karoo (22%), Western Folded Mountains (19%), and Western Coastal Belt (17%). Only 19% of the river length in the Olifants-Doring is considered endangered or critically endangered.

<u>Verlorenvlei</u>

The Kruis River of the Verlorenvlei system is the largest of the independent rivers that cross the coastal plain south of the mouth of the Olifants-Doring. It is adjacent to the Berg River and was part of the Berg during the Cenozoic (Hendey 1983). This system is largely a low gradient, mostly seasonal, reed-bed river with a mud or silt substrate, although some of the tributaries, specifically those draining the Piketberg, have cobble beds. The Verlorenvlei drains the Southern Folded Mountains (59% of catchment), the Western Folded Mountains (17%), and the South West Coastal Belt (24%) ecoregions. Almost 50% of the catchment has been transformed and 83% of the river length is considered endangered or critically endangered.

Berg

The Berg River drains the western slopes of the Cape fold mountains towards the Atlantic Ocean. The mountain tributaries have rocky or sandy substrates and waters with low pH, dissolved solids and ions. The waters of the main stem in the middle and lower reaches become highly mineralised, due to the underlying Malmesbury formation. The Berg drains the South West Coastal Belt (90% of catchment), the Western Folded Mountains (8%), and the Southern Folded Mountains (2%) ecoregions. Almost 65% of the catchment has been transformed and 85% of the river length is considered endangered or critically endangered.

Diep, Liesbeek, and Eerste

The Diep, Liesbeek and Eerste rivers drain the South West Coastal Belt ecoregion through severely transformed landscapes (< 30% natural). More than 90% of their river length (the Diep 98%) is considered endangered or critically endangered. The waters of these rivers are similar in character to the middle and lower reaches of the Berg River.

Palmiet

The Palmiet River drains the Southern Folded Mountains through highly transformed landscapes (about 50% natural). More than 40% of the river length is considered endangered or critically endangered. The waters of the Palmiet are oligotrophic, deeply peat stained and highly acidic. The majority of the upper reaches are highly transformed while the lower reaches flow through the Kogelberg Biosphere Reserve.

Overberg and the Agulhas Plain

A series of short streams drain the Southern Coastal Belt ecoregion near Cape Agulhas, most arising in the Southern Folded Mountains ecoregion. The distribution of *Pseudobarbus* redfin in this region suggests that the small independent streams lying east of Cape Agulhas were connected to the Breede during periods of low sea levels (Skelton 1984). More than 55% of the region has been transformed and more than 75% of the river length is considered endangered or critically endangered. The waters of the Overberg and Agulhas Plain are oligotrophic, deeply peat stained and highly acidic. High levels of evaporation result in many of the rivers on the Agulhas Plain becoming saline in their lower reaches.

Breede

The Breede River system is well watered and perennial with clear acidic water draining the Western Folded Mountains (30% of catchment), Southern Folded Mountains (29%) and the Southern Coastal Belt (41%) ecoregions. The two major tributaries, the Breede and Riviersonderend Rivers, flow south-east along strike valleys. Tributaries draining the fold mountains are clear acidic waters with cobble beds, whereas those draining the coastal belt become saline due to high levels of evaporation in summer. About 40% of the catchment has been transformed and 46% of the river length is considered endangered or critically endangered.

Duivenhoks and Goukou

These two independent rivers east of the Breede River were connected to the Breede during periods of lower sea levels (Skelton 1984). The Duivenhoks and Goukou predominantly drain the Southern Coastal Belt (> 95% of catchment) with their headwaters in the Southern Folded Mountains ecoregions. Almost 45% of the catchments have been transformed and more than 90% of the river length is considered endangered or critically endangered.

Gouritz

The Gouritz is a complex system with three major tributaries that drain the southern escarpment: the Buffels, the Dwyka-Gamka and the Olifants. [This Olifants River should not be confused with the Olifants River of the Olifants-Doring System.] Each main tributary is fed by smaller tributaries from the northern slopes of the Swartberg Range and breaches this range independently to unite within the large valley between the Swartberg and the Langeberg-Outeniqua mountains (Skelton 1984). The Gouritz then passes through the Langeberg-Outeniqua ranges to reach the Indian Ocean. Most of the drainage falls within the rain shadow of the coastal fold mountains. The mountain tributaries draining Table Mountain formations are clear and acidic, while the main tributaries have turbid, alkaline, and mineralised waters resulting from the underlying Bokkeveld and Cretaceous formations of marine origin, or Karoo beds north of the fold mountains. The Gouritz drains the Great Karoo (44% of catchment), the Southern Folded Mountains (49%), and the Southern Coastal Belt (4%) ecoregions. Less than 10% of the catchment has been transformed and 25% of the river length is considered endangered or critically endangered.

Southern Cape

The rivers of the Southern Cape are perennial, having no clear seasonal variation in flow and have deep brown stained low pH (4.5 - 5.5) water with low levels of suspended solids. The mean annual rainfall is 800 – 900 mm/a and distributed throughout the year. The Southern

Cape catchments drain the Southern Coastal Belt (98% of catchment) and the South Eastern Coastal Belt (2%) ecoregions. Almost 45% of the catchments have been transformed and less than 11% of the river length is considered endangered or critically endangered.

<u>Keurbooms</u>

The Keurbooms is perennial, having no clear seasonal variation in flow, and has deep, brown-stained, low pH (4.5 - 5.5) water with low levels of suspended solids. The Keurbooms drains the South Eastern Coastal Belt (84% of catchment) and the Southern Folded Mountains (16%) ecoregions. Less than 12% of the catchment has been transformed and 12% of the river length is considered endangered or critically endangered.

<u>Krom</u>

The Krom River flows in a strike valley and has acid, peat-stained waters typical of the fold mountain region. The Krom drains the Southern Folded Mountains (35% of catchment area) and the South Eastern Coastal Belt (65%) ecoregions. Almost 40% of the catchment has been transformed and 45% of the river length is considered endangered or critically endangered. The Krom River joined the Gamtoos during times of lower sea level.

<u>Gamtoos</u>

The Gamtoos River system may be considered in two parts, the tributaries draining south-east within the strike valleys of the fold mountains and the Groot River which drains a large inland area from over the Great Escarpment. The Groot River traverses the relatively soft formations of the Karoo Supergroup resulting in turbid and silty water. The mountain tributaries drain Table Mountain sandstones and are clear acidic waters with sandy-rocky substrates. The Gamtoos drains the Great Karoo (40% of catchment), the Southern Folded Mountains (33%), and the Drought Corridor (20%) ecoregions. Less than 3% of the catchment has been transformed and 32% of the river length is considered endangered or critically endangered.

<u>Swartkops</u>

The Swartkops River flows south-east into Algoa Bay via the strike valleys of the Great Winterhoek and Elands mountains. Rainfall within the catchment is high (650-700 mm/a) and the streams mainly perennial. The water of the mainstem is mildly acidic and clear arising from Table Mountain formations. The salinity increases in the lower reaches as a result of the underlying marine beds of the Uitenhage Group. The Swartkops drains the Southern Folded Mountains (50% of catchment) and the South Eastern Coastal Belt (50%) ecoregions. Almost 25% of the catchment has been transformed and 30% of the river length is considered endangered or critically endangered.

3.1.8 Implications of Climate Change for the Cape Floristic Region

Climate change predictions for rainfall patterns and surface run-off in the Cape Floristic Region are not encouraging for the conservation of freshwater ecosystems and freshwater biota. It has been predicted by the end of the 21st century the annual rainfall for the western drainage basins of the Cape Floristic Region (including Cape Town) will drop by up to 20%, while a drop of 10% is predicted for the eastern and southern drainage basins (de Wit and Stankiewicz 2006). The drop in rainfall is predicted to result in a reduction in surface run-off. De Wit and Stankiewicz (2006) predict that areas near Cape Town will suffer most, losing more than half their perennial supply of surface water. These reductions in surface run-off will place increasing pressure on the already stressed water resources of the region and intensify competition for water resources between the human population and the ecological reserves legally required for the maintenance of river functioning by the National Water Act of South Africa [See section 3.4.3].

Climate change predictions also forecast increases in the water temperature of the freshwater systems (de Wit and Stankiewicz 2006, Rahel *et al.* 2008, Britton *et al.* 2010a), although no predictions specific to the rivers of the Cape Floristic Region have been published. The impact of climate change on the non-native and native freshwater fishes of the Cape Floristic Region has not been evaluated. It is particularly important that this subject receives attention, as many of the popular aquarium species and proposed aquaculture species (e.g. *Oreochromis niloticus x Oreochromis mossambicus* hybrids) that are considered "safe" introductions under present conditions, but may be able to survive and reproduce in the wild at elevated temperatures (Zambrano *et al.* 2006).

3.2 Freshwater Fishes of the Cape Floristic Region

Isolation of the river systems has resulted in the evolution of a unique assemblage of freshwater fishes in the Cape Floristic Region. Twenty species of primary freshwater fish have historically been recognised for the Cape Floristic Region (Skelton 2001, Impson *et al.* 2002b). This number has recently been increased to 27, following the genetic and morphological revision of the cyprinid genus *Pseudobarbus* Smith, 1841 (Swartz 2005, Swartz *et al.* 2007, 2009, Tweddle *et al.* 2009) and further increases in the number of recognised species are expected on completion of taxonomic reviews of the genera *Galaxias* and *Sandelia*. [Note: several of the taxa referred to as species in this text, specifically those from the *Pseudobarbus* genus, have not been formally described as separate species. It would be more appropriate to refer to these taxa as either taxa or Evolutionary Significant Units

(ESU) *sensu* Moritz (1994). In this work, the names for the respective ESU included in the 2009 IUCN Red Data List are used.] A full list of the currently recognised primary freshwater fish species of the Cape Floristic Region is presented in Table 3.1. In addition, a number of secondary freshwater species are found in the rivers of the region: anguillid eels: the longfin eel *Anguilla mossambica* Peters, 1852; the shortfin eel *A. bicolor bicolor* McClelland, 1844; and the giant mottled eel *A. marmorata* Quoy and Gaimard, 1824; the freshwater mullet *Myxus capensis* (Valenciennes, 1936), the flathead mullet *Mugil cephalus* Linnaeus, 1758, the river goby *Glossogobius callidus* (Smith, 1937) and the checked goby *Redigobius dewaalii* (Weber, 1897). The mullets and eels are diadromous, both breeding in the oceans while completing their life cycles in fresh water. The gobies are mostly estuarine, but have been recorded as resident in fresh waters. This thesis concentrates on the primary freshwater species of the region, because they are the more threatened group and have been most impacted by the introduction of non-native fishes, but acknowledges the need to include the secondary freshwater fishes in conservation planning. All of the secondary freshwater species are currently listed as being of least concern (Tweddle *et al.* 2009).

Four families of primary freshwater fishes are native to the Cape Floristic Region, the Galaxiidae, Austroglanididae, Anabantidae and Cyprinidae. The fish of the Cape Floristic Region exhibit typical characteristics of old, well-established mountain faunas: a high degree of endemicity, isolated and geographically restricted ranges, relative inflexibility in life history styles, and low resilience to disturbance, especially to introduced piscivorous fish (Skelton 1987). Regional endemicity is high, with 89% of the species limited to the region Most are relatively small, attaining < 150 mm SL when adult. None is entirely piscivorous, although the anabantid *Sandelia capensis* (Cuvier, 1831) and some of the larger cyprinids, such as Clanwilliam yellowfish, *Labeobarbus capensis* (A. Smith, 1841), are partly piscivorous as adults (Skelton 2002).

3.2.1 Galaxiidae

The family Galaxiidae is confined to the southern hemisphere and contains both primary freshwater and diadromous species. The galaxioid fishes have an ancient evolutionary history, being derived from the lower euteleostean fishes, with galaxiid fossils having been dated back to the Miocene (McDowall 2006). A single "species", *Galaxias zebratus* (Castelnau, 1861), the Cape galaxias, is confined to the Cape Floristic Region. Barnard (1934) found difficulty in finding clear-cut and constant characters of specific value for Cape galaxias. Skelton (2001) and McDowall (2006) considered *G. zebratus* a species-complex with genetic and morphological work confirming that as many as 10 species may be ultimately recognized from the Cape Floristic Region, many known only from a single location (P. Skelton and E. Swartz SAIAB, pers. comm. 2011). Skelton (1994) described the family Galaxiidae as having Gondwanan distribution, although this has been questioned by McDowall (2006), who cites oceanic dispersal as the mechanism for their current distribution. *G. zebratus* is listed as Data Deficient in the 2007 IUCN Red Data List (Tweddle *et al.* 2009).

3.2.2 Anabantidae

The anabantid genus *Sandelia* Castelnau, 1861 is endemic to South Africa and represented by two morphologically similar species, *Sandelia capensis* and *S. bainsii* Castelnau, 1861, both of which have vestigial labyrinth organs. The monophyly of *Sandelia* has been debated as the species differ significantly in several characters (Rüber *et al.* 2006): for instance, *S. bainsii* develops contact organs when breeding, while *S. capensis* does not (Cambray 1997b, 2004). The genus has a disjunct distribution, with *S. capensis* having a continuous distribution from Verlorenvlei in the West to the Coega River in the east, while *S. bainsii* is limited to the Winterberg-Amatola range east of the Cape Floristic Region (Skelton 2001). Rüber *et al.* (2006) suggest that *S. capensis* is a sister group to *Anabas* Cloquet, 1816 of Asia and the African *Ctenopoma* "*multispine"* species group Peters, 1844. The split between the *Ctenopoma multispine* species group and *S. capensis* is estimated to have occurred during the early Miocene, about 24 Mya (Rüber *et al.* 2006). Roos (2005) identified two historically isolated lineages of *S. capensis,* separating by the Hottentots-Holland mountains. Molecular clock results indicated that this split occurred in the Pliocene (2-4 Mya) (Roos 2005). Within these lineages, further divergence has taken place. Further morphological and genetic work is required to resolve the taxonomy of the *S. capensis* species complex. *S. capensis* forms a fundamental component of the Cape Floristic Region fish assemblage and is one of its characteristic species. *S. capensis* is listed as Data Deficient in the 2007 IUCN Red Data List (Tweddle *et al.* 2009).

3.2.3 Austroglanididae

The siluriform catfish family Austroglanididae has been described by taxonomists as unusual, problematic and difficult to place in phylogenetic trees (Sullivan *et al.* 2006). The origin of catfishes probably occurred before the breakup of Gondwana in the late Mesozoic (Teugels 1996). The Austroglanididae is an ancient family with the closest relatives amongst African catfish (Paul Skelton, SAIAB, pers. comm. 2008). Austroglanididae, known only from the Orange and Olifants River Systems in South Africa, comprise one genus, *Austroglanis* Skelton, Risch & De Vos, 1984, of three species: the spotted rock catlet *Austroglanis* *barnardi* (Skelton, 1981) and Clanwilliam rock catlet *A. gilli* (Barnard, 1934) from the Olifants River and the rock catlet *A. sclateri* (Boulinger, 1901) from the Orange River (Skelton 2001). *A.gilli* is listed as Vulnerable and *A. barnardi* as Endangered in the 2007 IUCN Red Data List (Tweddle *et al.* 2009).

3.2.4 Cyprinidae

The Cyprinidae are a large family of primary freshwater fish with about 275 genera and more than 1600 species from Africa, Europe, Asia, and North America (Skelton 2001). They are the largest family in southern Africa with eight genera and more than 80 species. Cyprinids have a wide range of sizes and shapes, life-history styles and habitat preferences. They lack teeth but have strong pharyngeal bones. They lack a true stomach and, especially in detritus feeders such as labeos, the gut may be extended and convoluted. Cyprinids are often strong swimmers with males differing from females, having longer fins, brighter breeding colours and tubercles on the head, body and fins. There are a large number of varied barbine cyprinids in Africa, most nominally included in the genus *Barbus* Cuvier and Cloquet 1816, on the basis of general morphological similarity. It is now recognised that the genus is valid only for certain tetraploid European and north-west African species. Reclassification of African barbs is in progress, based on a new understanding of their relationships, genetics, chromosomes, biology and morphology (Skelton 2001).

The primary freshwater fish fauna of the Cape Floristic Region is dominated by cyprinids (23 species), mainly barbine species (21 species). Of the barbine species, 18 are tetraploid (86%), *Labeobarbus capensis* is hexaploid (5%) while *Barbus anoplus* Weber, 1897 (a putative species-complex) and *Barbus pallidus* A. Smith, 1841 are diploid. This high proportion of polyploid species is unusual when compared to the whole family (85% diploid, 8-10% tetraploid and 4% hexaploid) (Klinkhardt 1998). Two groups of tetraploid 'barbs' occur in the region: the genus *Pseudobarbus* Smith, 1841 and the sawfin barbs (*B. calidus* Barnard, 1938; *B. erubescens* Skelton, 1974; and *B. serra* Peters, 1864 from the Olifants River System and *B. andrewii* Barnard, 1937 from the Berg and Breede Rivers). The tetraploid *Barbus* of South Africa lie in the same clade and are closely related to the small diploid African *Barbus* species (Machordom and Doadrio 2001, Tsigenoloulos *et al.* 2002), implying that their tetraploidy is ancient (Tsigenoloulos *et al.* 2002).

Pseudobarbus redfin

Historically, the genus *Pseudobarbus* was though to consist of seven species, six endemic to the Cape Floristic Region. The seventh species, *P. quathlambae* (Barnard, 1938), found only in the headwaters of the Orange River in Lesotho, consists of two distinct populations, now recognized as separate taxa (Swartz 2005, Swartz *et al.* 2007, 2009, Tweddle *et al.* 2009). *Pseudobarbus* is a tetraploid monophyletic genus (Machordom and Doadrio 2001, Tsigenoloulos *et al.* 2002) descended from an ancient lineage (Skelton 2001) and most closely related to the tetraploid sawfin barbs of the Cape Floristic Region (Machordom and Doadrio 2001). Genetic and morphological studies of *Pseudobarbus* have revealed that at least 14 taxa should be recognised for the Cape Floristic Region (Swartz 2005, Swartz *et al.* 2007, 2009). See Table 3.1 for the full list of described and undescribed *Pseudobarbus* taxa.

Tetraploid Sawfin Barbs

The South African tetraploid barbs consist of two closely-related large-bodied species, sawfin *Barbus serra* Peters, 1864 (Olifants River System) and whitefish *B. andrewii* Barnard, 1937 (Berg and Breede Systems) and two small-bodied sister species, Clanwilliam redfin *B. calidus* Barnard, 1938 and Twee River redfin *B. erubescens* Skelton, 1974 (Olifants River System). Each of these clades is most closely related to each other and both clades are closely related to the genus *Pseudobarbus* and *B. trevelyani* Günther, 1877 from the Eastern Cape (Machordom and Doadrio 2001, Tsigenoloulos *et al.* 2002). *B. erubescens* is currently listed a Critically Endangered, *B. serra* and *B. andrewi* as Endangered and *B. calidus* as Vulnerable in the 2007 IUCN Red Data List (Tweddle *et al.* 2009).

Diploid African Barbs

The *Barbus anoplus* group is endemic to the cooler areas of South Africa and may comprise several described (*B. anoplus* Weber, 1897, *B. gurneyi* Günther, 1868, *B. motobensis* Steindachner, 1894, *B. amatolicus* Skelton, 1990, and *B. breviceps* Trewavas, 1936) and undescribed taxa (Skelton 2001). *B. anoplus* is absent from the streams in the Cape Fold mountains, but is found in the Karoo catchments of the Olifants, Gouritz, Gamtoos and Sundays Rivers (Skelton 2001). A genetic and morphological study of the *B. anoplus* group is underway (Paul Skelton, pers. comm., 2011), but no results are available. The species is listed as Data Deficient in the 2007 IUCN Red Data List (Tweddle *et al.* 2009). *B. pallidus* A. Smith, 1841 is endemic to South Africa and has two distinct populations, one in the headwaters of the Vaal River and the other from the Krom River to the Great Fish River in the Eastern Cape (Skelton 2001), at the eastern extremity of the Cape Floristic Region. This species is listed a Least Concern in the 2007 IUCN Red Data List (Tweddle *et al.* 2009)

Labeobarbus

Species of the genus *Labeobarbus* Rüppell, 1936 are hexaploid barbine cyprinids characteristic of many African rivers and lakes (Ollermann and Skelton 1990, Skelton 2001). They are long-lived and grow to large sizes (Skelton 2001). One species, the Clanwilliam yellowfish *L. capensis* of the Olifants River System, is native to the Cape Floristic Region. Another Orange River species, *L. aeneus* (Burchell 1822), has been introduced into the Gouritz River System. *L. capensis* is listed as Vulnerable in the 2007 IUCN Red Data List (Tweddle *et al.* 2009).

Labeo

Labeo Cuvier, 1817 is a diverse genus, widely distributed in Africa and south-east Asia (Reid 1985). Labeos are specialist feeders on algae and detritus, having large inferior mouths with well-developed lips, grinding pharyngeal teeth, and very long, coiled intestines (Reid 1985). Labeos are strong swimmers and migrate *en masse* upstream to breed. Two of the four species in the *Labeo umbratus* group occur in the Cape Floristic Region, the Clanwilliam sandfish, *L. seeberi* Gilchrist and Thompson, 1911, and the moggel, *L. umbratus* (A. Smith, 1841). Based on general morphology, the *Labeo umbratus* group is more closely related to the Asian labeos than other African labeos (Skelton 1994). *L. umbratus* is currently listed as Least Concern and *L. seeberi* as Endangered in the 2007 IUCN Red Data List (Tweddle *et al.* 2009).

3.2.5 Current IUCN Conservation Status

Three species, *Sandelia capensis*, *Galaxias zebratus* and *Barbus anoplus*, are listed as data deficient in the 2007 IUCN Red Data List (Tweddle *et al.* 2009). Of the 27 primary freshwater fish taxa recognised for the Cape Floristic Region (Table 3.1), six are listed as critically endangered (22%), ten as endangered (37%), three as vulnerable (11%), three as near threatened (11%), two as least concern (7%), and three (11%) as data deficient (Impson 2007, Tweddle *et al.* 2009).

3.2.6 Current Threats

The major threats to the 24 endemic primary freshwater fish taxa of the Cape Floristic Region are non-native fish (a threat to 23 species), habitat destruction (to 18 species), pollution, including pesticides (2 species), utilisation (3 species) and genetic integrity (4 species) (Tweddle *et al.* 2009). Genetic integrity is of concern when closely-related taxa are translocated, potentially homogenizing genetic diversity or promoting hybridisation between different but closely-related taxa (Swartz 2008). Researchers and conservation officials in the

Cape Floristic Region agree that non-native fishes are the most significant threat to the long-term survival of indigenous freshwater fish assemblages in this region (Barnard 1943, Coke 1988, Skelton *et al.* 1995, Impson and Hamman 2000, Skelton 2000, Impson *et al.* 2002a, b, Skelton 2002, Cambray 2003a, b, Impson 2007, Tweddle *et al.* 2009).

3.3 History of Freshwater Fish Introductions

Non-native fish, Eurasian carp *Cyprinus carpio* Linnaeus, 1758 and goldfish *Carassius auratus* (Linnaeus, 1758), were introduced into the Cape Floristic Region as early as the late 1700s by Dutch and English settlers (de Moor and Bruton 1988, Picker and Griffiths 2011). After numerous failed attempts, brown trout *Salmo trutta* Linnaeus, 1758 were successfully imported into South Africa in the 1890s and established self-sustaining populations in several rivers (de Moor and Bruton 1988). Rainbow trout *Oncorhynchus mykiss* (Walbaum, 1792) were later imported and released. By 1920, rainbow and brown trout had been widely introduced into cooler rivers across South Africa by enthusiasts, angling societies and government authorities (Skelton 2000). Protecting non-native fishes and ensuring the high water quality needed for trout to survive were the primary freshwater conservation concerns at that time. In the 1920s, an overview of the physical state of the rivers, the aquatic fauna and their suitability for the development of sport-fisheries led to the creation of the provincial 'Inland Fisheries' departments that later evolved to form the provincial nature conservation departments in the 1950s and 1960s (Hey 1926, Coke 1988, Skelton 2000). Recommendations for the introduction of other sport fishes for warm-water fisheries, in particular North American smallmouth bass *Micropterus dolomieu* Lacepède, 1802 and largemouth bass *M. salmoides* (Lacepède, 1802), were implemented and constituted freshwater fishery policy throughout South Africa for the following decades (Coke 1988, Skelton 2000). By the late 1930s, researchers became aware of the negative ecological impact of non-native fish on the native fishes (Barnard 1943). The establishment and enhancement of sport fisheries continued to dominate state policies until the 1970s (Coke 1988, Skelton 2000), with regulations introduced to protect non-native species. Non-native species were considered to be model species reflecting the health of the environment (Skelton 2000). The well-being of native fishes was considered irrelevant until the 1960s, as they were considered unsuitable fishing targets, or as problem species hindering the establishment of non-native sport fishes (Skelton 2000). Occasionally, South African species, such as Mozambique mouthbrooder *Oreochromis mossambicus* (Peters, 1852) and banded tilapia *Tilapia sparrmanii* A. Smith, 1840, were cultured and distributed as fodder species for the non-native sport fishes, along with the North American bluegill sunfish *Lepomis macrochirus* Rafinesque, 1819 (de Moor and Bruton 1988).

In the 1980s, the Cape Department of Nature and Environmental Conservation declared a moratorium on the stocking of fish into the rivers of the Cape Floristic Region (Coke 1988, Skelton 2000). Since 1988, no new non-native species have been recorded in the Cape Floristic Region. This moratorium failed to prevent the further distribution of non-native fish within the Cape Floristic Region and secondary introductions of species already present in the region are regularly reported (Impson *et al.* 2002a, b, Skelton 2002, Impson 2006, 2007). Of specific concern is the sharptooth catfish *Clarias gariepinus* (Burchell, 1822). This is native to South Africa, but has been introduced into several river systems of the Cape Floristic Region by anglers, farm labourers and land-owners, raising concerns regarding its impact on the native fish of the region (Cambray 2003c, Impson 2006, 2007). The introduction of additional species is possible, but these have not been recorded by researchers or conservation officials.

To date, 24 species of freshwater fish have been introduced to the inland waters of the Cape Floristic Region for recreational angling, biocontrol or aquaculture (Table 3.2). Of these, seven failed to establish self-sustaining populations and one, the Eurasian perch *Perca fluviatilis* Linnaeus, 1758, has been extirpated. Twelve of the 22 species are currently viewed as angling species. In addition, triploid grass carp *Ctenopharyngodon idella* (Valenciennes, 1844) have been introduced to farm dams for aquatic weed control. Although these fish are sterile, they are long-lived and grow to be very large (over 1 m), and escapees might have a large impact on aquatic vegetation and invertebrates (Pípalová 2006).

Three endemic species been translocated within the Cape Floristic Region. Cape kurper was introduced into a dam in the Suurvlei River of the Twee River Catchment (Olifants-Doring System) by a private landowner in the 1950s (Hamman *et al.* 1984) and subsequently escaped. The conservation authority in the Western Cape (Cape Department of Nature and Environmental Conservation) intentionally introduced the Clanwilliam yellowfish above waterfall barriers in the Twee (Impson *et al.* 2007), Ratels (Impson 2010) and Boontjies rivers (Impson and Tharme 1998) of the Olifants-Doring catchment as a conservation measure for this species in the 1980s. The Breede River redfin *Pseudobarbus* sp. "burchelli Breede" were cultured in outdoor ponds at the Jonkershoek fish hatchery by the Cape Department of Nature Conservation from which they escaped and invaded the Eerste River near Stellenbosch in the 1980s, displacing the native *Pseudobarbus* population of the Eerste River (ND Impson, CapeNature, pers. comm.)

3.3.1 Vectors and Pathways

The vectors and pathways for the introduction of non-native fishes have been discussed in Section 2.3 and are summarised in Table 2.1. The risks posed by each vector-pathway combination in the Cape Floristic Region are summarised in Table 3.3. There is no risk of introductions via the ballast water vector, because there are no navigable rivers or estuarine harbours in the region. Although the nature conservation authority was historically responsible for the majority of the introductions, these practices have ceased and the policies revised. It is now necessary to obtain a permit to introduce or transport non-native fish. Although legislation and permit review system are in place, enforcement remains inadequate (Impson *et al.* 2002b, Impson 2007). Nature conservation is now considered a low-risk vector for the future introduction of non-native fish (Richardson *et al.* 2004).

The rivers of the Cape Floristic Region fall entirely within the region, but the region extends over provincial boundaries. The majority of the region falls within the Western Cape Province but extends into the Northern Cape and Eastern Cape Provinces in the north and east respectively. There is therefore a risk that species introduced into neighbouring provinces could enter the region unaided. The majority of fish species not native to South Africa have already been introduced into all three provinces, therefore the translocation of native species poses the greater risk. Of greatest concern is the potential for the introduction of Orange River species, particularly *Barbus*, *Labeobarbus*, *Austroglanis*, *Clarias* and *Labeo* species, into the Olifants River System in the Northern Cape Province. Several inter-basin transfer schemes have been built in the Cape Floristic Region and these have resulted in the transfer of non-native fish between catchments in the region, e.g. Theewaterskloof-Berg-Eerste transfer scheme, and from other regions, e.g. the Orange-Fish transfer scheme, that transfers water from the Orange River to the Swartkops River via the Fish River and other rivers in the Eastern Cape. The existing inter-basin transfer schemes remain high-risk vectors for the transfer of native and non-native fish between catchments and regions. On a smaller scale, agricultural furrows pose a similar risk for non-native species introductions from farm dams into streams and rivers and the transfer of native and non-native species between adjacent catchments.

Several non-native fish have been introduced in the Cape Floristic Region for the bio-control of mosquitoes and aquatic vegetation. Mosquitofish *Gambusia affinis* (Baird and Girard, 1853) and guppies *Poecilia reticulate* Peters, 1859 were evaluated as mosquito control agents, but only mosquitofish established recruiting populations (de Moor and Bruton 1988, Skelton 2001). Mosquitofish are no longer officially distributed for bio-control, but illegal spread continues. Of the species introduced for the control of aquatic vegetation, only triploid grass carp are currently available for introduction. All other species introduced for aquatic vegetation control failed to establish recruiting populations and have been extirpated (de Moor and Bruton 1988). Grass carp are produced for introduction in the Cape Floristic Region at a single hatchery near Montagu. For the bio-control vector, legal introduction poses a low risk, because only certified triploid grass carp are available for introduction. There is a moderate risk that these legally-introduced grass carp will escape from the farm dams into natural water courses. The risk of illegal introduction of mosquitofish remains high.

The ornamental fish trade in South Africa operates on the basis of a "white list" for species permitted for import and a "black list" for species that may not be imported. These lists were recently reviewed during the development of the regulations for the National Environmental Management: Biodiversity Act (NEM:BA) [Section 3.4.1] and should be comprehensive, although no formal risk assessment procedure was used in compiling the lists. All pathways of introduction of ornamental fish into natural water-bodies are illegal. There is a high risk of illegal release by aquarists and numerous locations in Cape Town and Port Elizabeth contain populations of released ornamental fish. No survey has been conducted to determine whether any of the unwanted aquarium fish have established recruiting populations, because it is widely believed that the species released would not survive local climatic conditions (ND Impson CapeNature, pers. comm.). Scenarios developed for climate change predict that water temperatures will increase in the Cape Floristic Region (de Wit and Stankiewicz 2006) and it may be necessary to re-evaluate the risk of ornamental fish establishing under various climate change scenarios (Xenopoulos *et al.* 2005, Rahel *et al.* 2008).

The aquaculture industry in South Africa has received substantial governmental support to generate employment in previously disadvantaged groups. These initiatives have resulted in a renewed interest in aquaculture and numerous species have been proposed for aquaculture in the Western Cape. The climate of the Western Cape, and indeed most of South Africa, is not conducive to the high growth rates for aquaculture species and locally produced fish may not be competitive on the global market. However, backyard aquaculture of species such as sharptooth catfish, Mozambique mouthbrooder, or carp, may be profitable and provide valuable income for entrepreneurs in impoverished communities. Commercial aquaculture ventures are strictly controlled and pose a low risk for legal release, escapement and contamination pathways of non-native fish introduction. However, informal aquaculture, should it be developed further, poses a high-risk pathway for illegal release of non-native species into natural water courses. Some argue that these species have already established in most of the catchments of the Cape Floristic Region and thus pose little risk of further introduction (ND Impson CapeNature, pers. comm.). These species are known to be invasive and have detrimentally impacted native biodiversity. If appropriate non-native fish control techniques are developed in the future, the risk of re-introduction of the eradicated species would be greater if they have commercial value and are present in diffuse population sources in back yards all over the region.

Anglers world-wide are renowned for the illegal release of non-native species. South African anglers are no exception. The illegal release of non-native fish by recreational and subsistence anglers remain a high risk pathway for new introductions. Subsistence anglers are more likely to introduce large-bodied fish, such as sharptooth catfish, while recreational anglers are more likely to introduce game fish, such as trout, bass and yellowfish. There is a low risk of non-native fish introduction from the legal release of non-native species while there is a medium risk of contamination of bait fish used by subsistence anglers.

3.3.2 Extent of Freshwater Fish Invasion

Although it is known among researchers that the freshwater ecosystems of the Cape Floristic Region have been extensively invaded, the magnitude of the freshwater fish invasion has not been fully documented. Estimates of the extent of invasion have been made. For example Hamman (2008) suggests that 90% of the rivers in the Cape Floristic Region are invaded by non-native fish, but these do not present the necessary data to support the figures presented. In order to conserve the remaining native fish populations, it is important that the scale of freshwater fish invasion is communicated to the general public in order to prevent further introductions. Therefore, an estimate of the extent of freshwater fish invasion in the Cape Floristic Region is presented based on the best available information. Due to the limited information available on the distribution of freshwater fishes in the Cape Floristic Region, native and non-native, the data are based on published literature and expert opinion. The estimate was compiled at two scales. Firstly, the extent of invasion was evaluated at a catchment level. Here it was assumed that a species recorded as having successfully established in the catchment had invaded the entire catchment. This is clearly not true as many species, such as the salmonids, have limited tolerance levels and will have restricted ranges within the catchments. However, most of the species introduced, such as cyprinids (*Cyprinus carpio*, *Tinca tinca* (Linnaeus, 1758) and *Labeobarbus aeneus*), centrarchids (*Lepomis macrochirus* and *Micropterus* sp.), silurids (*Clarias gariepinus*) and cichlids (*Tilapia sparrmanii* and *Oreochromis* sp.), have invaded extensive areas of the region. Secondly, the extent of freshwater fish invasion in three of the five major catchments of the

Cape Floristic Region was compiled based on the river habitat invaded. [See Appendix A.3 for details of the methods]

Reliable records of fish presence were available for 42 of 58 catchments of the Cape Floristic Region, covering 86% of the surface area of the region. A total of 16 non-native fish species of have successfully established in the Cape Floristic Region (Table 3.2). The Mozambique mouthbrooder *Oreochromis mossambicus* has been introduced into the largest number of catchments (30 catchments, 97% of the study area), but largemouth bass *Micropterus salmoides* has been introduced into catchments representing the largest area (97% of the study area, 26 catchments). Nine species have been introduced into catchments representing more than 90% of the study area: common carp; largemouth bass, smallmouth bass, spotted bass, bluegill sunfish, rainbow trout, Mozambique mouthbrooder and banded tilapia (Figure 3.2).

Ten or more non-native fish species have been introduced into five of the catchments: the Olifants-Doring; Berg; Eerste-Kuils; Breede and Gouritz catchments (Figure 3.3). These catchments represent 68% of the study area. A further eight catchments host between seven and nine non-native species: the Keisers, Lourens, Palmiet, Bot, Klein, Heuningness, Gamtoos and Bakens (23% of the study area). These 13 catchments together represent more than 90% of the study area. Only seven catchments, together representing 0.8% of the study area, contain no non-native fish.

The extent of river habitat invaded was explored using river length for three river systems: the Olifants-Doring; Berg; and Breede. The invasion status of 31% of the perennial river length in these three catchments was unknown. Of the length of river where the invasion status is known, an estimated 85% has been invaded by non-native fishes. At least 10% of the perennial river length of the three rivers has not been invaded. The Berg River has the highest level of invasion, with an estimated 93% of river length invaded. Conversely, the Olifants-Doring River has the lowest level of invasion with an estimated 80% of river length invaded. When these estimates are corrected for the three-dimensional nature of rivers (including an estimate of width and depth) to represent the proportion of river habitat invaded (width estimated by Strahler stream order and depth estimated as square root of Strahler stream order) the extent of river habitat invaded in the three catchments is estimated to be 93% Berg River 98%, Breede River 94% and Olifants-Doring 89%).

The results of this analysis suggest that the extent of invasion of the rivers in the Cape Floristic Region exceeds 85%, but further work is required to provide a better estimate. The inclusion of the smaller systems in the region may change this value, but it appears that the

90% level of invasion suggested by Hamman (2008) is a good, possibly even conservative, estimate of the extent of freshwater fish invasion in the region.

3.3.3 Impacts of Freshwater Fish Introductions

As early as the late 1930s, the negative impact of non-native fish on native fish assemblages was recognised, especially in the biodiversity hotspot of the Olifants-Doring system (Barnard 1943). Predation by non-native fish , especially by rainbow trout and smallmouth bass, was recognised as a significant concern (Hey 1926, Barnard 1943, Harrison 1950, 1951a, b, 1952a, b, Skelton 1983, Coke 1988, Skelton 1990, Skelton *et al.* 1995, Skelton 2000, 2002). Despite their negative impacts on native fish, non-native fish continued to be bred and stocked throughout the Cape Floristic Region until the 1980s (Hey 1926, Coke 1988, Impson and Hamman 2000, Impson *et al.* 2002b, Richardson *et al.* 2004, Tweddle *et al.* 2009). The initial impact of non-native fishes on native fish assemblages in the Cape Floristic Region was poorly recorded and not studied in depth, because most impacts occurred before any great interest in conserving the native species arose.

There is a strong correlation between the presence of predatory non-native fish and the absence of native species, a situation analogous to that in the Colorado River (USA) (Marsh and Pacey 2005). This inability of native fish assemblages to persists in the presence of non-native species is particularly evident where centrarchids (basses and sunfish) and salmonids are present. Non-native fish affect the behaviour and composition of the native fish assemblages in the Cape Floristic Region (Woodford and Impson 2004, Woodford *et al.* 2005, Shelton *et al.* 2008) as well as the lower trophic levels, including aquatic invertebrates and algae (Lowe *et al.* 2008). Because the native fishes of the Cape Floristic Region are extirpated following the introduction of non-native fish, the remaining native fish populations can only be conserved, and their ranges increased, through the elimination of the non-native species (Impson 2007).

3.4 Conservation of Freshwater Fishes in South Africa

A rapid decline in the native freshwater fish stocks of South Africa was recognised following the establishment of the nature conservation departments in the 1950s and 1960s (Coke 1988, Skelton 2000) and the completion of province-wide freshwater fish surveys between 1950 and 1980 (Skelton 2000). These surveys clearly indicated the negative impacts of non-native species on the native fishes and as a result conservation authorities changed their policies towards the breeding and stocking of non-native sport fishes (Coke 1988, Skelton 2000).

Research programmes into native fish species and their conservation were subsequently initiated (Coke 1988, Skelton 2000), culminating in the publication of the first IUCN Red Data List for the freshwater fishes of South Africa (Skelton 1977).

Conservation laws in South Africa have until recently been fragmented, uncoordinated and at times conflicting between national and provincial departments and lacked a coherent approach to conservation policy (Algotsson 2009). Prior to 2004, South Africa did not have national legislation that protected non-game fish (Helfman 2007) and any protection afforded to native fishes came from poorly-enforced provincial regulations. The situation changed following the passage in May 2004 of the National Environmental Management: Biodiversity Act (NEM:BA), Act 10 of 2004, administered by the Department of Environmental Affairs and Tourism.

3.4.1 National Environmental Management: Biodiversity Act

NEM:BA provides a framework for enacting regulations that identify and protect rare and threatened species, including species protected under international agreements (e.g. CITES, IUCN). It contains provisions for the protection of at-risk ecosystems and bioregions, rather than focussing solely on species. Ecosystems and species are ranked as Critically Endangered, Endangered, Vulnerable, and Protected using subjective criteria (i.e. not IUCN's quantitative criteria). A list of fishes to be protected under NEM:BA has been drawn up by local expert groups, informed by the IUCN Red Data Lists.

Biodiversity Management Plans

NEM:BA requires that biodiversity management plans be prepared for species (populations or metapopulations), ecosystems, or bioregions to ensure their long-term survival in nature. The development of biodiversity management plans incorporates a rigorous stakeholder engagement process. The plans must be consistent with all levels of legislation (i.e. they must not contradict national or provincial legislation) and provide the implementing body with mechanisms to monitor and report on the progress of implementing the plan. The plans should include the criteria used to select the species, information on the current status of the species, and information on known threats and their impacts on the species. An action plan stating the objectives and actions for dealing with each threat should be developed. Implementing parties and time-frames for implementation of the action plans must be stipulated. Monitoring and reporting plans with annual reports are required. The biodiversity management plan is a legally-binding document with the implementing party directly accountable to the minister of DEA for compliance with the management plan.

Management of Non-native Species

Chapter 5 of NEM:BA focuses on species and organisms posing potential threats to biodiversity. The development of regulations for implementation of this chapter was initiated in 2004 (van Rensburg *et al.* 2011). Several versions of the regulations have been produced, and a revised draft was published for public comment in 2009. NEM:BA requires the listing of non-native and invasive species as Prohibited Species or species that require compulsory control (Category 1a), control as part of an invasive species control programme (Category 1b), regulation by area (Category 2), or regulation by activity (Category 3) (van Rensburg *et al.* 2011). Prohibited Species are non-native species that may not be imported, kept, bred, moved, released or allowed to spread into South Africa. For all other non-native species, a risk assessment is required prior to the issue of permits (van Rensburg *et al.* 2011). In their review of NEM:BA, (van Rensburg *et al.* 2011) recommend: 1) focusing on the management of pathways to reduce introductions; 2) rigorous implementation of quantitative risk assessment; 3) the use of technology to improve surveillance and communication about invasive species, and 4) the creation or mandating of an institution that is responsible for prevention, early detection and rapid response for emerging invasions.

Under NEM:BA, the use of most non-native and many translocated fish species will be regulated by area i.e. zoning (van Rensburg *et al.* 2011). Acknowledging the use of non-native fish for angling and/or aquaculture is widely viewed as recognition of the value of these fishes provide through sport, recreation, income and economic benefits (van Rensburg *et al.* 2011). The negative impacts of these introductions on aquatic ecosystems are also recognised. The demarcation of areas for use of various non-native species allows for trade-offs between conservation priorities and recreational (and economic) interests (van Rensburg *et al.* 2011). However, enforcement of this legislation is complicated by the informal nature of illegal stocking of non-native and translocated fish within the country.

3.4.2 National Freshwater Ecosystem Priority Areas

The National Freshwater Ecosystem Priority Areas (NFEPA) project aims to identify a national network of freshwater ecosystem conservation priority areas, including rivers, wetlands and estuaries, using systematic biodiversity planning techniques incorporating expert review, and to develop an institutional basis for implementing freshwater ecosystem priority areas, through engaging with key stakeholders and through pilot projects in at least two Water Management Areas (Driver *et al.* 2011, Nel *et al.* 2011). The national goal of freshwater conservation policy in South Africa is "*to conserve a sample of the full diversity of* *species and the inland water ecosystems in which they occur, as well as the processes which generate and maintain diversity*" (Hill 2009).

One of the tasks of the NFEPA project is to produce maps of national freshwater ecosystem priority areas for each of the important inland aquatic taxa, including freshwater fish (critical biodiversity areas). These maps have been compiled into an atlas of priority national freshwater ecosystems (Driver *et al.* 2011, Nel *et al.* 2011). In addition, the project developed operational/management guidelines for freshwater ecosystem priority areas and explored legal and institutional mechanisms for implementing freshwater conservation areas (Driver *et al.* 2011, Nel *et al.* 2011).

3.4.3 National Water Act

The National Water Act (No. 36 of 1998) of the Republic of South Africa is widely recognized as the most comprehensive water law in the world (Tewari 2009). The purpose of the Act is to ensure that the water resources of South Africa are protected, used, developed, conserved, managed and controlled in ways that meet the basic needs of present and future generations; promote equitable access to water for all citizens; redress the results of past racial and gender discrimination; promote the efficient, sustainable and beneficial use of water; facilitate social and economic development; provide for growing demand for water use, protect aquatic and associated ecosystems and their biological diversity; reduce and prevent pollution and degradation of water resources; meet international obligations; promote dam safety; and manage floods and droughts (Tewari 2009). The National Government holds authority over the water resources of the country.

One of the most innovative aspects of the New Water Act is the concept of the ecological reserve that takes the basic human needs into account and provides for the needs of aquatic ecosystems (Palmer *et al.* 2002). This provision implies that water releases from dams can be enforced for river maintenance and aquatic ecosystem protection. However, the environmental water requirements for the majority of freshwater and estuarine fishes of South Africa have yet to be determined.

3.4.4 The Yellowfish Working Group

The Yellowfish Working Group is an organization consisting of anglers, conservation officials, members of government, academics, riparian landowners, and industry representatives working together to educate freshwater anglers and riparian landowners in aquatic biology, taxonomy and conservation (Mincher 2007, Skelton and Bills 2008). Their mission is to use yellowfish (*Labeobarbus* spp.) as flagships for the conservation of aquatic environments in South Africa (Mincher 2007). The Yellowfish Working Group and their programmes in yellowfish and riverine conservation are examples of what can be done in freshwater conservation (Skelton and Bills 2008). The group is active through newsletters, annual conferences and interactions with scientific programmes (Mincher 2007, Skelton and Bills 2008).

3.5 Freshwater Fish Conservation in the Cape Floristic Region

Although the negative impact of non-native fish on native fish was recognised in the late 1930s (Barnard 1943), policies regarding freshwater fish conservation did not change until the 1980s, when the conservation authority started taking responsibility for the management and conservation of native fish populations (Coke 1988). The first action was to declare open season on all non-native fishes (Coke 1988), provoking a strong response from organised angling, especially trout-orientated fly anglers As a compromise, the open season was revoked and a moratorium on the stocking of non-native fish instituted (Coke 1988). The CAPE Project provided direction to the conservation of native fishes and culminated in the publishing of the first "State of Biodiversity" report for freshwater fishes in 2002 (Impson *et al.* 2002b).

3.5.1 Freshwater Component of the CAPE Project

The freshwater component of the CAPE Project recognized that continued research on the diversity and conservation needs of freshwater ecosystems in the region, and integration of freshwater and terrestrial components in conservation planning, would be necessary in order to conserve freshwater biodiversity in the Cape Floristic Region (van Nieuwenhuizen *et al.* 1999). Biodiversity research projects of the taxonomy of the genera *Pseudobarbus*, *Sandelia* and *Galaxias* were initiated following the CAPE Project and a conservation plan proposed for the native freshwater fishes of the Cape Floristic Region (Impson *et al.* 2002a). The conservation plan includes:

- Developing a network of conservation areas that include key aquatic systems for native fish conservation,
- Addressing the capacity deficiencies within the nature conservation authority (CapeNature) for the conservation of native freshwater fishes,

- Release of funding to allow urgent research projects, detailed field surveys and related management actions to proceed,
- River rehabilitation projects – specifically the permanent eradication of non-native fish species, especially in protected areas (e.g. CAPE Alien Fish Eradication Project),
- Establishment of freshwater protected areas, and
- Environmental education and public awareness exercises to raise the profile of native freshwater fish in the region.

Although the conservation plan exists, none of the components of the plan have been implemented. Many of the components are vague, lack measurable outcomes, and do not apportion responsibilities for implementation. The result has been the continued spread of high-impact non-native species, such as bass and sharptooth catfish, and the decline of native species. It is clear that the current conservation plan is not being implemented and, without measurable outcomes or apportioned responsibilities, is flawed in its conception. The only component that has been initiated is a pilot river rehabilitation project evaluating the use of piscicides to eradicate non-native fish from selected rivers, the CAPE Alien Fish Eradication Project (Impson 2007).

3.5.2 CAPE Alien Fish Eradication Project

Section 28 of South Africa's National Environmental Management Act (NEMA) obliges conservation authorities to remedy degradation of environments harmed by non-native and/or invasive species. The CAPE Alien Fish Eradication Project, funded by the Global Environmental Facility (GEF) and managed by CapeNature, aims to evaluate non-native fish eradications and the subsequent recovery of natural ecosystems containing native fish. The project was separated into two phases (Impson 2007):

- Phase 1 – identify priority rivers for the eradication of non-native fish, and complete an Environmental Impact Assessment (EIA) on the most suitable rivers, and
- Phase 2 – execute the intervention if deemed to be effective and appropriate.

Four rivers having the highest likelihood of success were selected for the pilot project: the Krom, Rondegat and Twee Rivers in the Cederberg, and the Krom River (Eastern Cape). The four rivers were selected on the basis that (a) the native species were critically endangered and non-native fish were a major threat to their survival, (b) the eradication of non-native fish was feasible, (c) the rivers would provide healthy habitats for the native fish if the non-native species were removed, and (d) the rivers were not of major importance to anglers.

An Environmental Impact Assessment (EIA) was commissioned to independently evaluate the proposed pilot project. The EIA concluded that the justification for the project and the choice of rivers was sound, endorsing the project as vital for the survival of endangered fishes (Enviro-Fish Africa 2009). The use of a piscicide containing rotenone was recommended. The Rondegat River was recommended as the first river to be treated. If successful, the project will provide the basis for developing protocols to eradicate non-native fish from other rivers in the Cape Floristic Region. While large-scale eradication of non-native fish may not be technically feasible at present, small-scale projects to eradicate non-native fish from proposed native fish sanctuaries, or to expand the ranges of critically endangered native species, can be successfully completed with the current available technologies including piscicides (Clarkson *et al.* 2005).

3.5.3 Current Conservation Initiatives

Following the Cape Project, a number of advances have been made in the conservation of freshwater fishes in the Cape Floristic Region.

Taxonomic Revision

The taxonomy of the freshwater fishes of the region has been reviewed. The genus *Pseudobarbus* has been revised based on genetic and morphological studies (Swartz 2005, Swartz *et al.* 2007, 2009). The *Galaxias zebratus* species complex is currently being reviewed with more than ten species having been identified to date (McDowall 2006). Taxonomic evaluations of the *Sandelia capensis* and *Barbus anoplus* species complexes are also in progress.

Ecological Research

Several studies on the impacts of non-native fishes on native fish species and lower trophic groups have been completed (Woodford 2005, Lowe *et al.* 2008, Shelton *et al.* 2008). Studies of the breeding ecology of sawfin *Barbus serra* and Clanwilliam yellowfish *Labeobarbus capensis* (Paxton and King 2009), the rock catlets *Austroglanis barnardi* and *A. gilli* (Bills 1999), and the Twee River redfin *Barbus erubescens* (Marriott 1998) of the Cederberg have been completed. Several surveys of fish distributions in the Olifants-Doring catchment (Bills 1999, Paxton and King 2009), and distribution surveys of the Twee River (Impson *et al.* 2007, Marr *et al.* 2009) have been conducted. Studies of the fish assemblages of the Breede River and the impact of rainbow trout (*Oncorhynchus mykiss*) on the native fishes of the Breede River are currently underway. The knowledge of fish distributions is also improving through the River Health Project and associated fish surveys.

Biodiversity Management Plans

CapeNature has committed to preparing biodiversity management plans for all critically endangered species. Biodiversity management plans are currently being prepared for two freshwater fish species in the Cape Floristic Region: the Clanwilliam sandfish *Labeo seeberi* and the Tradou redfin *Pseudobarbus burchelli* Smith, 1841. Both projects are in the data collection phase.

Native and Non-native Fish Policies

CapeNature ha revised policies for the utilisation of native and non-native fish to reflect the current conservation priorities in the province. Restrictions on the translocation of native species and stocking of non-native fish have been incorporated in the revised policies. The policies are currently in use for screening applications for permits to introduce non-native fish. The policy for the utilization of native fish species has been released for comment and is near finalization. The non-native fish policy has not, as yet, been released for comment.

Yellowfish Working Group

A Western Cape chapter of the Yellowfish Working Group has been established. The group is active in the conservation of the three species targeted by anglers (whitefish, sawfin and Clanwilliam yellowfish) although their focus is orientated towards establishing angling resources rather than implementing conservation actions for the respective species.

Thee River Bass Eradication Initiative

Spotted bass invasion of the Thee River was first recorded in 2007. This tributary of the Olifants River held one of the remaining uninvaded native fish assemblages of the Olifants-Doring catchment. No action was taken when the bass were first reported and they have now established a recruiting population. The native fish population in the lower reaches declined dramatically as the bass population expanded up the river. In 2010, a private individual, who had been lobbying CapeNature to take action since 2007, lost patience and, after garnering support from the landowner, paid for the installation of a gabion weir at a point above the upper limit of bass invasion. Following the installation of the weir, regular visits through the 2010/2011 summer by volunteers removed all adult bass and the majority of the young-of-year cohort from below the weir. The eradication teams used seine nets and spearfishing, and herded bass into gill nets. The Groot Winterhoek Aquatic Corridor Stewardship manager (contracted to CapeNature) actively maintained the removal effort. It is expected that the small number of young-of-year bass that remained will be removed in the summer of 2011/2012. This project illustrates that there is scope for the general public to be involved in non-native fish eradication projects and that not all such projects require the use of piscicides.

Figures

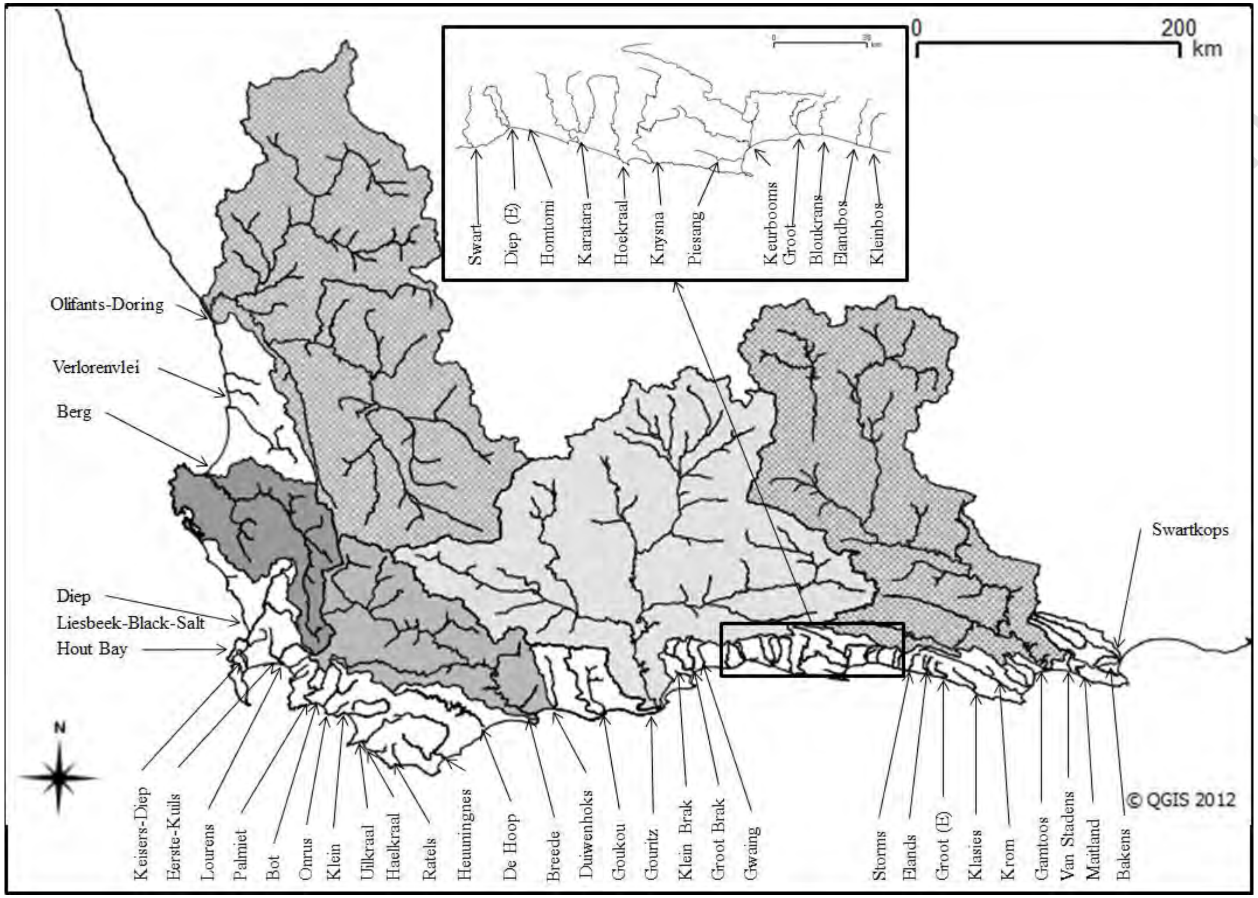


Figure 3.1: Rivers of the Cape Floristic Region, South Africa. Shaded areas represent the five major catchments: Olifants-Doring, Berg, Breede, Gouritz and Gamtoos.

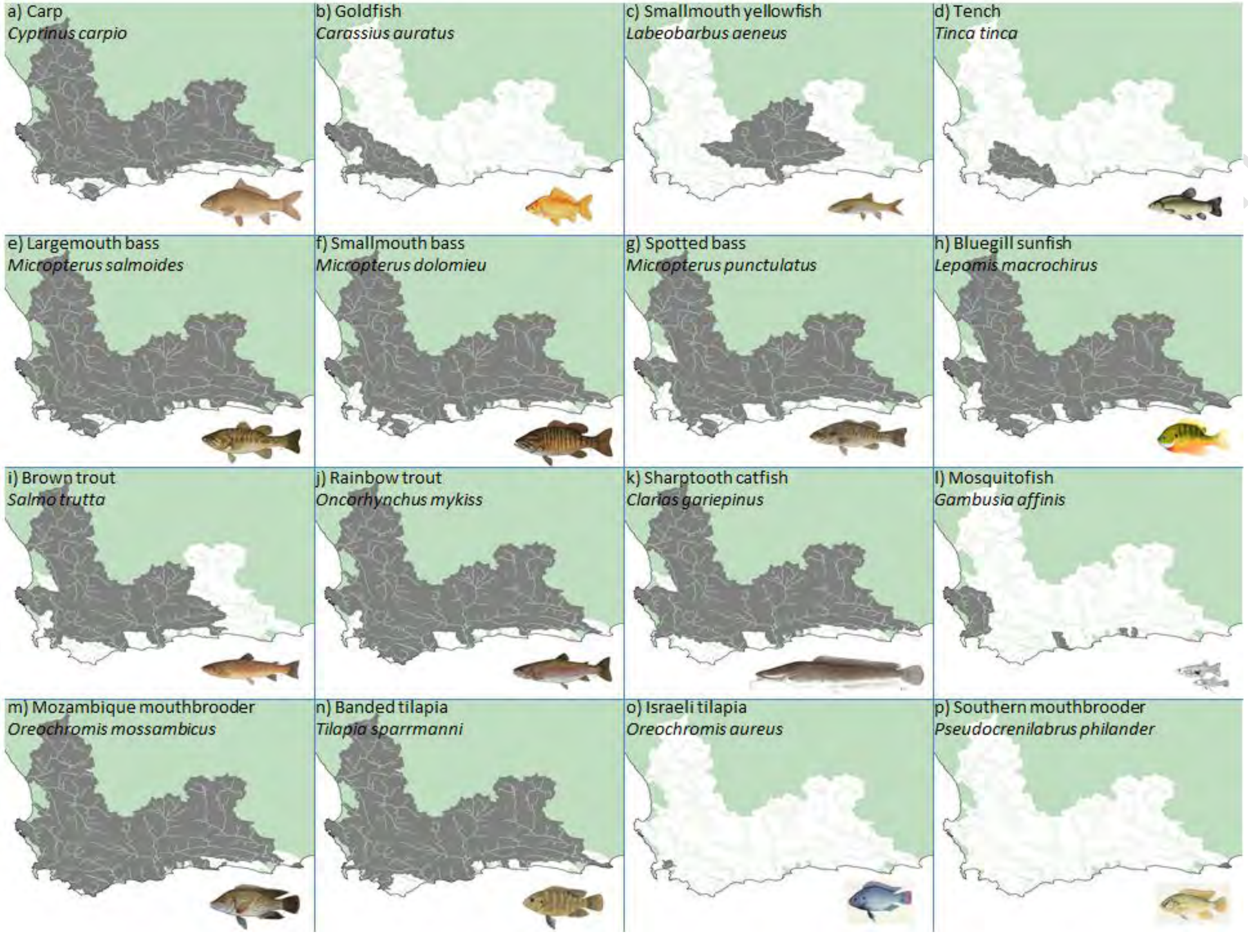


Figure 3.2: Catchments holding populations of various non-native fish in the Cape Floristic Region by species: a) carp; b) goldfish; c) smallmouth yellowfish; d) tench; e) largemouth bass; f) smallmouth bass; g) spotted bass; h) bluegill sunfish; i) brown trout; j) rainbow trout; k) sharptooth catfish; l) western mosquitofish; m) Mozambique mouthbrooder; n) banded tilapia; o) Israeli tilapia; and p) southern mouthbrooder.

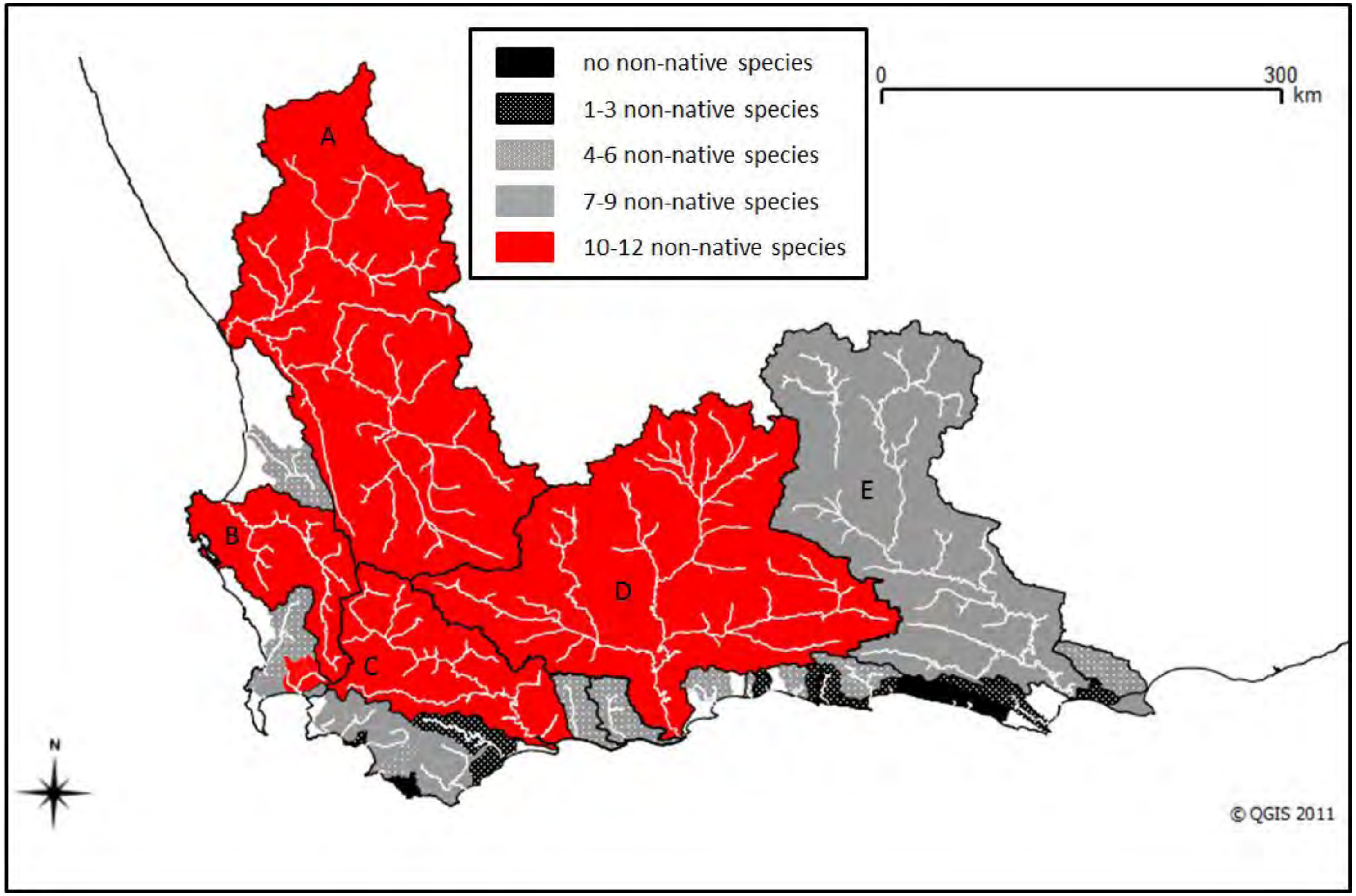


Figure 3.3: Number of non-native fish species present in the catchment of the Cape Floristic Region. Major catchments are A) Olifants-Doring, B) Berg, C) Breede, D) Gouritz, and E) Gamtoos

Table 3.1: IUCN Conservation status of the currently recognised native primary freshwater fish taxa of the Cape Floristic Region (Tweddle *et al.* 2009)

Species		Distribution	Endemic to Cape Floristic Region	Conservation status
Scientific Name	Common Name			2007
Family: Cyprinidae				
Pseudobarbus sp. "burchelli Breede"	Breede River redfin	Breede River System	√	NT
Pseudobarbus burchelli Smith, 1841	Tradou redfin	Tradou River of Breede system	√	CR
Pseudobarbus sp. "burchelli Heuningness"	Heuningnes redfin	Heuningness River System	√	CR
Pseudobarbus burgi (Boulenger, 1911)	Berg River redfin	Berg River System	√	EN
Pseudobarbus sp. "burgi Verlorenvlei"	Verlorenvlei redfin	Verlorenvlei River System	√	CR
Pseudobarbus phlegethon (Barnard, 1938)	Fiery redfin	Olifants River System	√	EN
Pseudobarbus sp. "phlegethon Doring"	Doring River redfin	Olifants River System	√	CR
Pseudobarbus afer (Peters, 1864)	Mandela redfin	Swartkops and Sundays rivers	√	EN
Pseudobarbus sp. "afer Forest"	Forest redfin	Coastal rivers of southern Cape	√	NT
Pseudobarbus sp." afer Gamtoos"	St Francis Redfin	St Francis Bay systems	√	EN
Pseudobarbus sp. "afer Krom"	Krom Redfin	Krom River (Eastern Cape)	√	CR
Pseudobarbus asper (Barnard, 1938)	Small-scale redfin	Gouritz and Gamtoos River System	√	EN
Pseudobarbus tenuis (Barnard, 1938)	Slender redfin	Gouritz River System	√	NT
Pseudobarbus sp. "tenuis Keurbooms"	Keurbooms redfin	Keurbooms River System	√	EN
Barbus anoplus Weber 1897 +	Chubbyhead barb	Gouritz, Gamtoos, Olifants & Orange River Systems		DD
Barbus pallidus A. Smith 1841	Goldie barb	Krom River to Great Fish River (Eastern Cape)		LC
Barbus calidus Barnard, 1938	Clanwilliam redfin	Olifants River System	√	VU
Barbus erubescens Skelton, 1974	Twee River redfin	Olifants River System	√	CR
Barbus andrewi Barnard, 1937	Whitefish	Berg & Breede River systems	√	EN
Barbus serra Peters, 1864	Sawfin	Olifants River System	√	EN
Labeobarbus capensis (A. Smith, 1841)	Clanwilliam yellowfish	Olifants River System	√	VU
Labeo umbratus (A. Smith 1841)	Moggel	Gouritz, Orange, Sundays systems		LC
Labeo seeberi (Gilchrist & Thompson, 1911)	Clanwilliam sandfish	Olifants River System	√	EN
Family: Austroglanididae				
Austroglanis barnardi (Skelton, 1981)	Barnards rock catlet	Olifants River System	√	EN
Austroglanis gilli (Barnard, 1934)	Clanwilliam rock catlet	Olifants River System	√	VU
Family: Galaxiidae				
Galaxias zebratus Castelnau, 1861+	Cape galaxias	Widespread in Cape Floristic Region (Streams and wetlands)	√	DD
Family: Anabantidae				
Sandelia capensis (Cuvier, 1831)+	Cape kurper	Widespread in Cape Floristic Region (Streams and wetlands)	√	DD

Key: IUCN Conservation Status - CR = Critically Endangered, EN = Endangered, VU = Vulnerable, NT = Near Threatened, LC = Least Concern, DD = Data Deficient. + indicates a putative species complex

Table 3.2: Summary of freshwater fish introductions to the Cape Floristic Region (de Moor and Bruton 1988, Skelton 2001)

Year	Aquaculture	Ornamental	Angling	Bio-control	Fodder fish
1700s	Carp (*Cyprinus carpio*)	Carp (*Cyprinus carpio*) Goldfish (*Carassius auratus*)	Carp (*Cyprinus carpio*)		
1890s			Brown trout (*Salmo trutta*) Rainbow trout (*Oncorhynchus mykiss*) Atlantic salmon (*Salmo salar*) [1]		
1910			Tench (*Tinca tinca*)		
1912				Guppy (*Poecilia reticulata*)[1]	
1915	Israeli tilapia (*Oreochromis aureus*)		Eurasian Perch (*Perca fluvialitis*)[3]		
1928			Largemouth bass (*Micropterus salmoides*)		
1936	Mozambique mouthbrooder (*O. mossambicus*)			Mosquitofish (*Gambusia affinis*)	
1937			Smallmouth bass (*Micropterus dolomieu*)		
1938					Bluegill (*Lepomis macrochirus*)
1939			Spotted bass (*Micropterus punctulatis*)		
1941					Banded tilapia (*Tilapia sparrmanii*)
1950			Brook char (*Salvelinus fontinalis*) [1]		
1953			Smallmouth yellowfish (*Labeobarbus aeneus*)		
1959	Nile tilapia (*Oreochromis niloticus*) [2] Mango tilapia (*Sarotherodon galilaeus*)[1]			Red-bellied tilapia (*Tilapia zilli*)[+]	
1980	Sharptooth catfish (*Clarias gariepinus*)		Sharptooth catfish (*Clarias gariepinus*)	Grass carp (*Ctenopharyngodon idella*)[4]	Southern mouthbrooder (*Pseudocrenilabrus philander*)
Established	3	2	9	1	3
Failed	3	0	2	2	0
Extirpated	0	0	1	0	0

Note - 1 failed introduction (de Moor and Bruton 1988, Skelton 2001)

2 Nile tilapia was recorded as being established on the Cape Flats near Cape Town, but no populations have been recently recorded for this species and it is believed that this species did not established in the Cape Floristic Region, even though climate models predict that this species should be able establish in the region (Zambrano *et al.* 2006). The aquaculture industry is currently pushing for permission for hybrid *O. mossambicus* x *O. niliticus* to be introduced into the Cape Floristic Region.

3 Eurasian perch successfully established in two systems in the Cape Floristic Region but has since been extirpated from both systems. One system was treated with rotenone and drained.

4 Grass carp has been introduced into water bodies in the Cape Floristic Region to control aquatic vegetation. Only sterile triploid fish from certified hatcheries are permitted to be introduced. The species has established in the Orange-Vaal and Pongola systems even though they were expected not to establish in these systems. The species is long lived and could survive in natural water courses if they escape from the farm dams.

Table 3.3: Relative risks associated with known vectors and pathways for the introduction and/or spread of non-native freshwater fishes in the Cape Floristic Region

		Pathways						
		Release		Escape	Contaminant	Corridors	Stowaway	Unaided
		Legal	Illegal					
Vectors	Conservation Authorities	Low			Low			
	Recreational Anglers	Low	High		Medium			
	Subsistence Anglers		High		High			
	Bio-control	Low	High	Low	Low			
	Aquaculture Industry	Low	High	Low	Low			
	Ornamental Fish Industry		Medium	Medium	Medium			
	Inter-basin Transfer Schemes					High		
	International shipping						Low	
	Neighbouring Provinces							Low

CHAPTER 4. NON-NATIVE FISH IN MEDITERRANEAN-CLIMATE REGIONS

In the previous chapter the freshwater fish of the Cape Floristic Region were discussed with particular reference to current conservation status, threats, current conservation measures, and non-native fish management initiatives. In this chapter non-native fish introductions in various Mediterranean-climate regions are reviewed at a macro-ecological scale to determine whether lessons for regional-level non-native fish management programmes can be gleaned from macro-ecological studies. Biotic homogenization, taxonomic and functional, is the key macro-ecological concept investigated here. The study has been confined to a single climatic type to reduce the confounding effect of comparing data across multiple climatic types. Note that in this chapter, the geographical south-western Cape has been used for the primary study region in preference to the Cape Floristic Region, because of the ecoregional nature of the chapter.

4.1 Introduction

Large-scale comparative investigations provide an opportunity to explore questions related to the patterns and drivers of the present-day biogeography of freshwater fishes (Leprieur *et al.* 2008a, Blanchet *et al.* 2009). A comparison of introductions across climatically-similar regions promotes an understanding of invasion processes by isolating large-scale drivers other than regional climate (Pauchard *et al.* 2004), and focuses on the role of human activities (Jiménez *et al.* 2008) and the characteristics of the invading species (Moyle and Marchetti 2006). Comparative studies also provide valuable information for the conservation of native species, and the management of non-native species, by identifying potentially harmful species, introduction vectors, and dispersal pathways, providing the scientific information necessary to evaluate taxonomic and functional changes and develop guidelines for sustainable ecosystem management (Pauchard *et al.* 2004).

Europe has a long history of non-native fish introductions, dating back to the Roman Empire (Holcík 1991, Bianco 1998, Bianco and Ketmaier 2001, Copp *et al.* 2005a). The culture of fish in monasteries and parishes during the medieval period further contributed to the spread of freshwater fish (Holcík 1991). During the Renaissance, the nobility engaged in the culture of freshwater fish resulting in species from other continents being introduced into Europe (Holcík 1991). This led to the formations of "Acclimation Societies" in many countries during the 19th century which provided incentives for the establishment (acclimation) of non-native species (plants and animals) (Keith and Allardi 1998). The stocking of native and non-native fish was encouraged to compensate for the decline in native fish stocks and to add "diversity" to the local fish assemblages. During this period, numerous species from North America were introduced into European waters (Holcík 1991, Keith and Allardi 1998). During the 1960s and 1970s, numerous non-native fish from a variety of sources were introduced across Europe for the biological control of aquatic plants and mosquitoes, aquaculture, and to create more diverse recreational fisheries (Cowx 1997). Subsequently, government-sanctioned introductions has ceased in many countries such as Spain and Portugal (Elvira and Almodóvar 2001). Subsequent introductions in these countries have however, continued via illegal releases by anglers and aquarists, or accidental releases from aquaculture facilities (Elvira and Almodóvar 2001).

The freshwater ecosystems of northern Mediterranean Basin are highly modified and under increasing pressure from water development schemes (Economidis 1995, Collares-Pereira *et al.* 2000, Shumka *et al.* 2010b). The major threats to native fish populations are habitat degradation, dam construction and canalisation, water abstraction, alterations to hydrological regimes, pollution, overfishing, and the introduction of non-native species (Elvira 1990, Balik 1995, Bianco 1995, Changeux and Pont 1995, Crivelli 1995, Crivelli and Maitland 1995, Economidis 1995, Elvira 1995a, b, Moyle 1995, Economidis *et al.* 2000, Economidis 2002, Morgan *et al.* 2003, Bobori and Economidis 2006, Innal and Erk'akan 2006, Lorenzoni *et al.* 2006, Shumka *et al.* 2010b). Non-native species have been introduced into the northern Mediterranean Basin for a variety of reasons: the "improvement" of wild stocks, as ornamental species, or for biological control of mosquitoes or aquatic weeds, establishment of fisheries, recreational angling, and aquaculture (Elvira and Almodóvar 2001). Fish stocking is, however, central to the management of freshwater fish populations in many European countries. For example, in Turkey fish stockings are funded by the Turkish government, hydro-electric power companies, universities, and anglers or fisheries organizations (Innal and Erk'akan 2006). In Italy, non-native fish have been introduced so frequently and native fish translocated so extensively that the native composition and taxonomic status of the fish fauna is the subject of debate (Bianco 1998). Many catchments in peninsular Italy have been repeatedly stocked with '*white fish*', a mixture of unidentified species from northern Italy or Eastern Europe (Bianco 1990, 1995, 1998, Bianco and Ketmaier 2001).

Many non-native species have been spread rapidly through their introduced ranges (Elvira 1998, Economou *et al.* 2007). The rapid spread of gibel carp *Carassius gibelio* (Bloch, 1782) and the topmouth gudgeon *Pseudorasbora parva* (Temminck & Schlegel, 1846) has been traced to the widespread stocking of non-native fish throughout Europe (Copp *et al.* 2005a, Gozlan *et al.* 2010a). Both species are common contaminants in "Chinese carp" consignments commonly stocked for aquaculture or aquatic plant control (Copp *et al.* 2005a, Gozlan *et al.* 2010a). Additional species have colonised Mediterranean countries, such as France, through shipping canals connecting formerly independent river systems (Keith and Allardi 1997, 1998), or via trans-national rivers such as the rivers entering Greece and Turkey from Bulgaria (Zenetos *et al.* 2009). Non-native species have resulted in the decline of the native species by introducing pathogens and parasites, loss of genetic identity, high levels of hybridization, and local extirpation or decline of native populations through predation by, or competition with, non-native species (Bianco 1990, 1995, Cowx 1997, Keith and Allardi 1997, 1998, Bianco and Ketmaier 2001, Lorenzoni *et al.* 2006). Despite some positive influences (contribution to fishery production, aquaculture development, mosquito control and reduction of heavy algal blooms), fish introductions have strongly negative ecological, socio-economic and genetic impacts, particularly in regard to fish population interactions, environmental degradation, habitat modification, new diseases and deterioration of fishing opportunities (Cowx and Bergot 1997, Zenetos *et al.* 2009). Commercial and recreational angling are of major economic and political importance (Economou *et al.* 2007, Gozlan 2008, Cowx and Van Anrooy 2010) and the risk of further introductions remains extremely high due to increasing interest in amateur angling, low public awareness about non-native fish impacts, and poor mechanisms to enforce bans on non-native fish introductions (Gozlan 2008, Zenetos *et al.* 2009, Gozlan *et al.* 2010b).

In Mediterranean-climate regions outside of the Mediterranean Basin, the introduction of non-native fish was strongly associated with European colonisation and the advent of suitable transport vectors. Tribal communities in California had utilised the native fish-resources in a seemingly sustainable manner prior to European settlement (Moyle 1995). Large-scale European settlement during the mid-19th century gold rush resulted in large-scale degradation of river systems as a result of hydraulic mining and large-scale water projects for agriculture (Moyle 1995). By the time the first non-native species were introduced, the river systems had been degraded through unrestricted mining, logging, and wetland conversion. Further, many of the native fish populations, specifically Pacific salmon stocks, were in decline as a result of habitat modification and unregulated commercial fishing (Moyle 1976, 1995, Moyle and Marchetti 2006, Moyle *et al.* 2011). Non-native species were introduced for recreational angling, commercial fisheries, as forage or bait fish, for biological control of mosquitoes and aquatic weeds, or as ornamental species (Moyle 1976). In many cases, the non-native species were the only species able to survive in the degraded river systems (Moyle 1976, 1995, Moyle *et al.* 2011). In addition, the translocation of native fish between isolated catchments for recreational angling and native species conservation compounded the problem (Moyle 1976). Today, most of California's waterways are dominated by non-native fishes (Moyle 1976, 1995, Moyle *et al.* 2011) and the native fish populations continue to decline (Moyle *et al.* 2011).

The freshwater fish of Mediterranean-climate regions in the southern hemisphere are mostly small-bodied and poorly adapted to the presence of non-native fish (Morgan *et al.* 2002, Skelton 2002, Morgan *et al.* 2004, Penaluna *et al.* 2009, Tweddle *et al.* 2009, Young *et al.* 2010). In Chile, the creation of recreational fisheries was the primary reason for fish introductions prior to the 1980s (Basulto 2003), while government-sponsored aquaculture has been the major driving force for fish introductions since the 1980s (Iriarte *et al.* 2005). Chile is currently one of the world's largest producer of cultured salmonids (Arismendi *et al.* 2009), accounting for more than 73% of Chile's aquaculture production (Buschmann *et al.* 2009). In southern Chile, non-native brown trout *Salmo trutta* Linnaeus, 1758 and rainbow trout *Oncorhynchus mykiss* (Walbaum, 1792), account for more than 60% of total fish abundance and more than 80% of total biomass, while native fish are absent from 40% of the streams (Soto *et al.* 2006). Recently, the rapid colonization of Chinook salmon *Oncorhynchus tshawytscha* (Walbaum, 1792) in South America has raised concerns regarding the impact of escapees from salmonids culture facilities on the native fish assemblages in Chile (Soto *et al.* 2001, Soto *et al.* 2007, Correa and Gross 2008, Arismendi *et al.* 2009, Garcia De Leaniz *et al.* 2010, Young *et al.* 2010, Ibarra *et al.* 2011). The introduction of non-native fish has coincided with the degradation of the river systems making it difficult to clearly delineate the relative importance of the respective factors in the decline of the native species (Habit *et al.* 2010). Furthermore, native species are not afforded any legal protection, while introduced trout are protected under Chilean law (Penaluna *et al.* 2009).

In the former British colonies of South-western Australia and the south-western Cape (South Africa), the initial introductions of non-native fish were government-sanctioned introductions of ornamental common carp *Cyprinus carpio* Linnaeus, 1758 and goldfish *Carassius auratus* (Linnaeus, 1758), followed by the prized angling species brown and rainbow trout (de Moor and Bruton 1988, Morgan *et al.* 2004). Further government-sanctioned introductions of non-native species were for additional angling species and the control of mosquitoes (de Moor and Bruton 1988, Morgan *et al.* 2004). Eurasian perch *Perca fluviatilis* Linnaeus, 1758 was introduced for angling in South-western Australia, whereas a suite of North American and South African species were introduced for angling, or as fodder fish for angling species, in the south-western Cape (See Section 3.3 for further details on the introduction of freshwater fish to the south-western Cape). Legal and illegal introductions of non-native fish for mosquito control continue in both ecoregions, as does the illegal release of angling and ornamental species (Impson *et al.* 2002b, Morgan *et al.* 2004, Impson 2007). Many of the native species of the south-western Cape are restricted to sanctuaries above barriers, while the main stems and tributaries below these barriers are dominated by non-native species (Skelton 1983, 2000, Impson *et al.* 2002b, Skelton 2002, Impson 2007, Tweddle *et al.* 2009). As with other Mediterranean-climate regions, the rivers in South-western Australia and the south-western Cape are subject to high levels of water abstraction, habitat degradation, eutrophication, salinization, fragmentation, pollution and the presence of non-native fishes (Impson *et al.* 2002b, Morgan *et al.* 2003, Impson 2007).

Freshwater fish are the most frequently introduced aquatic animal group (Gozlan 2008). As with plants and birds (Lockwood 1999, McKinney and Lockwood 1999), phylogenetic history and human affiliation have been identified as predictors favouring species of freshwater fish selected for introduction (Alcaraz *et al.* 2005, Marr *et al.* 2010). Certain families and taxa have more non-native species because of strong human biases towards introducing species, such as game fish, forage fish, and bio-control agents for aquatic weeds or mosquitoes (Kolar and Lodge 2002, Ruesink 2003, Ruesink 2005, Clavero and García-Berthou 2006, García-Berthou 2007, Rahel 2007, Marr *et al.* 2010).

Biotic homogenization describes the phenomenon by which range-limited endemic species are replaced by widespread species, ultimately reducing spatial biodiversity (McKinney and Lockwood 1999) [see Section 2.5]. Several authors have emphasized the importance of identifying and understanding present-day patterns of biotic homogenization with the intention of establishing conservation goals aimed at reducing potential future ecological impacts (Lockwood and McKinney 2001, Olden *et al.* 2004, Olden 2006, Rooney *et al.* 2007, Olden *et al.* 2010). Although a number of biotic homogenization studies of freshwater fish assemblages have been completed (Radomski and Goeman 1995, McKinney and Lockwood 1999, Rahel 2000, Duncan and Lockwood 2001, Marchetti *et al.* 2001, Rahel 2002, Marchetti *et al.* 2004a, Taylor 2004, Clavero and García-Berthou 2006, Eberle and Channell 2006, Marchetti *et al.* 2006, Rahel 2007, Leprieur *et al.* 2008b, Olden *et al.* 2008, Taylor 2010), the majority have focussed on taxonomic homogenization in temperate latitudes of the northern hemisphere (García-Berthou 2007). Comparative studies between different regions, or over multiple scales, are lacking (García-Berthou 2007). Functional changes in biotic communities resulting from taxonomic homogenization have been raised as a concern (Clavel *et al.* 2010).

To date, functional homogenization has only been evaluated for bird communities in the U.S.A., France, and the Iberian Peninsula (Devictor *et al.* 2007, Devictor *et al.* 2008, Clavero *et al.* 2009, Clavero and Brotons 2010, Baiser and Lockwood 2011). This chapter documents freshwater fish introductions across Mediterranean-climate regions to determine biases in the taxonomic selection and geographic origin of non-native fishes. Biotic homogenization, taxonomic and functional, is explored to evaluate the convergence of regional faunas across a single climatic region and to identify species and functional traits contributing to the observed convergence.

4.2 Methods

This chapter focuses on freshwater bony fish (Osteichthyes), excluding marine species that occasionally enter freshwaters in Mediterranean-climate regions. Non-native species are defined as species that did not historically occur in the area, but have subsequently established self-sustaining populations as a result of human activities. The "historical" species assemblage for each region was reconstructed from literature, whereas the "present" species assemblage was based on the latest surveys, taking into account recorded introductions and extirpations.

4.2.1 Study Region

The northern Mediterranean Basin and the Mediterranean-climate ecoregions of California, central Chile, south-western Australia and the south-western Cape, South Africa are included in this study (Figure 1.2). Ecoregions, delineated *sensu* Abell *et al*. (2008), include Western Iberia, Southern Iberia, Eastern Iberia, Cantabrian Coast and Languedoc (in part), Peninsular Italy, Gulf of Venice (Po Drainage), Dalmatian Coast (Eastern Adriatic), Southern Adriatic Drainages (Albania), Ionian Sea Drainages, Aegean Sea Drainages, Vardar, Thrace, Western Anatolia, Southern Anatolia, Central Anatolia, and Northern Anatolia ecoregions for the northern Mediterranean Basin, Oregon and Northern California Coast (in part) and Sacramento-San Joaquin ecoregions for California (USA), the South Andean Pacific Slope ecoregion for Chile, the south-western Australian ecoregion, and the Cape Fold ecoregion for the south-western Cape, South Africa.

4.2.2 Sources of Data

A database of catchment-level freshwater fish presence-absence records was assembled for 333 catchments within Mediterranean-climate ecoregions: the Iberian Peninsula (35), France

(20), the eastern Adriatic Coast (17), Greece (90), Turkey (53), California (32), Chile (7), south-western Australia (33), and the south-western Cape (48). A catchment is defined as a drainage basin including all the tributaries of the main river. The catchment level data were compiled from distributional data made available by collaborators working in the respective regions and supplemented from literature (Table 4.1).

Presence-absence data for the historical and present-day fish distributions were compiled for each of the Mediterranean-climate ecoregions. A taxonomic revision of the freshwater fishes of the south-western Cape is currently in progress. The native fish for this region were limited to the taxa listed by Tweddle *et al.* (2009). For the northern Mediterranean Basin, species were defined according to Kottelat and Freyhof (2007).

Functional trait data for fish species were compiled from FishBase (Froese and Pauly 2010) to evaluate functional homogenization of the freshwater fish assemblages. Due to the lack of data for all the 620 species in the database, a rudimentary set of seven functional traits was selected: adult trophic level, size of native range, level of parental care, population doubling time, maximum adult size, physiological tolerance, and extent of migration. See Table 4.2 for explanation of each variable.

The database compiled for this study is presented in Appendix 4: catchments by ecoregion (Appendix A.4.1); species introduced into Mediterranean-climate regions detailing taxonomic authority, order, family, and geographical origin (Appendix A.4.2); species presence-absence by catchment (Appendix A.4.3); and species functional trait data (Appendix A.4.4). The assistance of Julian Olden, Michael Marchetti, Emili García-Berthou, Ivan Arismendi, David Morgan, Fabien Leprieur, Annamaria Nocita, Sergio Zerunian, Predrag Semonic, Marko Ćaleta, Radek Šanda, Spase Shumka, and Serhan Tarkan in compiling the catchment-level database used in this study is acknowledged.

4.2.3 Statistical Methods

Ecoregion-level species lists were compiled from the catchment-level distributions. The ecoregion- and catchment-level data were used in different analyses. Ecoregion level data were used to evaluate biases in patterns of taxonomic and geographical origin of the species introduced. Both data sets were used to evaluate changes in compositional similarity (ΔCS), i.e. biotic homogenization, of the regional freshwater fish assemblages, both taxonomic and functional.

Biases in taxonomic selection and area of origin

Taxonomic biases in non-native fish introductions were evaluated at the ordinal and familial levels following Alcaraz *et al.* (2005). The global number of freshwater species in each order, and family, was compiled from Nelson (2006), Berra (2007), and Eschmeyer and Fricke (2011) [Appendix A.4.5]. The total number of species introduced to each ecoregion from each order, and family, was compared to the expected value using a binomial distribution, based on a random sample of the global species pool. The probability of the observed number of species being introduced into an ecoregion, given the number of species introduced into that ecoregion and the global number of extant species in the order/family, is presented as an *R* value (Lockwood 1999). These *R* values were corrected to control the false discovery rate using the procedure of Benjamini and Hochberg (1995), as implemented in the *R* package (R Development Core Team 2011).

Variations in the geographical origins of non-native species were determined by allocating the native range of the non-native species to one of seven regions: Africa, Eastern Asia (including Australia), Central Eurasia, Northern Europe, Eastern North America, Pacific and Polar North America, and South America. The independence of region of origin and introduction was tested using a chi-square test of independence (Sokal and Rohlf 1995) in the *R* package, with a Monte Carlo simulation to determine the level of significance of the result.

Functional and Taxonomic Homogenization

A functional trait matrix was constructed using each level of each functional trait as a separate variable. The regional presence-absence matrix for each time period (historical and present-day) was multiplied by the functional trait matrix using the matrix outer product routine, as implemented in the *R* package. The species presence-absence matrix was thus converted into a matrix containing the number of occurrences of each functional trait level per region/catchment. Bray-Curtis similarity was used to evaluate the functional similarity between the regions/catchments whereas the Jaccard's index of similarity (Jaccard 1900) was used for the taxonomic component because they are the most commonly used indices for these types of studies (Clarke and Warwick 2001, Olden and Rooney 2006).

Compositional similarity of historical assemblages (CS_H) and present-day assemblages (CS_P), both taxonomic and functional, were calculated separately using the "VEGAN" package (Oksanen *et al.* 2010) in the *R* package. Taxonomic and functional homogenization were quantified as the change in CS index (ΔCS) for each pair of regions between the present-day and historical CS resemblance matrices (calculated as $\Delta CS = CS_P - CS_H$) (Olden and Rooney

2006). The catchment CS_H and ΔCS were calculated as the average of the pair-wise catchment CS_H and ΔCS values (Marchetti *et al.* 2006). In addition, the compositional similarity between the historical and present-day fish assemblages was calculated for each region to confirm the level of change in compositional similarity. Non-metric multi-dimensional scaling (NMDS) was performed to summarise multivariate patterns in taxonomic and functional homogenization using the PRIMER-E 6.1.5 statistical software package (Clarke and Warwick 2001). The PERMANOVA and PERMDISP routines in the PRIMER-E 6.1.5 statistical software package (Anderson 2001b, Anderson *et al.* 2008) were used to ascertain whether the changes in compositional similarity were statistically significant. The PERMDISP routine was used to determine whether the multivariate dispersion about the group centroid differed between the two time periods while the PERMANOVA routine was used to determine whether the position of the group centroids in multivariate space and/or multivariate dispersion about the group centroid differed between the two time periods (Anderson 2001b, Anderson *et al.* 2008). A significant result from the PERMDISP test reduces the interpretability of significant result from a PERMANOVA tests because it is not possible to determine whether the significant result in PERMANOVA is due to multivariate dispersion, change in the position of the group centroids, or both (Anderson 2001b, Anderson *et al.* 2008). PERMANOVA tests were therefore only performed when the PERMDISP test returned a non-significant result. A SIMPER analysis (Clarke and Warwick 2001) was performed in PRIMER-E 6.1.5 to identify the species and functional traits that contributed most to the observed changes in taxonomic and functional compositional similarities between the historical and present-day assemblages.

4.3 Results

A total of 136 species from 26 families in 13 orders have been successfully introduced into the Mediterranean-climate regions included in this study. The Mediterranean Basin received 91 species from 21 families in 10 orders, while the other Mediterranean-climate regions received 66 species from 19 families in 10 orders. More species were introduced into California (44) and Peninsular Italy (39) than the number of species in the historical native assemblage (Figure 4.1; Table 4.3). More than 20 species have been introduced into nine regions, while only Western, Southern and Central Anatolia and South-western Australia received 10 species or fewer. A number of species have been translocated within the regions (Figure 4.1; Table 4.3), the highest in California (12 species) and the Cantabrian coast Languedoc (10). No translocations were recorded for the Aegean Sea drainages, Western, Southern, Central and Northern Anatolia, Chile and South-western Australia. No catchment-level data were available for Peninsular Italy and the Gulf of Venice drainages. Ten regions recorded extirpations (Figure 4.1; Table 4.3), the highest in Central Anatolia (8 species) followed by California (3), Cantabrian coast Languedoc and Southern Iberia (2), and Western Iberia, Eastern Iberia, Dalmatian Coast, South East Adriatic, Western Anatolia and Northern Anatolia (1). The loss of *Gila crassicauda* (Baird & Girard 1854) and *Pogonichthys ciscoides* Hopkirk 1974 from California and *Alburnus akili* Battalgil, 1942 and *Pseudophoxinus handlirschi* (Pietschmann, 1933) from Central Anatolia are the only known global extirpations.

4.3.1 Biases in Taxonomic Selection and Area of Origin

Of the 136 species introduced into the Mediterranean-climate regions, 121 are from five orders: the Cypriniformes (52), Cyprinidontiformes (8), Perciformes (33), Salmoniformes (18) and Siluriformes (10) (Figure 4.2). Thirteen orders of freshwater fish have been introduced into one or more of the Mediterranean-climate ecoregions (Table 4.4). Three orders (Cypriniformes, Cyprinodontiformes, and Salmoniformes) have been introduced to all 20 ecoregions, Perciformes to 19 ecoregions and Siluriformes to 14 ecoregions. Salmoniformes, Acipenseriformes, Cypriniformes, Polydontiformes and Perciformes were over-represented (have more species introduced than predicted by a random selection from the global species pool) amongst species introductions when all regions are considered together (Table 4.4) ($P < 0.001$), whereas Characiformes and Siluriformes were under-represented (have fewer species introduced than predicted by a random selection from the global species pool) ($P < 0.001$). For the northern Mediterranean Basin, Salmoniformes, Acipenseriformes, Cypriniformes, Polydontiformes and Perciformes were over-represented amongst introductions, whereas Siluriformes were under-represented (Table 4.4). For the other mediterreanean-climate regions, Salmoniformes, Acipenseriformes, and Perciformes were over-represented amongst introductions whereas Characiformes and Siluriformes were under-represented (Table 4.4). Cypriniformes were over-represented in northern Mediterranean Basin ecoregions west of the Aegean Sea, Salmoniformes in almost all ecoregions, and Perciformes in California and the south-western Cape (Table 4.4). The set of taxonomic orders introduced to each ecoregion was non-random and did not vary significantly between ecoregions ($\chi^2 = 506.93$, Monte Carlo $P = 0.217$), but did vary significantly between ecoregions in the northern Mediterranean Basin and those in the other Mediterranean-climate ecoregions ($\chi^2 = 46.08$, Monte Carlo $P < 0.001$).

Three families (Cyprinidae, Poeciliidae, and Salmonidae) have been introduced to all 20 ecoregions, while Centrarchidae have been introduced to 15 ecoregions and Ictaluridae to 12 ecoregions. Salmonidae, Acipenseridae, Cyprinidae, Polydontidae, Ictaluridae, Loricariidae, Moronidae, Centrarchidae, Cichlidae and Gobiidae were over-represented amongst introductions when all ecoregions were considered together (Table 4.5) ($P < 0.001$), whereas Characidae were under-represented ($P < 0.001$). For the northern Mediterranean Basin, Salmonidae, Acipenseridae, Cyprinidae, Polydontidae, Ictaluridae and Cichlidae were over-represented amongst introductions (Table 4.5). For other Mediterranean-climate regions, Salmonidae, Acipenseridae, Ictaluridae and Centrarchidae were over-represented amongst introductions (Table 4.5). Cyprinidae were over-represented in northern Mediterranean basin regions west of the Aegean Sea, Salmonidae in almost all regions, with the exceptions of Anatolia, South-western Australia and the south-western Cape. Centrarchidae were over-represented in northern Mediterranean Basin regions west of the Ionian Sea, California and the south-western Cape (Table 4.5). The set of taxonomic families introduced to each region was non-random and did not vary significantly between ecoregions ($\chi^2 = 203.58$, Monte Carlo $P = 0.776$), but did vary significantly between ecoregions in the northern Mediterranean Basin and those in the other Mediterranean-climate ecoregions ($\chi^2 = 31.30$, Monte Carlo $P < 0.001$).

Species successfully introduced to the Mediterranean-climate regions originated from seven areas of origin: Africa, Eastern Asia, Central Eurasia, Northern Europe, Eastern North America, Pacific and Polar North America, and South America (Figure 4.4). Each ecoregion received species from Eastern Asia, Central Eurasia, Northern Europe, Eastern North America, and Pacific and Polar North America, with the remainder of the species originating from either Africa or South America. All ecoregions received species from six areas of origins, with the exception of Peninsular Italy, which received species from all seven areas of origins. The northern Mediterranean basin received more species from Northern Europe and Central Eurasia, whereas the other regions received predominantly from Eastern North America, a result biased by the Californian introductions (Figure 4.3). Each ecoregion was found to have received species from a unique composition of geographic origins that varied significantly among ecoregions ($\chi^2 = 233.7$, Monte Carlo $P < 0.001$) and between ecoregions in the northern Mediterranean Basin and those in the other Mediterranean-climate ecoregions ($\chi^2 = 51.90$, Monte Carlo $P < 0.001$).

The native freshwater fish assemblages of the northern Mediterranean Basin and the south-western Cape are dominated by Cypriniformes (family Cyprinidae), making up more than 60% of the native fish assemblage. Salmoniformes (Salmonidae) and Perciformes (Blenniidae and Gobiidae) are the next greatest contributors to the native fish assemblages of the northern Mediterranean Basin, with the remainder of the orders and families constituting less than 8%

each. For California, Cypriniformes (Cyprinidae and Catostomidea) make-up about 40% of the native fish assemblages, whereas Salmoniformes (Salmonidae) and Scorpaeniformes (Cottidae) each contributing about 20%. In Chile, Atheriniformes (Atheriniopsidae), Osemeriformes (Galaxiidae) and Siluriformes (Trichomcteridae, Nematogenydae and Diplomystidae) each order contributing 26% of the native fish assemblage. In South-western Australia, Osmeriformes (Galaxiidae and Lepidogalaxiidae) make up 60% of the native fish assemblage.

The species introduced into the northern Mediterranean Basin are also dominated by Cypriniformes (Cyprinidae), but to a lesser degree, with only 50% of the non-native fish assemblage coming from this order. Salmoniformes (Salmonidae) and Perciformes (Centrarchidae and Percidae) constitute the next largest components of the non-native assemblages, followed by Cyprinidontiformes (Poeciliidae). In California, Perciformes principally from the family Centrarchidae constitute more than 40% of the non-native fish assemblage, followed by Cypriniformes (Cyprinidae), Siluriformes (Ictaluridae), Salmoniformes (Salmonidae) and Cyprinodontiformes (Poeciliidae). In Chile, the non-native fish assemblage is dominated by Salmoniformes (Salmonidae) (about 30%), followed by Cypriniformes (Cyprinidae), Siluriformes (Ictaluridae) and Cyprinodontiformes (Poeciliidae). In South-western Australia, the non-native fish assemblage contains Cyprinifoprmes (Cyprinidae) - 30%), Perciformes (Cichlidae, Percidae and Tetrapontidae - 30%), Salmoniformes (Salmonidae - 20%) and Cyprinodontiformes (Poeciliidae - 20%). The non-native fish assemblage in the south-western Cape is made up of Perciformes (Centrarchidae and Cichlidae - 50%), Cypriniformes (Cyprinidae - 25%) and Salmoniformes (Salmonidae - 12.5%).

4.3.2 Regional-Level Biotic Homogenization

The catchment-level database was aggregated to an ecoregional database to facilitate the regional-level analyses. The historical taxonomic similarity among native freshwater fish faunas of the northern Mediterranean Basin was calculated to be 12.2% whereas that for the other Mediterranean-climate ecoregions was 0.46%, with only one species, *Galaxias maculatus* (Jenyns 1842), historically shared between South-western Australia and Chile. The fish faunas of the Mediterranean-climate ecoregions have homogenized over time, with the mean compositional similarity increasing between 3.5 and 10.6% (mean 7.7%, Table 4.3). Taxonomic homogenization was highest in Mediterranean Basin ecoregions west of the Adriatic Sea (Table 4.3). NMDS showed a strong overall tendency toward more similar fish faunas (Figure 4.4). Although present-day regional fish assemblages remain more similar to their historical assemblages than to those of other regions, it is apparent that regions have become considerably more similar. The PERMDISP analysis of homogeneity of multivariate dispersion confirmed that there was a significant change in the multivariate dispersion about the historical and present-day group centroids (p = 0.027).

Two species, *Oncorhynchus mykiss* (Walbaum, 1792) and *Cyprinus carpio* Linnaeus, 1758, are now present in all the ecoregions considered here. Both are, however, native to Mediterranean-climate ecoregions: *Oncorhynchus mykiss* to California and *Cyprinus carpio* to six ecoregions in central Europe. A further two species, *Carassius auratus* (Linnaeus, 1758) and *Gambusia holbrooki* Girard, 1859, have been introduced into 18 Mediterranean-climate ecoregions. A further seven species *Lepomis gibbosus* (Linnaeus, 1758), *Pseudorasbora parva* (Temminck & Schlegel, 1846), *Sander leucoperca* (Linnaeus, 1758), *Ameiurus melas* (Rafinesque, 1820), *Micropterus salmoides* (Lacepède, 1802), and *Salvelinus fontinalis* (Mitchill, 1814) and *Carassias gibelio* (Bloch, 1782) have been introduced to ten or more ecoregions. Twenty species from the families Acipenseridae (1), Atherinopsidae (1), Cyprinidae (4), Poeciliidae (3), Salmonidae (3), Centrarchidae (2), Ichtaluridae (3), Clariidae (1), Percidae (1), and Cichlidae (1) have been introduced to both the Mediterranean Basin and the other Mediterranean-climate ecoregions. The Atlantic sturgeon *Acipenser sturio* Linnaeus, 1758 showed the greatest range contraction, having being extirpated from six of the ecoregions in the Mediterranean Basin.

Not only have the fish assemblages in Mediterranean-climate ecoregions become more similar taxonomically, they have become more similar functionally. The mean functional compositional similarity increased between 2.4 and 19.0% (mean 8.1%, Table 4.3). Functional homogenization was highest in Peninsular Italy (19.0%) with a further six ecoregions (Eastern Iberia, Central and Northern Anatolia, Chile and the south-western Cape) recording a greater than 10% increase in functional similarity (Table 4.3). NMDS showed a strong overall tendency toward more similar fish faunas (Figure 4.5), The PERMDISP analysis found that there was no significant difference in the multivariate dispersion about the historical and present-day group centroids (p = 0.188), however, the PERMANOVA test confirmed that there was a significant difference in the position of the historical and present-day group centroids in multivariate space (p = 0.005).

Eight functional traits contributed to more than 60% of the increase in functional similarity, each increasing in average abundance by more than 5% between the historical and present-day assemblages. The present-day assemblages have more species with native ranges that extend to more than one zoogeographical sub-regions, more non-migratory species, more species with a population doubling time between 1.4 and 4.4 years, more invertivores, more species with no parental care, more species with moderate levels of physiological tolerance and more species with the size ranges of 40–80 cm and 81–160 cm.

4.3.3 Inter-Regional Homogenization

The catchment-level database was analysed in the inter-regional analyses to evaluate all ecoregions simultaneously, i.e. how the ecoregions have changed with respect to all other ecoregions. Inter-regional changes in taxonomic and functional compositional similarity were observed for all regions (Figures 4.6, 4.7, and 4.8). The NMDS ordination plot of taxonomic changes shows that the respective regions have responded in different ways to catchment level species introductions and extirpations (Figure 4.6). On the whole, all catchments have moved towards a central point. Some regions, such as Chile and the western Mediterranean Basin, show a clear separation between historical and present-day fish assemblages, with the present-day assemblages placed closer to the central point, indicating that the fish assemblages in these regions are now more similar to fish assemblages from other regions. For California, the south-western Cape and south-western Australia two patterns are evident: some catchments are now more similar to fish assemblages from other regions (closer to the central point), while others show no change. All regions show inter-regional taxonomic homogenization, with the exception of the south-western Cape and the Aegean Sea drainages, which show differentiation in more than 50% of their catchments (Figure 4.8). Homogenization is highest in Chile and the western Mediterranean Basin. The PERMDISP analysis showed that the overall change in multivariate dispersion of the catchments between the historical and present-day assemblages was significant ($p < 0.05$), but the pair-wise test within ecoregions showed that this was only true for Western Iberia, California, Chile and the south-western Cape. The PERMANOVA analysis showed that the overall change between the historical and present-day assemblages was significant ($p < 0.05$), but the pair-wise test within ecoregions showed that this was true for all ecoregions with the exception of the Mediterranean Basin east of the Ionian Sea. These results show that there has been a change in the multivariate dispersion of the catchments about the group centroid for Western Iberia, California, Chile and the south-western Cape and possibly a shift in the position of group centroid. For ecoregions west of the Aegean Sea, there has been no statistically significant change in the dispersion of the catchments about the group centroid, but a significant change in the position of the group centroid.

The observed change in taxonomic compositional similarity can be attributed to the introduction of 12 species (*Gambusia holbrooki*, *Cyprinus carpio*, *Oncorhynchus mykiss*, *Carassius auratus*, *Micropterus salmoides*, *Lepomis gibbosus*, *Carassius gibelio*, *Salmo trutta*, *Lepomis macrochirus*, *Gambusia affinis*, *Pseudorasbora parva,* and *Oreochromis mossambicus*) to more than 10% of the catchments.

The functional traits of fish assemblages in each region have moved towards a point to the side of the NMDS plot (Figure 4.7), a composition of functional traits that differs from those historically occurring in Mediterranean-climate ecoregions. All ecoregions show inter-regional functional homogenization in more than 50% of their catchments with the exception of the Aegean Sea drainages, which shows differentiation (Figure 4.8). Functional homogenization is highest in Chile. The PERMDISP analysis showed that the overall change in multivariate dispersion of the catchments between the historical and present-day assemblages was not significant ($p = 0.125$). The PERMANOVA analysis showed that the overall change between the historical and present-day assemblages was significant ($p < 0.05$), but the pair-wise test within ecoregions showed that this was true for all ecoregions with the exception of the Mediterranean Basin east of the southern Adriatic drainages. These results show that there has been no statistically significant change in the dispersion of the catchments about the group centroid, but for the Mediterranean Basin ecoregions west of the Ionian Sea, California, Chile, South-western Australia, and the south-western Cape there has been a statistically significant change in the position of the group centroid.

The changes in functional similarity can be attributed to the introduction of species with native ranges that extend to more than one zoogeographical sub-regions, are non-migratory species, have a population doubling time between 1.4 and 4.4 years, are invertivores, and have moderate levels of tolerance. Lake Amik in Southern Anatolia showed the greatest change in functional homogenization as a result of the entire assemblage of this water-body being lost following the draining of the lake from the 1940s to the 1970s.

4.3.4 Intra-Regional Homogenization

The catchment-level database for each ecoregion was analysed separately in the intra-regional analyses to evaluate changes within the ecoregions. Intra-regional changes in taxonomic and functional compositional similarity were observed for all ecoregions included in the study (Figure 4.9). The majority of ecoregions show inter-regional taxonomic differentiation in more than 50% of their catchments, with the exception of Western Iberia, Eastern Iberia, the Dalmatian coast, Western Anatolia and Central Anatolia, which show homogenization in more than 50% of their catchments (Figure 4.9). There is a distinct difference in the level of differentiation between the catchments of the Mediterranean Basin and those of the other Mediterranean-climate regions with the latter regions displaying considerably greater levels of taxonomic differentiation in more than 75% of their catchments. The observed changes in intra-regional taxonomic compositional similarity vary greatly among the ecoregions, with unique sets of species being introduced to, or lost from, each region (Table 4.7). However, a good proportion of these taxonomic changes can be attributed to the introduction of ten species (*Gambusia holbrooki*, *Cyprinus carpio*, *Oncorhynchus mykiss*, *Carassius auratus*, *Lepomis gibbosus*, *Carassius gibelio*, *Micropterus salmoides*, *Salmo trutta*, *Pseudorasbora parva,* and *Sander leucoperca*) to five or more ecoregions.

All regions show inter-regional functional differentiation in more than 50% of their catchments with the exception of Western Iberia, Eastern Iberia, the Dalmatian coast, Vardar, Western Anatolia, Central Anatolia and Chile, which show functional homogenization in more than 50% of their catchments (Figure 4.9). Differentiation is highest in Eastern Iberia, California and the south-western Cape. The observed changes in functional compositional similarity at an intra-regional level vary greatly among the regions as a result of fish species with unique sets of species functional traits being introduced to, or lost from, each region (Figure 4.9). The changes in functional similarity can be attributed to the introduction of species with native ranges that extend to more than one zoogeographical sub-regions, are non-migratory, have a population doubling time between 1.4 and 4.4 years, are invertivores, and have moderate levels of physiological tolerance.

4.3.5 Intra-Regional Homogenization in the south-western Cape

In the south-western Cape, taxonomic homogenization of the catchments has resulted in the natural east-west differences in assemblage composition being compromised. A clear gradient in historical freshwater fish assemblages, from the Olifants-Doring system in the west to the Swarkops in the east, can be seen along the bottom of the NMDS plot (Figure 4.10). Similarly, a clear gradient among the major catchments can be seen for the present-day data. Although, the present-day communities of the minor catchments fall between their historical position along the historical major catchment transect and the present-day major catchment transect. Taxonomic homogenization was only found in two catchments, the Olifants-Doring and the Berg Rivers, with taxonomic differentiation being recorded for the rest of the catchments. Both PERMDISP and PERMANOVA analyses returned significant results ($p < 0.05$) indicating that there has been a change in the multivariate dispersion of the catchments about the group centroid and possibly a shift in the position of group centroid, or both. The introduction of *Oreochromis mossambicus* (67% of catchments), *Micropterus salmoides* (54%), *Tilapia sparrmanii* (46%), *Lepomis macrochirus* (42%), *Cyprinus carpio* (35%), *Micropterus dolomieu* (35%), *Oncorhynchus mykiss* (33%), *Gambusia affinis* (23%), and *Clarias gariepinus* (27%) accounts for almost 45% of the dissimilarity between the historical and present-day assemblages.

Although not as clear as in the taxonomic analysis, gradients can be seen in functional trait composition of historical freshwater fish assemblages, from the Olifants-Doring system in the west to the Swarkops in the east, in the NMDS plot (Figure 4.11). Similarly, a gradient among the major catchments can be seen for the present-day data. Although the present-day communities of the minor catchments fall between their historical position along the historical major catchment transect and the present-day major catchment transect. Functional homogenization was only found in catchments west of the Agulhas Plain, with the exception of the Onrus and Berg Rivers, functional differentiation being recorded for catchments from the Agulhas Plain eastwards. The PERMDISP analysis returned a non significant result ($p = 0.766$) whereas the PERMANOVA analysis returned a significant result ($p < 0.05$) indicating the change in multivariate dispersion was not significant, but that the shift in group centroid was significant. The changes in functional similarity can be attributed to the introduction of species with native ranges that extend to more than one zoogeographical sub-regions, are non-migratory, have a population doubling time between 1.4 and 4.4 years, are invertivores or omnivores, are moderate tolerant to tolerant, are brood guarders and have size ranges of 21-40 cm, 41 – 80 cm and 81 – 160 cm. Changes in the above functional traits accounts for almost 60% of the dissimilarity between the historical and present-day assemblages.

4.4 Discussion

The freshwater fish assemblages of the Mediterranean-climate regions have undergone profound changes, taxonomically and functionally. The introduction of species has resulted in the reduction of the characteristic endemism of the regions (Marr *et al.* 2010) and has increased the number of species in each. The highest number of introduced species was found for California, followed by Peninsular Italy, the Gulf of Venice Drainages and the Dalmatian Coast. The widespread introduction of non-native fishes in Italy is a by-product of that country's lack of control on freshwater fish introductions (Copp *et al.* 2005a). Similarly, California has been recognised as a hotspot for introductions of taxonomic groups such as plants (Jiménez *et al.* 2008). The lowest number of introductions and no translocations were recorded within Anatolian ecoregions.

The results demonstrate that the majority of non-native species in Mediterranean-climate regions come from five taxonomic orders (Cypriniformes, Cyprinodontiformes, Perciformes, Salmoniformes and Siluriformes), as noted in previous studies: California (Moyle and Marchetti 2006), the Iberian Peninsula (Alcaraz *et al.* 2005), south-western Australia (Morgan *et al.* 2004), and five Mediterranean-climate regions (Marr *et al.* 2010). More than 90% of all introductions (121 of 136) were of fish in these orders, confirming the role of phylogenetic preference and human association in freshwater fish introductions as these orders only contain 76% of the global freshwater fish species pool. Kark and Sol (2004) found that six families accounted for more than 78% of bird introductions into Mediterranean-climate regions and that species and families were non-randomly introduced to the respective regions. In the current study, nine families of freshwater fish (Cyprinidae, Salmonidae, Centrarchidae, Cichlidae, Gobiidae, Acipenseridae, Ictaluridae, Poeciliidae, and Percidae) accounted for 81% of the species established in Mediterranean-climate ecoregions. Further, as with birds, the five taxonomic orders and nine families were non-randomly introduced.

Many bird and plant families that have been widely introduced possess specific traits that encourage further introduction, e.g. birds for sport (pheasants: Phasianidae) and pets (parrots: Psittacidae), and plants as ornamentals (roses: Rosaceae) (McKinney and Lockwood 1999, Blackburn *et al.* 2009). Similarly, specific families of freshwater fish have been widely introduced because they have highly desirable traits (Alcaraz *et al.* 2005). Salmoniformes of the family Salmonidae are important recreational angling and aquaculture species and were significantly over-represented in most ecoregions, with the exception of Anatolia. Perciformes contain important recreational angling (e.g. Centrarchidae) and aquaculture (e.g. Cichlidae) species and were over-represented only in California and the south-western Cape. The family Centrarchidae were overrepresented in the Mediterranean Basin West of the Aegean Sea, California, and the south-western Cape, highlighting the importance of recreational angling in these regions. Characiformes and Siluriformes were under-represented when all regions were considered together, but not for any specific ecoregion. However, the Siluriform family Ictaluridae was over-represented in Italy, California and Chile. Siluriformes and Characiformes have large numbers of tropical species that may not be able to establish in Mediterranean-climate regions, where the temperature ranges and harsh abiotic conditions may extend beyond their physiological tolerances. Cypriniformes is a large order which has not been introduced to the same extent as smaller families, such as Salmonidae, perhaps due to the lack of value placed on members of this order by global recreational angling and aquaculture industries. However, Cypriniformes, and their family Cyprinidae, were over-represented in the western portion of the northern Mediterranean Basin. Cyprinodontiformes were not significantly over-represented in any regions, even though the order contains the most widely introduced genus, *Gambusia.*

The results highlight the fact that only in the northern Mediterranean Basin does the make-up of the non-native fish assemblage of a region resemble that of the native fish assemblage. In the Mediterranean-climate regions on other continents, the composition of the non-native fish assemblage contains orders and families not historically present in these regions. This is particularly noticeable in the southern hemisphere: Chile, South-western Australia and the south-western Cape. Historically, Salmoniformes and Cyprinodontiformes were not present in any of these regions, while Cypriniformes were never present in Australia or Chile. In addition, the families Centrarchidae and Cichlidae were never present in any of the southern hemisphere's Mediterranean-climate regions.

Each region received species from a unique set of geographical origins. The diversity of geographical origins poses a challenge to conservation authorities making it difficult to identify potential source regions of species that would successfully establish. A similar result was obtained for plants in central Chile and California (Jiménez *et al.* 2008). The diversity of origins highlights the importance of studies aimed at identifying characteristics of species that have successfully established self-sustaining populations in other Mediterranean-climate ecoregions. Eastern North America was an important source of introductions to most regions, but particularly to California. The northern Mediterranean Basin received species predominantly from Northern Europe or Central Eurasia.

4.4.1 Pathways for Introductions

The pathways for these introduction of species to Mediterranean-climate regions have been legal and illegal stocking, secondary spread of legally and illegally stocked species, contamination of stocks used in the legal and illegal stocking, release of species for the biological control of mosquitoes and aquatic plants, deliberate or accidental release of aquaculture species, and illegal release of unwanted aquarium fish. In addition, some regions have recorded species expanding their ranges through canals connecting previously isolated river systems, species being introduced into one or more countries sharing a river system, and anadromous species expanding their ranges to adjacent river systems.

Stocking is one of the major pathways for freshwater fish to be introduced into new regions. Stocking is an integral part of the European freshwater fisheries culture and it is unlikely that this practice will stop in the foreseeable future. The first response of fishermen, fishery managers and local political leaders has been to stock waters when catch returns begin to fall (Cowx and Gerdeaux 2004), rather than to address the underlying cause of the decline in the fishery, for example over-exploitation, poor water quality, unsustainable water extraction, habitat degradation, or the presence of non-native fishes. While the stocking of water bodies can be expected to continue, there are numerous improvements that can be implemented to reduce the risk of stocking introducing further non-native fish species. It is clear that the Italian stocking practices of indiscriminately introducing any fish they could acquire (Bianco and Ketmaier 2001) are not conducive to controlling the introduction and spread of non-native fish. Clear stocking policies should be established for each water body to justify the need for the stocking, and to outline the contribution of regular stocking to the overall management objectives for the water body. From this, a list of acceptable species and sources can be drawn up to provide guidance for the long-term management of the native and non-native species present in the water body. Further, there is clear evidence that unwanted species have been spread as contaminants in legal stocking consignments e.g. *Pseudorasbora parva* and *Carassius gibelio* (Copp *et al.* 2005a, Gozlan *et al.* 2010a). Contamination can be reduced by increasing the accountability of the hatcheries supplying stocking programmes to provide only specific fish species. Quality control by the hatchery and stocking agency could substantially reduce the risk of spreading undesirable species. Encouraging sustainable, or non-consumptive, utilization of fisheries could reduce the demand for future stocking, especially for naturalised species. Transferring the cost of the stocking to the end user, for example through license fees, may further reduce the demand for stocking programmes. However, transferring the cost to the users may increase the risk of poaching or illegal introduction of non-native species, and may suffer from numerous logistical challenges such as enforcement, collection of license fees etc.

In many countries, the release of species for biological control has stringent testing protocols specifically to ensure host-specificity of the agent before it is released. For freshwater fish used in biological control, these stringent protocols for host-specificity are not applied. If fish are to be used for biological control, they should be required to meet the testing protocols enforced for other organisms. Certainly, adequate risk assessments should be performed prior to planned introductions for biological control, or any other reason, are authorised.

Aquaculture is the fastest growing primary industry globally (Tacon *et al.* 2010). The establishment of new aquaculture facilities in many countries is inevitable. Since aquaculture is a commercially-driven activity, one possible management mechanism could be to engender self-regulation, with the provision that practitioners pay for eradication of new populations of aquaculture species. An alternative could be annual permitting of aquaculture facilities with a penalty of substantial fines or permit revocation should the facility be implicated in the release of species to natural watercourses. Aquaculture facilities should be inspected regularly and limited to techniques that pose a low risk of release to natural environments. In Chile, as in many countries, government regulatory, monitoring and enforcement efforts are compromised by limited financial and technical resources and a shortage of relevant scientific research (Buschmann *et al.* 2009). As a result, the private sector has created different forms of self-regulation for salmonid aquaculture. These efforts appear to be modifying the behaviour of the salmon producers, but an open, multidisciplinary and independent science-based assessment of their ability to control the environmental and social impacts of the industry is required (Buschmann *et al.* 2009). In Europe, a new regulation (Council Regulation 708/2007, of 11 June 2007) establishes guidelines for the use of non-native and locally absent species in aquaculture.

Illegal stocking and secondary spread of angling species and the release of unwanted aquarium fish is nullifying multi-million dollar native fish recovery projects, damaging sustainable recreational fisheries worth billions of dollars, threatening native species with extinction, and diverting conservation resources away from programmes that benefit fishing and aquatic resources and into expensive and often recurring remediation programmes (Johnson *et al.* 2009). These introductions are difficult to plan for, or manage, because of the selfish nature of the people involved in these releases. Education programmes may be effective in reducing the releases by the uninformed people who view their releases as more humane than killing the unwanted pets. Provision of facilities to receive unwanted aquarium pets may reduce the incidence of aquarium fish releases. There will, however, always be a rogue element that will continue to spread specifically angling species and the limited resources available for enforcement are not sufficient to prevent further introduction by such individuals.

The release of non-native species in waterways shared between neighbouring countries, the connection of previously isolated river systems by shipping canals, and the release of marine dispersing anadromous species in adjacent catchments are challenging non-native fish management issues. Management of river systems across political and provincial/regional boundaries requires consensus in management objectives between the respective stakeholders involved.

4.4.2 Taxonomic Homogenization

The results show strong evidence of on-going taxonomic homogenization in the fish faunas of the Mediterranean-climate regions. The regional-level homogenization was similar (about 7%) and appears to be independent of the number of species historically native to the area. Levels of regional homogenization in the northern Mediterranean Basin are highest in the vicinity of Italy and decrease in both east- and west-wards. Levels of regional homogenization in the other Mediterranean-climate regions are constrained to between 5 and 7.5%.

Spatial scale is an important consideration when studying biotic homogenization (Olden 2006). The results are consistent with the prediction that levels of homogenization would be greater the coarser the spatial scale of the study (Olden 2006). As shown by Marchetti *et al.* (2006), the changes in compositional similarity observed using regional level data differed from those calculated using catchment level data. Using regional level data, homogenization was found for all Mediterranean-climate ecoregions. Using catchment level data, inter-regional taxonomic homogenization was observed in more than 50% of the catchments in all Mediterranean-climate ecoregions with the exception of the Aegean Sea and the south-western Cape. At an intra-regional scale, there is a clear separation between the homogenization of the northern Mediterranean Basin and the four other Mediterranean-climate regions. Overall, a greater proportion of catchments recorded differentiation at the intra-regional scale but the magnitude of differentiation in the ecoregions outside the Mediterranean Basin were 2-3 times greater than those in the Mediterranean Basin. This could be as a result of a common set of species being spread through Europe, the longer history of introductions in Europe, or European species being introduced into the non-European regions.

The regional level analysis identified ten species that had been introduced into five or more ecoregions whereas the same analysis using catchment level data identified 12 species that had been introduced into more than 10% of the catchments. Eight species are common to both lists: *Gambusia holbrooki*, *Cyprinus carpio*, *Oncorhynchus mykiss*, *Carassius auratus*, *Lepomis gibbosus*, *Carassius gibelio*, *Micropterus salmoides*, and *Pseudorasbora parva.* It is clear that species widely introduced at the regional-level, such as *Sander leucoperca*, *Ameiurus melas*, and *Salvelinus fontinalis*, are widespread at the catchment-level whereas species widely introduced at the catchment-level, such as *Lepomis macrochirus*, *Gambusia affinis*, *Salmo trutta,* and *Oreochromis mossambicus*, are more likely to be illegally spread within the regions. For the management of non-native species, the catchment-level analysis is possibly more informative because it highlights species which are more likely to be illegally spread after introduction.

4.4.3 Functional Homogenization

All Mediterranean-climate regions showed changes in functional similarity. It is clear that a set of functional trait that characterise the changes in the fish assemblages resulting from non-native fish introductions. The results of the analysis of functional homogenization confirm the general predictions of biotic homogenization: specialist species with limited ranges are being replaced by wide-spread generalist species. The present-day assemblages have more species with native ranges that extend to more than one zoogeographical sub-regions, more non-migratory species, more species with a population doubling time between 1.4 and 4.4 years, more invertivores, more species with no parental care, more species with moderate levels of physiological tolerance and more species with the size ranges of 40 – 80 cm and 81 – 160 cm. These shifts in the functional composition may have many subtle impacts on the recipient systems. The increase in non-migratory species may result in reduced input of marine derived nutrients into the freshwater system. The increase in larger bodied, and longer lived, species may result in the increased hold-up of nutrients in the freshwater system reducing freshwater derived nutrients to estuaries and inshore marine systems. The naturally low nutrient status of rivers in Mediterranean-climate regions may result in the larger bodied non-native species being in a poorer condition, and possibly at a lower biomass due to nutrient limitations. Species with wider native distributions may be more tolerant and adaptable in the recipient system possessing the capacity to survive the seasonal river water quality extremes natural in Mediterranean-climates. These extremes may be buffered by the hydrological control of seasonal flow in these regions due to damming and flood controls.

The selection of functional traits to represent ecosystem changes requires knowledge of the organism's interactions with their environment, their community, and how these traits vary across environmental gradients (Petchey and Gaston 2006). For this study, seven functional trait characteristics were chosen to represent functional changes in energy and nutrient flow in the system and the specialization a species contributes to the assemblage. Detailed data were not available for all 620 species included in this study, therefore a rudimentary set of functional traits (adult trophic level, size of native range, level of parental care, population doubling time, maximum adult size, physiological tolerance, and extent of migration) were selected to represent the majority of energy-flow and specialization traits. This leads to the question: *Were the selected functional traits sufficient to summarise the function of fish species in freshwater ecosystems?* A set of functional traits should include a measure of the fishes position in the food web (trophic level), the level of energy/nutrient accumulation/turn-over (longevity or body size), and the transfer of energy/nutrients to other systems (migration) to summarise the species role in energy and nutrient cycling. Secondly, a measure of the species specialization in habitat preferences (flow, depth, substrate, turbidity), distribution (size of native range), sensitivity to change (physiological tolerance, salinity, temperature, oxygen levels), feeding style *sensu* Matthews (1998), and reproductive strategy (parental care, reproductive guild *sensu* Balon (1975, 1981)) is required to summarize the uniqueness of the function that the species fulfils within the community. The set of functional traits selected include all of these aspects, with the exception of habitat specialization. This information was not consistently available in FishBase and this variable was therefore excluded. Whether the functional traits selected were suitably comprehensive will only become evident once further studies of functional homogenization in freshwater fish assemblages are published.

4.4.4 Limitations of Macroecological Homogenization Studies

While the results of this study are very informative, there are a number of limitations in this study. Firstly, the nature of the scale and the data used in the study over-accounts for the impact of non-native species and does not fully account for other impacts such as habitat degradation. Using catchment scale data does not communicate the full extent of the impact non-native species and habitat degradation where historically widespread species are now restricted to one or two populations above barriers preventing non-native species invasion. The immiscibility between native and non-native fish has been recorded for the Colorado River, USA (Marsh and Pacey 2005) and is evident in the south-western Cape (Marr *et al.* in press), south-western Australia (Morgan *et al.* 2004), and Chile (Penaluna *et al.* 2009). There is a need for studies at finer scales, such as regional scale Marchetti *et al.* (2001), Marchetti *et al.* (2006), or catchment scale Scott and Helfman (2001), Scott (2006), and Hermoso *et al.* (2011b), to fully explore the catchment-level drivers of homogenization. Collection of data for the 333 catchments in this study would be a challenging assignment, even for a rudimentary set of variables. The scale of the study, to a large degree, precludes a more detailed analysis of habitat drivers of biotic homogenization. The use of catchment level data further reduces full analysis of the impacts of non-native species introductions, although it does allow for the inclusion of regional translocation, and catchment level extirpation, of native species.

4.4.5 Lessons for the south-western Cape

The majority of the species widely introduced in Mediterranean-climate regions have already been introduced into the south-western Cape, the exceptions being *Gambusia holbrooki*, *Lepomis gibbosus*, *Carassius gibelio*, and *Pseudorasbora parva*. It is unlikely the either

Gambusia holbrooki, or *Lepomis gibbosus* would be introduced into the south-western Cape because the closely related *Gambusia affinis* and *Lepomis macrochirus* have already established in the region and are recognised as invasive species. Although both *Carassius gibelio* and *Pseudorasbora parva* are currently included on the prohibited list of non-native species in South Africa, both have been widely introduced in Europe as contaminants in legal stockings (Copp *et al.* 2005a, Gozlan *et al.* 2010a). It is possible that both species could already be present in the region, having entered as contaminants in ornamental fish consignments. The illegal release of unwanted ornamental fish would result in these species being released into urban watercourses. If these species are present in the region, their distribution is likely to be limited to high density urban areas and their spread is likely to be limited.

Grass carp *Ctenophyrangodon idella* have successfully established recruiting populations in some Mediterranean-climate ecoregions. This species is currently available in the south-western Cape as triploids for the control of aquatic vegetation. Grass carp have invaded the Orange-Vaal system and viable individuals could be illegally introduced to the south-western Cape, especially by farmers who do not want to pay for triploid fish.

Ornamental species, such as *Australocherus facetum*, *Fundulus heteroclitus*, *Misgurnus anguillicaudatus*, *Poecillia reticulata*, and *Puntius conchonius*, have established recruiting populations in Mediterranean-climate region with the possibility that many more have remained undetected. Climate change predictions of increases in temperatures for the majority of the Mediterranean-climate regions will increase the probability of establishment for unwanted aquarium species. Several aquaculture species such as *Odontesthes bonariensis*, *Oreochromis* species, and Ictaluridae species have established in other Mediterranean-climate regions. Ictalurid species are prohibited in South Africa.

The changes in the taxonomic composition of the freshwater fish assemblages in the south-western Cape have resulted in changes at lower trophic levels (Lowe *et al.* 2008). These changes have not been linked to the functional changes resulting from the introduction of non-native species. This is an important research area which will increase our understanding of the changes resulting from biotic homogenization. Further, studies exploring the fine-scale drivers of biotic homogenization, *sensu* Marchetti *et al.* (2001), Marchetti *et al.* (2006), Scott (2006), and Hermoso *et al.* (2011b), would contribute towards fully documenting the impact of biotic homogenization in the south-western Cape.

Figures

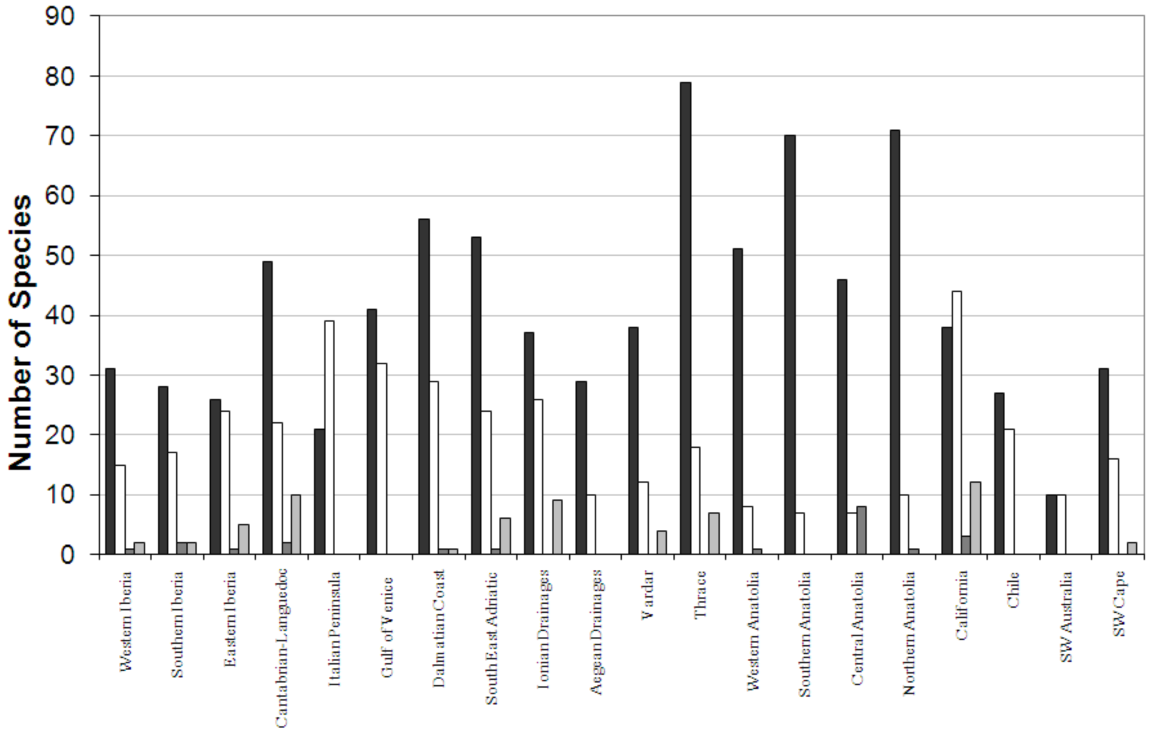


Figure 4.1: Number of freshwater fish species native (solid bars), successfully introduced (open bars), extirpated (dark grey bars) and translocated (light grey bars) for the northern Mediterranean Basin, California, central Chile, south-western Australia and the south-western Cape.

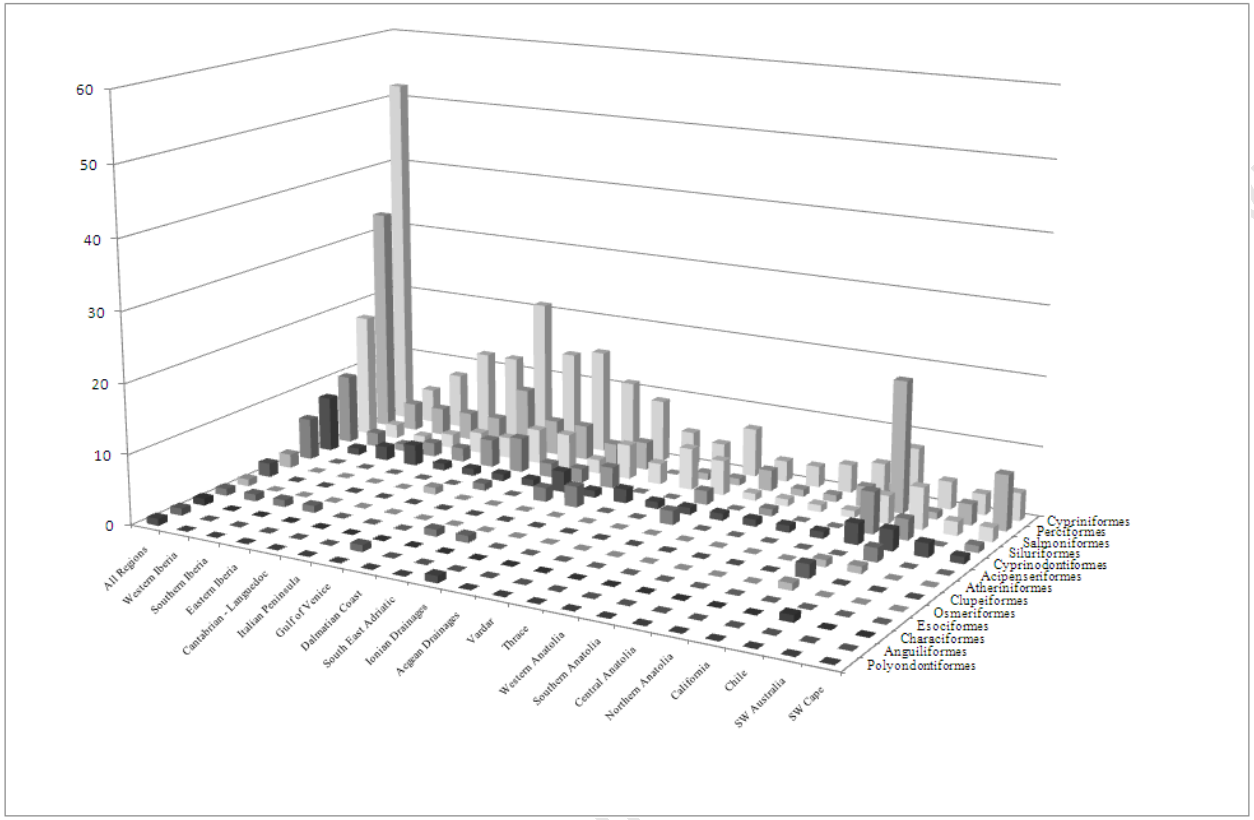


Figure 4.2: Number of freshwater fishes introduced to the northern Mediterranean Basin, California, central Chile, south-western Australia, and the south-western Cape by taxonomic order.

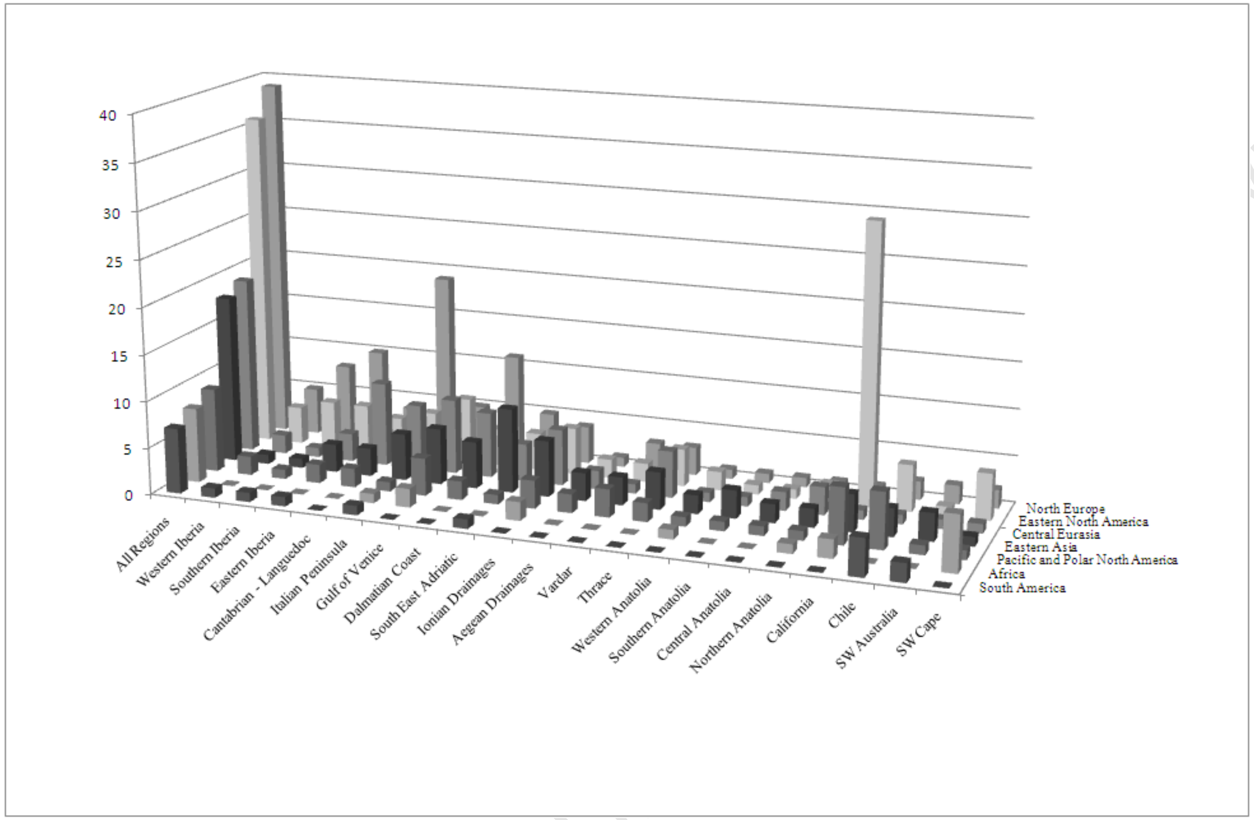
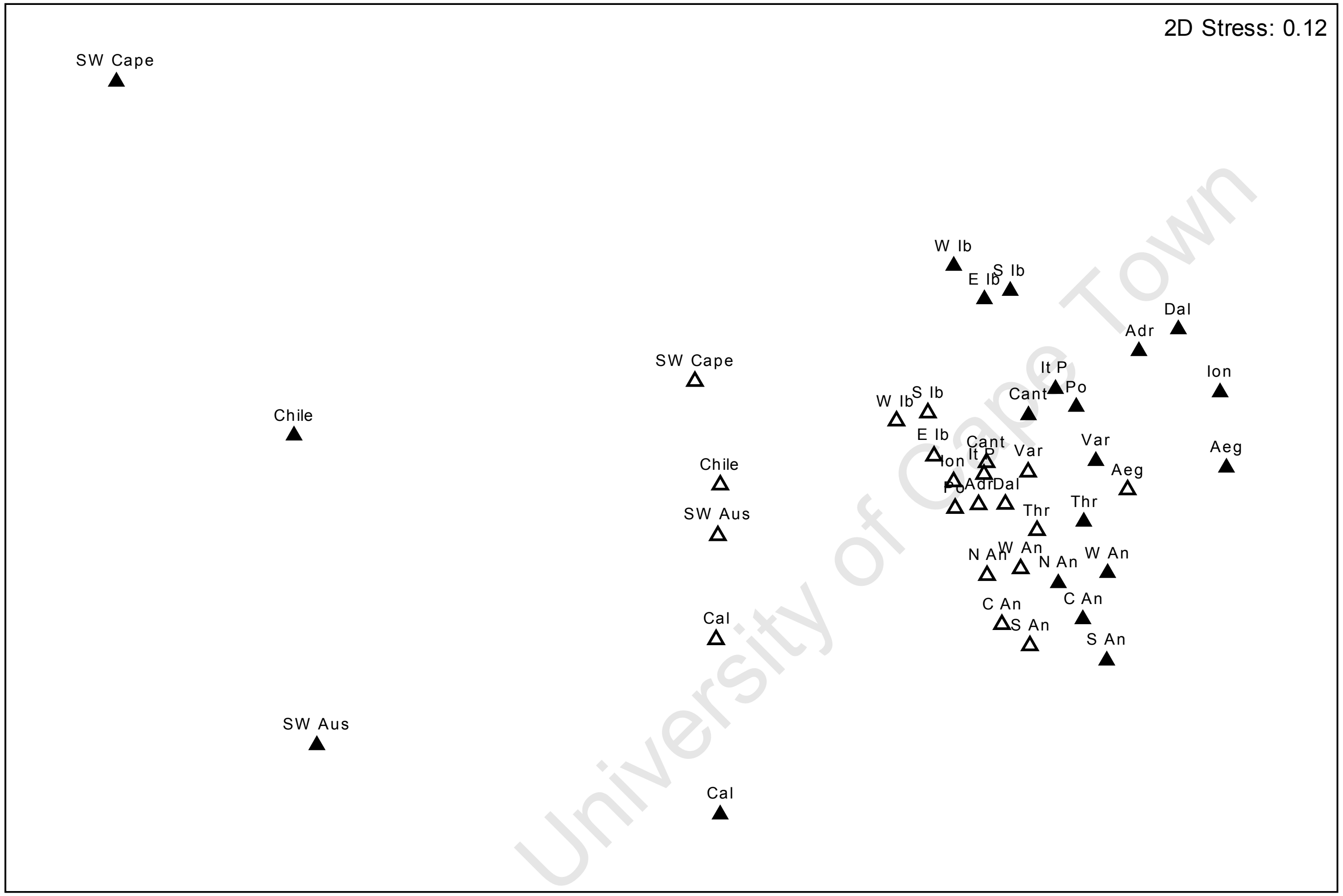
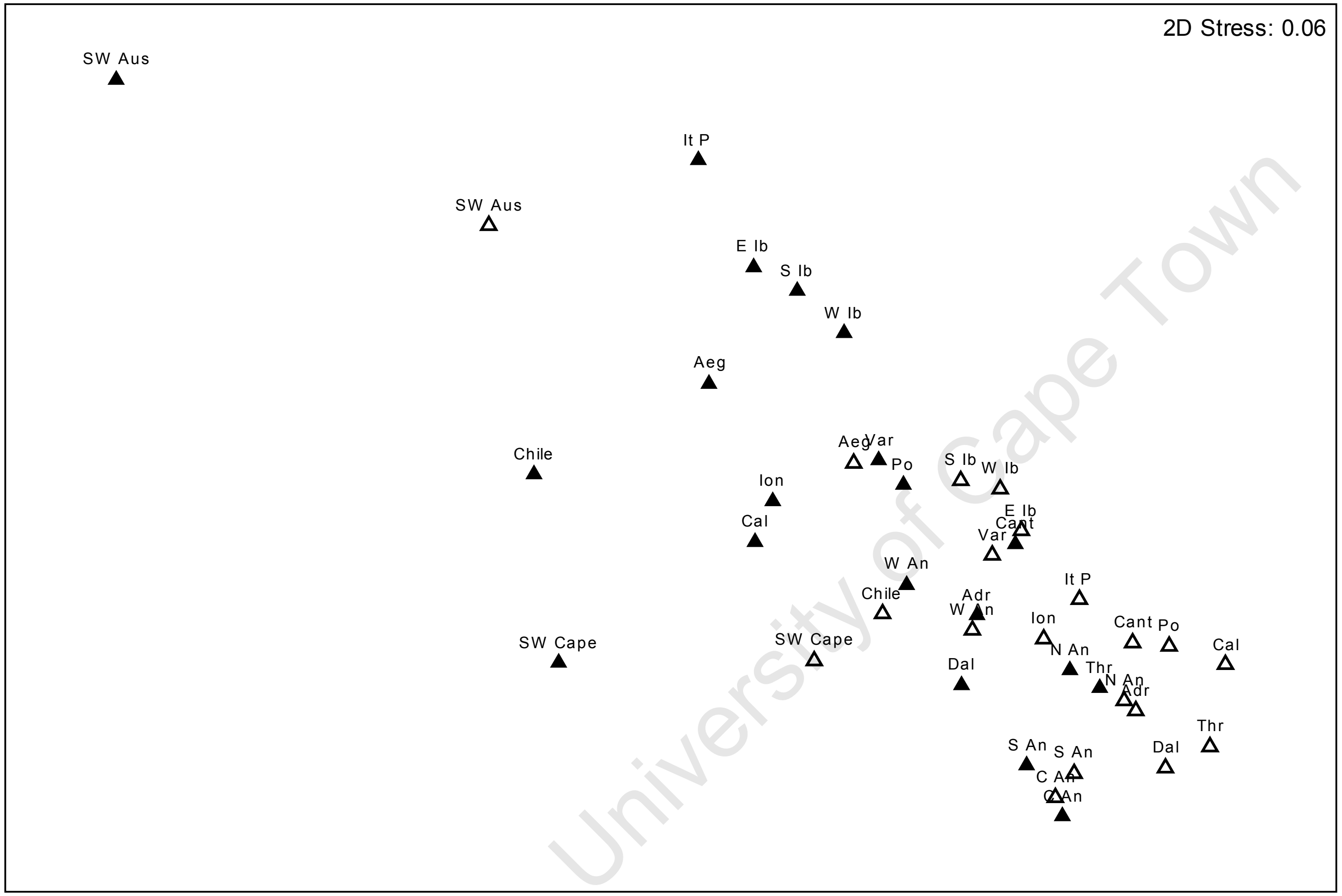


Figure 4.3: Number of freshwater fishes introduced to the Northern Mediterranean Basin, California, central Chile, south-western Australia, and the south-western Cape by geographic origins.



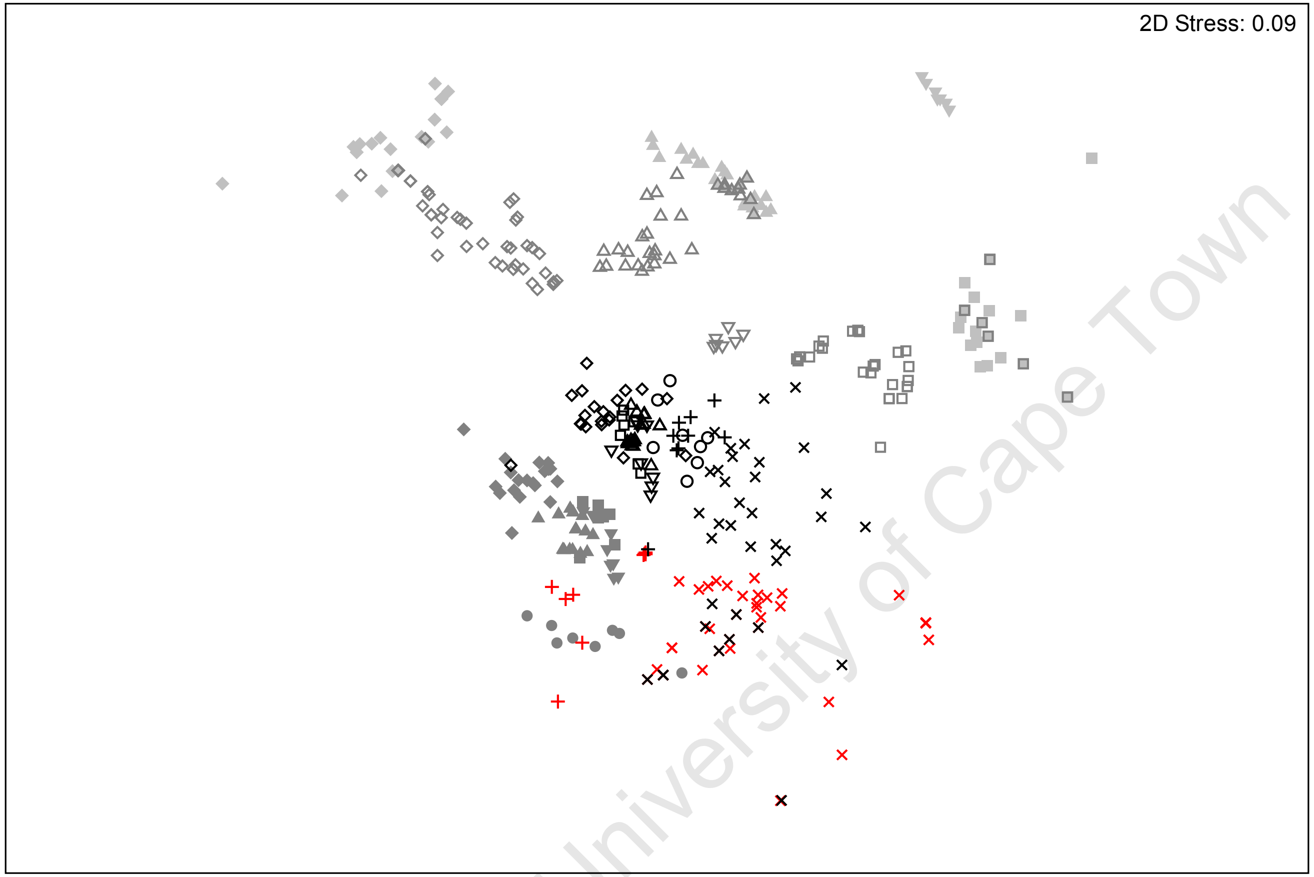
Region Abbreviations: Western Iberia (W Ib); Southern Iberia (S Ib); Eastern Iberia (E Ib); Cantabria coast and Languedoc (Cant); Italian Peninsula (It P); Gulf of Venice (Po); North Eastern Adriatic (Dal); South Eastern Adriatic (Adr); Ionian Sea Drainages (Ion); Aegean Sea Drainages (Aeg); Vardar (Var); Thrace (Thr); Western Anatolia (W An); Southern Anatolia (S An); Central Anatolia (C An); Northern Anatolia (N An); California (Cal); Chile; south-western Australia (SW Aus) and the south-western Cape (SW Cape).

Figure 4.4: Multidimensional Scaling Plot summarising changes in taxonomical similarity in eco-regional fish composition between the historical (▲) and present-day (Δ) assemblages. Convergence of regions into the same multivariate space provides evidence for taxonomic homogenization over time.



Region Abbreviations:
Western Iberia (W Ib); Southern Iberia (S Ib); Eastern Iberia (E Ib); Cantabria coast and Languedoc (Cant); Italian Peninsula (It P); Gulf of Venice (Po); North Eastern Adriatic (Dal); South Eastern Adriatic (Adr); Ionian Sea Drainages (Ion); Aegean Sea Drainages (Aeg); Vardar (Var); Thrace (Thr); Western Anatolia (W An); Southern Anatolia (S An); Central Anatolia (C An); Northern Anatolia (N An); California (Cal); Chile; south-western Australia (SW Aus) and the south-western Cape (SW Cape).

Figure 4.5: Multidimensional Scaling Plot summarising changes in functional similarity in eco-regional fish composition between the historical (▲) and present-day (Δ) assemblages. Convergence of regions into the same multivariate space provides evidence for taxonomic homogenization over time.



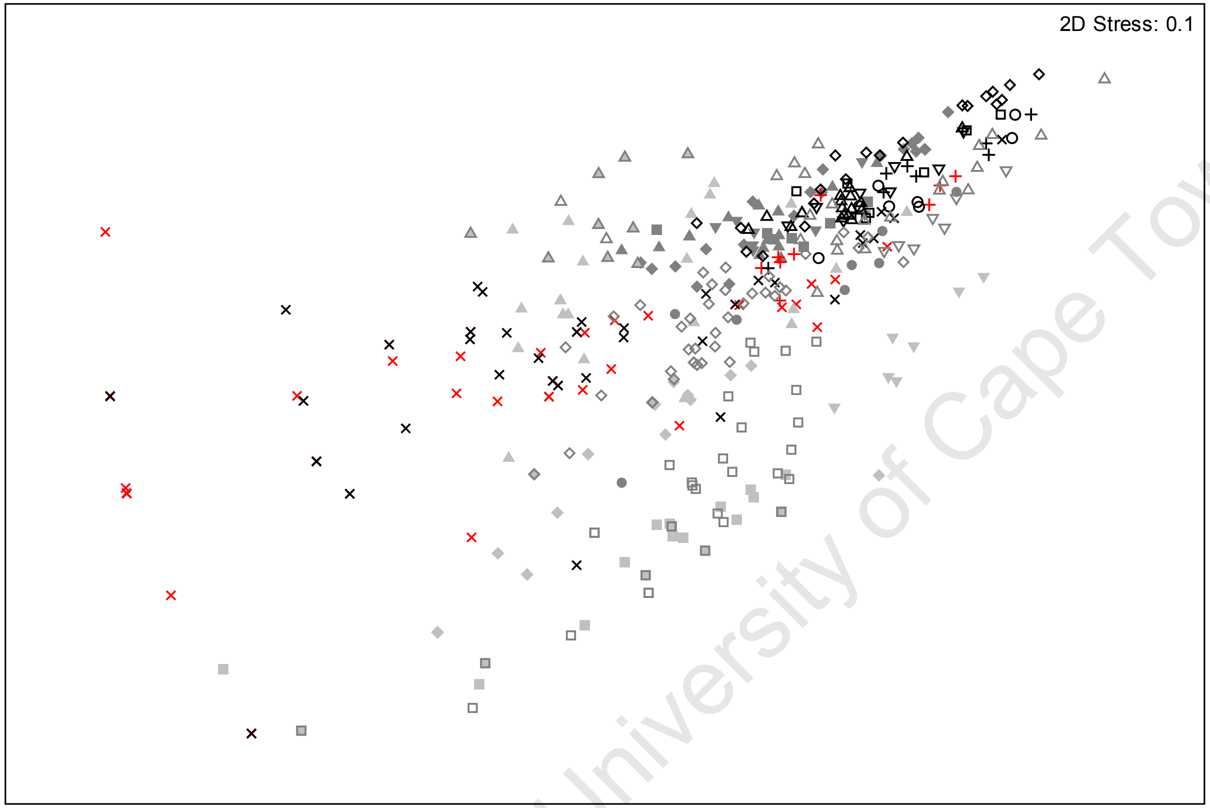
Open symbols indicate present day data and closed symbols historical data

Light grey and dark grey open symbols California ▲, Chile ▼, south-western Australia ■, and the south-western Cape ♦

Dark grey and balck open symbols Western Iberia ▲; Southern Iberia ▼; Eastern Iberia ■; Cantabria coast and Languedoc ♦; North Eastern Adriatic ●; South Eastern Adriatic +; Ionian Sea X

Not included - change not significant Aegean Sea; Vardar; Thrace; Western Anatolia; Southern Anatolia; Central Anatolia; and Northern Anatolia Drainages

Figure 4.6: Multidimensional Scaling Plot summarising the catchment-level differences in taxonomic similarity for the 13 Mediterranean-climate regions showing both historically reconstructed and present day species data as calculated using the Jaccard Index for presence-absence data..



Open symbols indicate present day data and closed symbols historical data.

Light grey and dark grey open symbols California ▲, Chile ▼, south-western Australia ■, and the south-western Cape ♦

Dark grey and balck open symbols Western Iberia ▲; Southern Iberia ▼; Eastern Iberia ■; Cantabria coast and Languedoc ♦; North Eastern Adriatic ●; South Eastern Adriatic Drainages +

Not included - change not significant Ionian Sea; Aegean Sea; Vardar; Thrace; Western Anatolia; Southern Anatolia; Central Anatolia; and Northern Anatolia Drainages

Figure 4.7: Multidimensional Scaling Plot summarising the catchment-level differences in functional similarity for the 12 Mediterranean-climate regions showing both historically reconstructed and present day functional trait data, calculated as the outer product of the species distribution and functional trait matrices, using the Bray-Curtis Index.

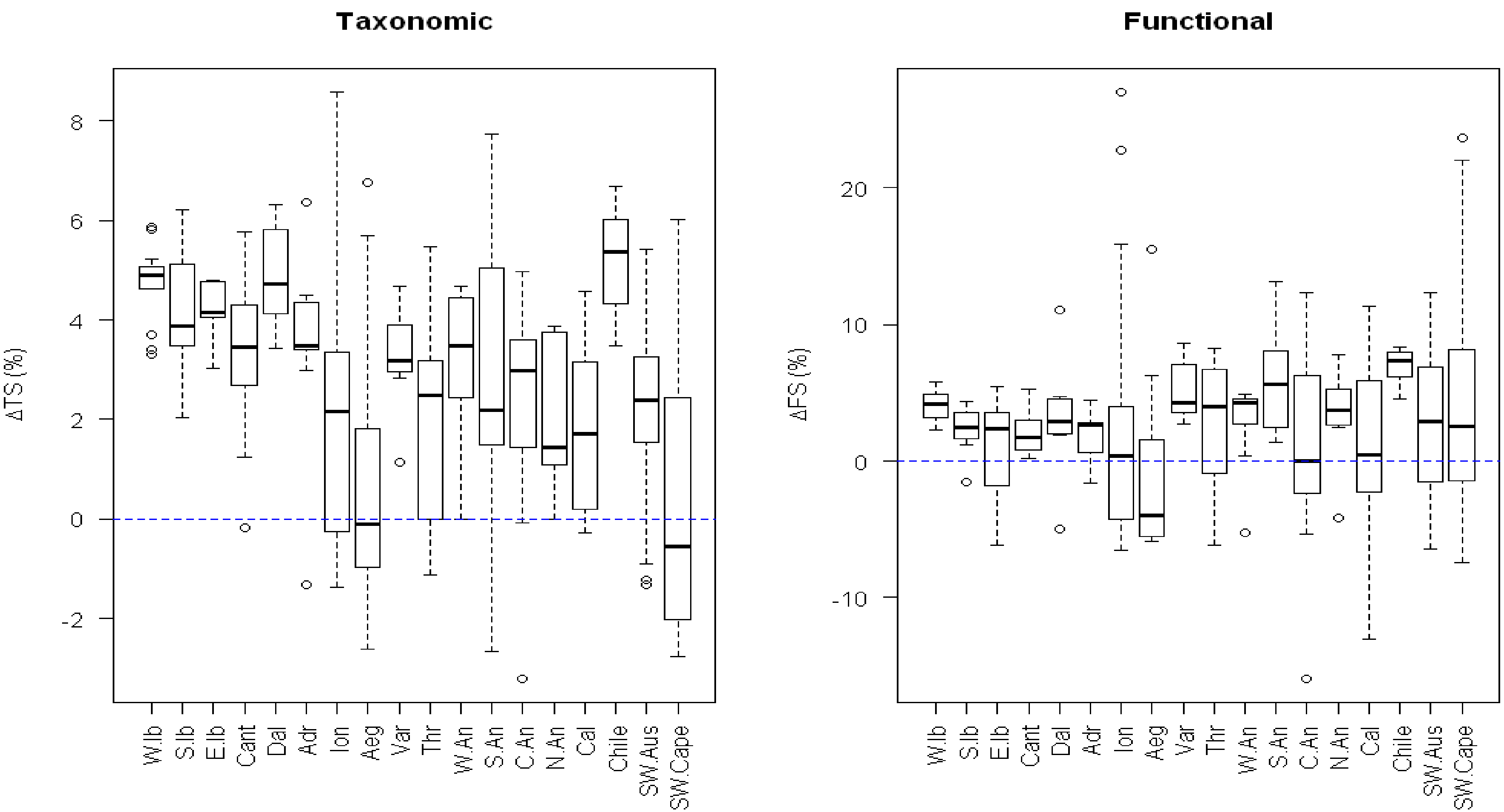


Figure 4.8: Box and whisker plots summarising the catchment-level changes in the inter-regional taxonomic (ΔTS) and functional (ΔFS) compositional similarity between the present day and historical freshwater fish assemblages in the northern Mediterranean Basin, California, Chile, south-western Australia and the south-western Cape. Region abbreviations as per Figure 4.5

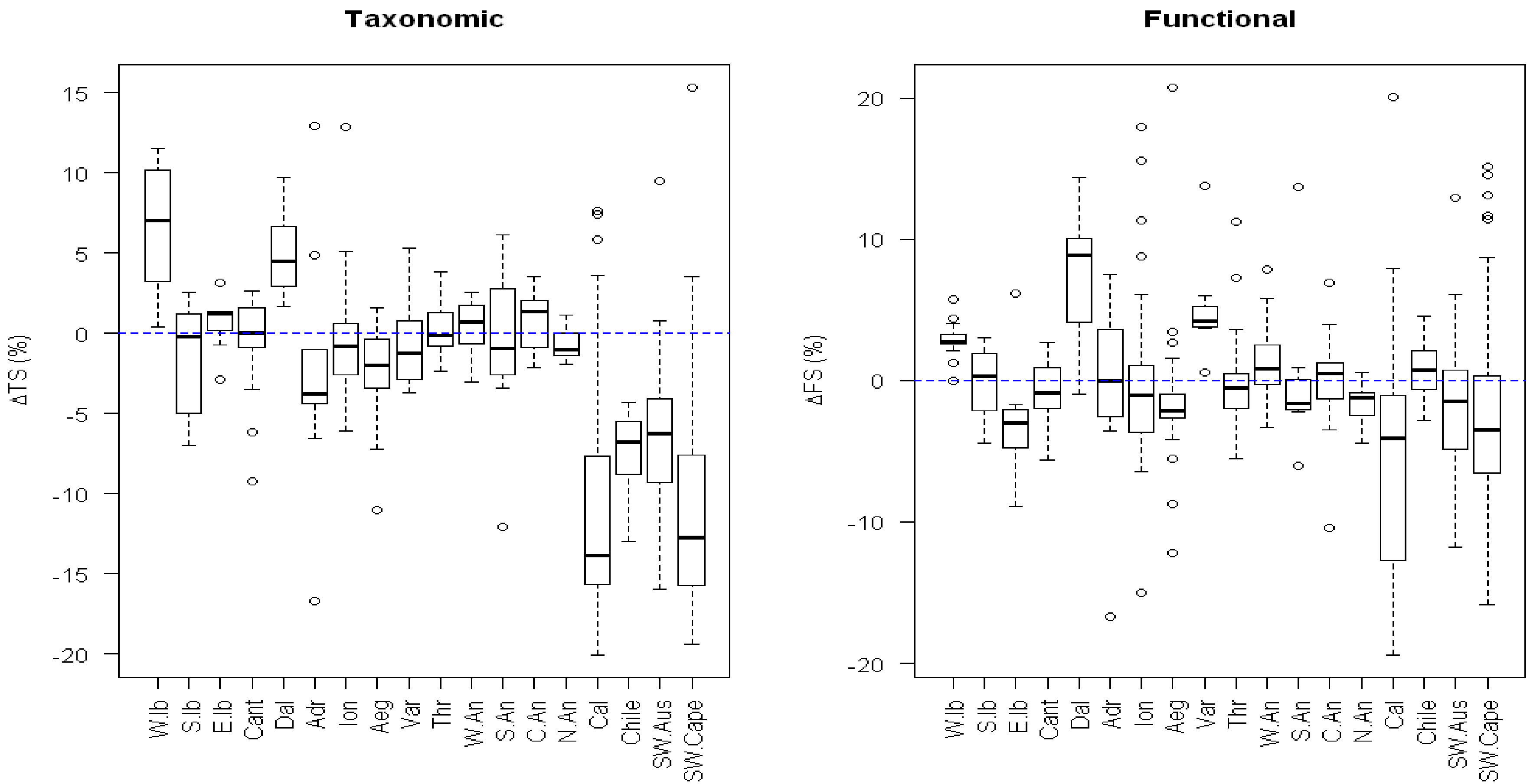


Figure 4.9: Box and whisker plots summarising catchment-level changes in the intra-regional taxonomic (ΔTS) and functional (ΔFS) compositional similarity between the present day and historical freshwater fish assemblages in the northern Mediterranean Basin, California, Chile, south-western Australia and the south-western Cape. Region abbreviations as per Figure 4.5

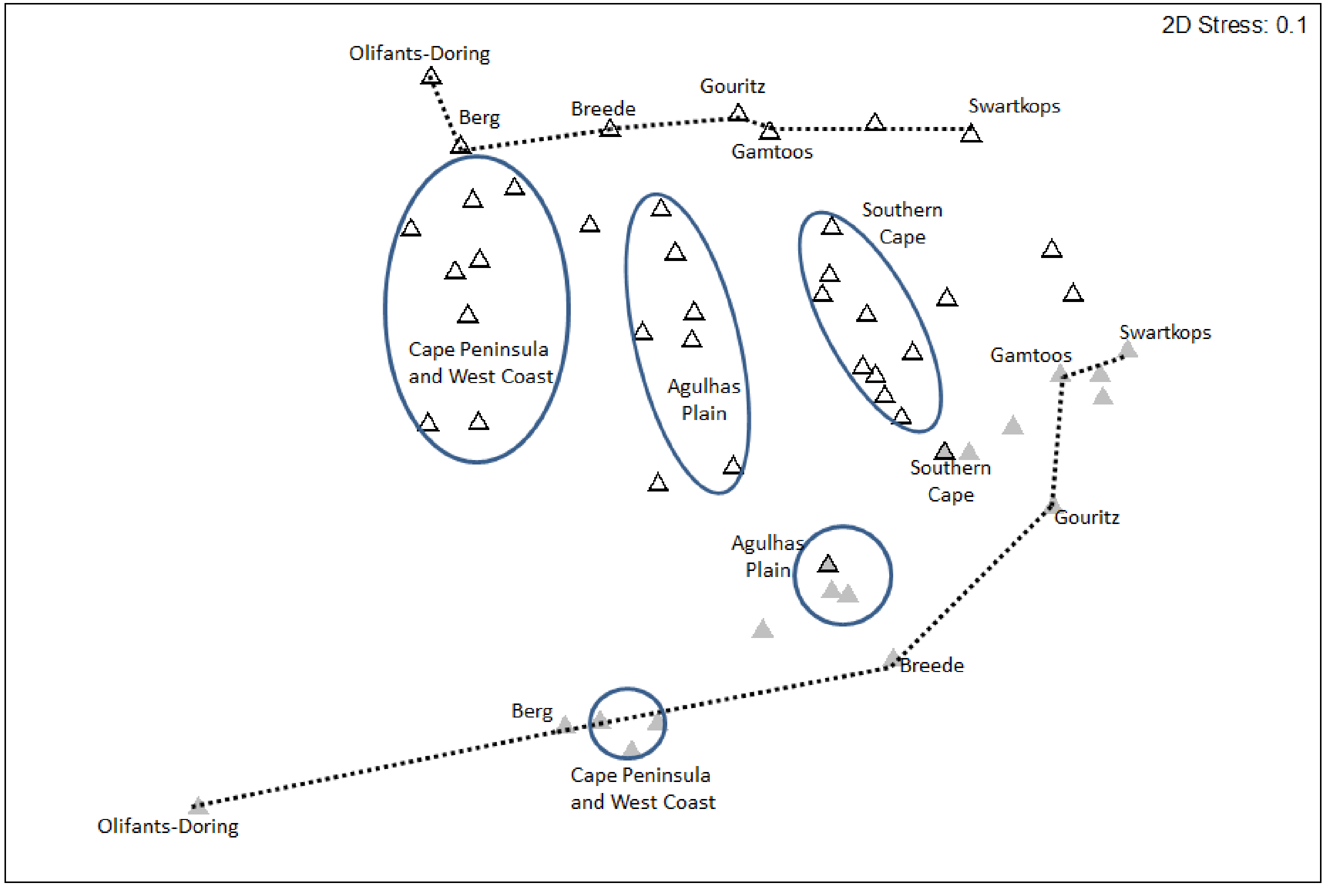


Figure 4.10: Multidimensional Scaling Plot summarising changes in catchment-level taxonomic similarity between the historical (▲) and present-day (Δ) freshwater fish assemblages of the south-western Cape, as calculated using the Jaccard Index for presence-absence data. Dotted lines represent a transect linking the major catchments of the region is presented for the historical and present-day assemblages.

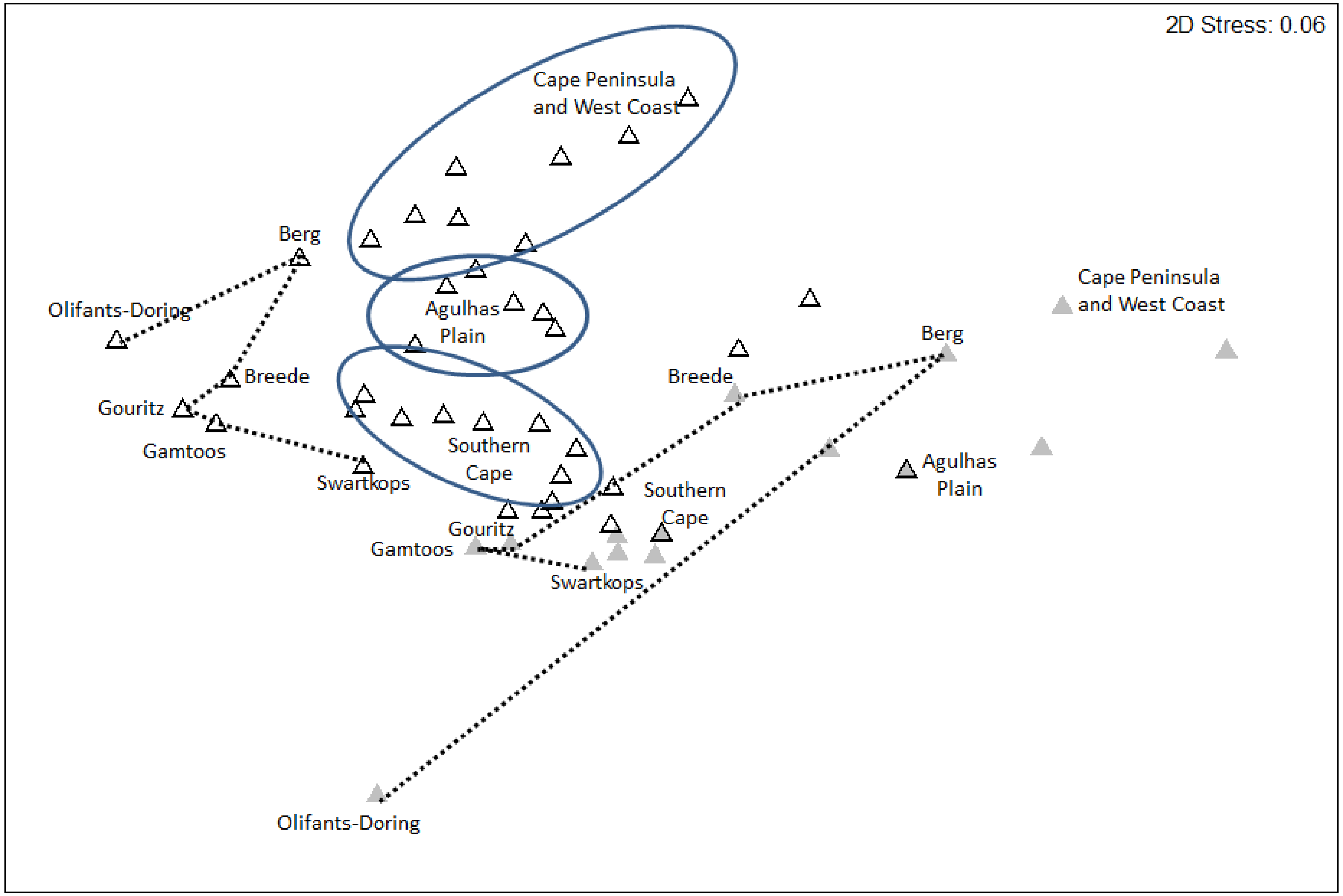


Figure 4.11: Multidimensional Scaling Plot summarising changes in catchment-level functional similarity between the historical (▲) and present-day (Δ) freshwater fish assemblages of the south-western Cape, as calculated using the Bray-Curtis Index. Dotted lines represent a transect linking the major catchments of the region is presented for the historical and present-day assemblages.

Tables

Table 4.1: Reference list used in compiling the catchment-level database for the freshwater fishes of Mediterranean-climate regions.

Region	**Number of catchments**	**References**
Iberian Peninsula	35	Clavero and García-Berthou (2006), Kottelat and Freyhof (2007), and Filipe *et al.* (2009)
France	20	Keith and Allardi (2001), Kottelat and Freyhof (2007) and the Office National de l'Eau et des Milieux Aquatiques (ONEMA - the national fisheries organization responsible for the protection and conservation of freshwater ecosystems in France)
Italy		Zerunian (2004), Kottelat and Freyhof (2007) and Nocita and Zerunian (2007)
Eastern Adriatic (Dalmatian Coast and Albania)	17	Mrakovčić and Mišetić (1990), Povž and Sket (1990), Mrakovcic and Misetic (1990), Leiner *et al.* (1995), Mrakovcic *et al.* (1995), Georgiev (1999), Georgiev (2004), Cake and Miho (2005), Cullaj *et al.* (2005), Šanda *et al.* (2005), Mrakovčić *et al.* (2006), Schneider-Jacoby *et al.* (2006), Šorić (2006), Economou *et al.* (2007), Kottelat and Freyhof (2007), Šanda *et al.* (2008a, b), Shumka *et al.* (2008), Zupančič (2008), Jelić *et al.* (2009), Marić and Šorić (2009), Šanda and Kovačić (2009), Šanda and Vukić (2009), Talevski *et al.* (2009), Bogutskaya *et al.* (2010), Bogutskaya and Zupančič (2010), Shumka *et al.* (2010a, b), Zupančič *et al.* (2010), and Šanda *et al.* (in preparation)
Greece	90	Economou *et al.* (2007) and Kottelat and Freyhof (2007)
Turkey	53	Herbek and Wildekamp (2003), Barlas and Dirican (2004), Özuluğ *et al.* (2005), Turan *et al.* (2005), Freyhof *et al.* (2008), Geldiay and Balık (2009), Yildirim *et al.* (2009), Freyhof and Özulug (2010), and Kara *et al.* (2010)
California	32	Moyle (2002)
Chile	7	Eigenmann (1908, 1921, 1924), Oliver (1949), Mann (1954), Campos (1970), Duarte *et al.* (1971), McDowall (1971), Arratia (1978), Bahamondes *et al.* (1979), Arratia *et al.* (1981), Vila *et al.* (1981), Campos (1982), Ruiz (1996), Habit (1998), Vila *et al.* (1999), Dyer (2000a, b), Iriarte *et al.* (2005), Habit *et al.* (2006), Ortiz-Sandoval *et al.* (2009), Unmack *et al.* (2009) and Zunino *et al.* (2009)
South-western Australia	33	Morgan *et al.* (1998, 2004, 2006)
South-western Cape	48	Scott and Hamman (1984), de Moor and Bruton (1988), Bills (1999), Skelton (2001), Impson *et al.* (2002b), Russell (2002), Russell and Impson (2006), Impson (2007), and Paxton and King (2009), supplemented with unpublished data from Cape Nature (N.D. Impson and A. Turner 2009 pers. comm.), South African Institute for Aquatic Biodiversity (E.R. Swartz, 2009 pers. comm.), Marine and Coastal Management (S. Lamberth, 2009 pers. comm.) and the River Health Project (S.N.P. Buthelezi, 2009 pers. comm.).

Table 4.2: Functional traits selected for the evaluation of functional homogenization of freshwater fish faunas in Mediterranean-climate regions.

Biological Variable	**Category**
1) Adult trophic status	Assigned according to the main food items (>50% of adult diet): 1 - planktivore; 2 – herbivore and detritivore; 3 - invertivore; 4 - omnivore; 5 - piscivore.
2) Size of native range	1 - range < 5% of one zoogeographic sub-region; 2 - range 5–50% of one zoogeographic sub-region; 3 - range >50% of one zoogeographic sub-region; 4 - range more than one zoogeographic sub-region.
3) Parental care	1 - spawn over open substrate with no parental care; 2 - brood hiders, hiding eggs with no additional care; 3 - guarders, guarding their embryos and/or larvae; 4 - bearers, carry their embryos with them.
4) Population doubling time	1 - < 15 months; 2 – 1.4-4.4 years; 3 – 4.4-14 years; 4 – > 14 years.
5) Maximum adult size	1 - < 10 cm; 2 - 11– 20 cm; 3 - 21–40 cm; 4 - 41–80 cm; 5 - 81–160 cm; 6 - >160 cm
6) Physiological tolerance	physiological tolerance of the species to salinity, pollution, habitat degradation or temperature. 1 - intolerant, fishes with low tolerance to temperature and/or salinity, 2 - moderately tolerant fishes, 3 - tolerant fishes, 4 - extremely tolerant fishes
7) Migration	extent of migration: 0 – non-migratory, 1 - potadromois, 2 - diadromous, 3 – amphidromous

Table 4.3: Regional summary statistics of fish introductions and biotic homogenization for the Mediterranean-climate ecoregions. Reported values include the number of native (N), extirpated (X), translocated (T), and introduced (I) freshwater fish species, the pair-wise taxonomic similarity (TS), the average historical pair-wise taxonomic similarity (TS_H), the average change in pair-wise taxonomic similarity (ΔTS_{av}), the pair-wise functional similarity (FS), the average historical pair-wise functional similarity (FS_H), and the average change in pair-wise functional similarity (ΔFS_{av}) between the historical and present-day fish faunas.

	No. of Species				Taxonomic Similarity			Functional Similarity		
	N	**X**	**T**	**I**	**TS %**	**TS_H %**	**ΔTS_{av} %**	**FS %**	**FS_H %**	**ΔFS_{av} %**
All Regions	**492**	**16**	**46**	**136**		**7.72**	**7.02**		**68.06**	**8.17**
Northern Mediterranean	**387**	**13**	**33**	**91**		**12.20**	**7.56**		**73.36**	**7.57**
Western Iberia	31	1	2	15	65.22	8.17	7.79	65.22	71.81	5.90
Southern Iberia	28	2	2	17	57.78	9.48	6.96	57.78	70.13	6.43
Eastern Iberia	26	1	5	24	50.00	9.59	10.07	50.00	68.95	10.87
Cantabrian Coast Languedoc	49	2	10	22	65.71	12.36	8.58	65.71	72.80	7.54
Italian Peninsula	21	0	*	39	35.00	12.36	9.18	35.00	62.49	19.04
Gulf of Venice (Po Drainages)	41	0	*	32	56.16	11.62	9.34	56.16	75.09	3.89
Dalmatian Coast	56	1	1	29	64.71	6.82	10.89	64.71	71.37	5.35
South East Adriatic	53	1	6	24	67.53	7.52	7.52	67.11	73.05	6.24
Ionian Drainages	37	0	9	26	58.73	7.38	8.66	58.73	73.60	7.37
Aegean Drainages	29	0	0	10	74.36	7.32	4.22	74.36	69.92	2.42
Vardar	38	0	4	12	76.00	11.68	5.97	76.00	74.94	5.07
Thrace	79	0	7	18	81.44	12.58	6.36	79.07	67.02	8.57
Western Anatolia	51	1	0	8	84.75	10.16	5.61	83.64	74.51	5.95
Southern Anatolia	70	0	0	7	89.74	6.40	4.48	90.54	67.32	9.70
Central Anatolia	46	8	0	7	80.39	9.79	4.19	81.71	63.40	12.63
Northern Anatolia	71	1	0	10	86.42	10.91	5.89	85.33	69.25	11.21
Non Mediterranean	**105**	**3**	**14**	**66**		**0.46**	**7.12**		**57.67**	**4.14**
California	38	3	12	44	46.67	0.00	5.41	42.68	71.09	3.93
Chile	27	0	0	21	56.25	0.15	7.36	55.10	63.73	12.11
SW Australia	10	0	0	10	50.00	0.15	6.61	52.63	37.66	9.06
SW Cape	31	0	2	16	65.96	0.00	5.38	65.96	62.98	10.10

Table 4.4: Ordinal level taxonomic selectivity of freshwater fish species introduced into Mediterranean-climate ecoregions compared to a random selection from the global species pool. Binomial probabilities adjusted for multiple comparisons by the procedure of Benjamini and Hochberg (1995). Significant level of the results: not significant (ns), $p < 0.1$ (*), $p < 0.05$ (**), $p < 0.001$ (***).

	Acipenseriformes	Polyodontiformes	Anguilliformes	Clupeiformes	Cypriniformes	Characiformes	Siluriformes	Esociformes	Osmeriformes	Salmoniformes	Atheriniformes	Cypriniodontiformes	Perciformes
All Regions	***	**	**ns**	**Ns**	***	***	***	**ns**	**ns**	***	**ns**	**ns**	*
Northern Mediterranean	***	**	**ns**		***		***	**ns**		***	**ns**	**ns**	*
Western Iberia					ns		ns	*		*		ns	ns
Southern Iberia					*		ns	*		ns		ns	ns
Eastern Iberia					**		*	*		*		ns	ns
Cantabrian Coast Languedoc					**		ns			**		ns	ns
Italian Peninsula					**		*			*	ns	ns	ns
Gulf of Venice (Po Drainages)	ns		ns		**		ns			***		ns	ns
Dalmatian Coast					**		**	**		***		ns	ns
South East Adriatic	**				*		*	*		*		ns	ns
Ionian Drainages	***	**			ns		ns			***		ns	ns
Aegean Drainages					*					**		ns	
Vardar					ns					***		ns	ns
Thrace	**				ns		ns			***		ns	ns
Western Anatolia					ns					ns		ns	ns
Southern Anatolia					ns		ns			ns		ns	ns
Central Anatolia					ns					ns		ns	ns
Northern Anatolia					ns					ns		ns	ns
Non Mediterranean	**			**ns**	**ns**	**	**		**ns**	***	**ns**	**ns**	**
California				ns	ns		ns		ns	**	ns	ns	**
Chile	**				ns	ns	ns			***	ns	ns	ns
SW Australia					ns					**		ns	ns
SW Cape					ns		ns			*		ns	**

Table 4.5: Family level taxonomic selectivity of freshwater fish species introduced into Mediterranean-climate ecoregions compared to a random selection from the global species pool. Binomial probabilities adjusted for multiple comparisons by the procedure of Benjamini and Hochberg (1995). Significant level of the results: not significant (ns), $p < 0.1$ (*), $p < 0.05$ (**), $p < 0.001$ (***).

	Acipenseridae	Polyodontidae	Anguillidae	Clupeidae	Cyprinidae	Cobitidae	Balitoridae	Catostomidae	Characidae	Ictaluridae	Siluridae	Clariidae	Loricariidae	Esocidae	Osmeridae	Salmonidae	Atherinidae	Atherinopsidae	Fundulidae	Poeciliidae	Moronidae	Terapontidae	Centrarchidae	Percidae	Cichlidae	Gobiidae
All Regions	***	**	ns	ns	***	ns	**	*	***	***	ns	ns	**	*	ns	***	ns	ns	*	ns	**	ns	***	*	**	**
Northern Mediterranean	***	**	ns		***	*	*	ns		**	ns	ns	*	*		***		ns	ns	ns			*	ns	**	*
Western Iberia					ns		ns			ns	ns			**		*				ns			**	ns	ns	
Southern Iberia					**					ns				**		ns			ns	ns			**	ns	ns	
Eastern Iberia					**	*				ns	ns			**		*			ns	ns			**	*		
Cantabrian Coast Languedoc					**	**				*	ns					**				ns			**	*		
Italian Peninsula					***					**	ns	ns				**		ns		ns			**	**		**
Gulf of Venice (Po Drainages)	ns		ns		**	ns				**	ns	ns				**				ns			**	ns	ns	
Dalmatian Coast					*					ns	ns			*		*				ns			*	ns		
South East Adriatic	**				***					*	*			**		**				**			**	**		
Ionian Drainages	***	**			**					ns	ns	ns				***				ns			**	ns	ns	
Aegean Drainages					ns											**				ns						
Vardar					ns											***				ns			*			
Thrace	**				ns	ns		ns		ns	ns					**				ns			ns			
Western Anatolia					ns											ns				ns			ns	ns	ns	
Southern Anatolia					***								**			ns				ns				ns		
Central Anatolia					ns											ns				ns				ns		
Northern Anatolia					ns											ns				ns			ns	ns	ns	
Non Mediterranean	**			ns	ns			*	*	***		ns			ns	***	ns	ns	ns	*	**	ns	***	ns	ns	ns
California				**	*			ns		***					**	**	*		*	ns	***		***	*	ns	*
Chile	**				ns				ns	***						***		ns		**					ns	
SW Australia					ns											ns				ns		ns		ns	ns	
SW Cape					ns							ns				ns				ns			***		ns	

Table 4.6: Summary of the native freshwater fish faunas, introductions and extirpations in the northern Mediterranean Basin, California, Chile, south-western Australia and the south-western Cape.

	Native Species per catchment					Introductions per catchment							Extirpations per catchment				
	No of catchments in study	mean	sd	max	min	No of catchments recording introductions	mean	sd	max	min	No of catchments with >25% introduced	% of catchments with >25% introduced	No of catchments recording extirpations	mean	sd	max	min
Western Iberia	20	12.05	3.09	21	7	20	7.30	1.84	13	5	20	100.0	15	0.80	0.52	2	0
Southern Iberia	8	14.25	4.23	21	10	8	8.63	3.50	13	4	8	100.0	4	0.75	0.89	2	0
Eastern Iberia	7	14.00	3.79	20	8	7	11.00	6.38	20	6	7	100.0	2	0.29	0.49	1	0
Cantabrian Coast Languedoc	20	17.15	8.29	32	7	20	8.35	4.92	18	2	17	85.0	2	0.15	0.49	2	0
Dalmatian Coast	8	15.38	7.67	29	5	8	10.75	5.82	21	5	6	75.0	2	0.25	0.46	1	0
South East Adriatic	9	18.11	7.37	30	12	9	9.67	4.82	17	1	8	88.9	6	1.00	1.00	3	0
Ionian Drainages	37	5.76	4.91	21	1	27	2.86	4.37	19	0	18	48.6	0	0.00	0.00	0	0
Aegean Drainages	25	3.28	2.91	13	1	11	0.92	1.53	5	0	10	40.0	0	0.00	0.00	0	0
Vardar	7	20.57	10.20	32	7	7	5.00	2.65	9	1	2	28.6	0	0.00	0.00	0	0
Thrace	24	13.50	9.92	35	1	16	2.96	3.59	12	0	7	29.2	1	0.04	0.20	1	0
Western Anatolia	12	11.17	5.41	23	1	11	2.42	1.83	7	0	2	16.7	1	0.25	0.87	3	0
Southern Anatolia	9	14.11	10.24	30	2	6	1.78	1.72	5	0	3	33.3	1	1.22	3.67	11	0
Central Anatolia	18	9.94	7.29	27	4	13	1.44	1.25	4	0	2	11.1	5	0.78	1.44	5	0
Northern Anatolia	9	14.78	8.58	31	1	6	2.33	3.00	9	0	0	0.0	1	0.11	0.33	1	0
California	32	8.69	4.31	24	4	22	10.66	10.57	41	0	22	68.8	11	0.69	1.15	4	0
Chile	7	16.29	4.35	24	11	7	10.43	4.43	18	5	6	85.7	7	1.29	0.49	2	1
SW Australia	33	4.58	1.94	8	1	25	1.91	1.81	6	0	19	57.6	2	0.06	0.24	1	0
SW Cape	48	5.83	2.35	11	2	38	4.25	3.68	12	0	28	58.3	2	0.04	0.20	1	0

Table 4.7: Summary of the freshwater fish species most widely introduced into, or extirpated from, 18 Mediterranean-climate ecoregions. The figures reported represent the change in the proportion of catchments occupied by the species within in the respective ecoregions. Only the species that recorded the greatest change in occupancy have been included here.

	Western Iberia	Southern Iberia	Eastern Iberia	Cantabrian Coast Languedoc	Dalmatian Coast	South East Adriatic	Ionian Drainages	Aegean Drainages	Vardar	Thrace	Western Anatolia	Southern Anatolia	Central Anatolia	Northern Anatolia	California	Chile	SW Australia	SW Cape
Gambusia holbrooki	0.90	1.00	0.71	0.50	1.00	0.89	0.57	0.32	0.71	0.63	0.75	0.67	0.58	0.38		0.71	0.76	
Cyprinus carpio	1.00	0.88	1.00	0.85	0.88	0.89	0.41								0.47	1.00		0.35
Oncorhynchus mykiss	0.45	0.38	0.71	0.85	0.75	0.89	0.27		0.57		0.33	0.33		0.25		1.00	0.3	0.33
Carassius auratus	1.00	0.75	1.00	0.35	0.25	0.78								0.25	0.38	0.71	0.3	
Lepomis gibbosus	0.75	0.75	0.43	0.65	1.00	0.44			0.57	0.29	0.25							
Carassius gibelio				0.40	0.88	0.33			0.86	0.45	0.42	0.33	0.42	0.38				
Micropterus salmoides	0.75	0.88	0.43												0.59			0.54
Pseudorasbora parva			0.29	0.30	0.38	0.89			0.43	0.25	0.25			0.25				
Salmo trutta	0.35		0.29	0.25	0.50										0.31	1.00		
Gobio lozanoi	1.00	0.75	0.71															
Esox lucius		1.00	0.71		0.50													
Tinca tinca					0.88											1.00		
Sander lucioperca			0.43	0.45	0.25						0.25			0.25				
Gambusia affinis						0.33									0.5	0.71		
Ctenopharyngodon idella					0.25	0.44			0.29							0.43		
Perca fluviatilis			0.29	0.40	0.25												0.27	
Alburnus alburnus		0.50	0.57															
Lepomis macrochirus															0.63			0.42
Australoheros facetum		0.25														0.71		
Ameiurus nebulosus															0.38	0.57		
Cobitis paludica	0.45		0.43															
Salvelinus fontinalis			0.29													0.57		
Carassius carassius				0.60	0.25													
Gymnocephalus cernua				0.45	0.25													
Rutilus rutilus			0.43		0.25													
Alburnoides bipunctatus						0.67												
Oreochromis mossambicus																		0.67
Micropterus dolomieu															0.31			0.35
Silurus glanis				0.35	0.25													
Odontesthes bonariensis																0.57		
Ameiurus melas		0.25													0.31			
Lepomis cyanellus															0.53			
Pimephales promelas															0.47			
Pomoxis nigromaculatus															0.47			
Tilapia sparrmanii																		0.46
Hypophthalmichthys molitrix						0.44												
Ictalurus punctatus															0.44			
Notemigonus crysoleucas															0.44			
Dorosoma petenense															0.38			
Fundulus heteroclitus		0.38																
Pomoxis annularis															0.38			
Thymallus thymallus					0.38													
Chondrostoma nasus				0.35														
Leucaspius delineatus				0.35														
Lepomis microlophus															0.31			
Coregonus cf. lavaretus									0.29									
Coregonus clupeaformis																0.29		
Misgurnus anguillicaudatus			0.29															
Clarias gariepinus																		0.27
Ameiurus catus															0.25			
Scardinius erythrophthalmus					0.25													
Luciobarbus comizo		-0.25																
Nematogenys inermis																-0.43		
Salmo salar	-0.70																	
Diplomystes chilensis																-0.71		
Acipenser sturio		-0.38	-0.29			-0.67												

CHAPTER 5. TWEE RIVER CASE STUDY: NON-NATIVE FISH AND HABITAT DEGRADATION

The previous chapter presented a broad-scale analysis of changes in taxonomic and functional attributes of the freshwater fish assemblages of Mediterranean-climate regions, finding evidence of taxonomic and functional homogenization. The introduction of non-native species and extirpation of native species at regional and catchment levels provided evidence of biotic homogenization at a large spatial scale. However, studies at large spatial scales do not always capture the complex interrelated interactions associated with native fish declines. For example, the loss of native species could be as a result of habitat degradation, land-use change, water quality deterioration (including nutrients and pesticides), or the introduction of non-native species (Marsh-Matthews and Matthews 2000). Many of these factors act synergistically on native fish populations and it is, therefore, important to identify whether the introduction of non-native species, or habitat factors, are the drivers of native species declines (Gurevitch and Padilla 2004, Light and Marchetti 2007, Hermoso *et al.* 2011a). However, this is not easily achieved. In this chapter, a case study is used to explore whether the presence of non-native fish, or habitat-related factors, can be identified as the primary factor preventing the recovery of a critically endangered, range-restricted, endemic species of the Cape Floristic Region, the Twee River redfin *Barbus erubescens* Skelton, 1974.

5.1 Twee River Catchment

The Twee River sub-catchment of the Olifants-Doring River system in the Western Cape Province, South Africa (Figure 5.1), comprises the Suurvlei and Middeldeur Rivers and their respective tributaries, the Buffelshoek and Hex Rivers. The sub-catchment has been isolated from the remainder of the Doring system by three large waterfalls – Die Drift, Middle and Little Augrabies (Impson *et al.* 2007). The geology of the catchment comprises primarily sandstones and quartzites of the Table Mountain Group, interspersed with shale bands of the Bokkeveld Group (Deacon 1983, Impson *et al.* 2007). Natural vegetation is mainly mountain fynbos, predominantly Cederberg sandstone fynbos with bands of northern inland bank shale vegetation and a small area of Kouebokkeveld alluvium fynbos in the upper Middeldeur (Mucina and Rutherford 2006). The Twee River falls entirely within the Western folded mountain aquatic ecoregion (Kleynhans *et al.* 2005). River water is clear, with pH 6.2–7.7, low level of dissolved solids (conductivity 20–120 mS/cm), and water temperatures range between 5°C in winter and 27°C in summer (Impson *et al.* 2007).

5.1.1 Land-use

The Twee River catchment has mixed land-use, with intensive fruit orchards (citrus and deciduous) and vegetable production on the banks of the Middeldeur and Suurvlei tributaries and private nature reserves in the lower Twee River. Cultivated land along the 30 km of river was estimated at 570 ha in 1996, comprising of 170 ha of mixed deciduous fruit (apples, pears and peaches), 75 ha of citrus, and 325 ha of vegetables (onions, tomatoes and pumpkins). A further 200 ha of orchards was being developed on the banks of the Suurvlei at the time (Marriott 1998, Impson *et al.* 2007). Orchards and vegetables grown in winter-rainfall areas of the Cape Floristic Region require fertiliser and pesticide applications, with supplementary irrigation during summer. In 1996, water abstraction for agriculture was estimated to be 15% of the catchment's total annual yield, with abstraction exceeding runoff from November–March (Marriott 1998, Impson *et al.* 2007). Subsurface drainage infrastructure returning excess irrigation water to the river may contribute to the reduced water quality in summer (Marriott 1998, Impson *et al.* 2007). A 2007 survey of land-use in the Suurvlei tributary estimated a 300% increase in the area under orchards between 1949 and 2003, while the production of one farm increased by 500% between 1999 and 2006 (Davies 2007).

5.1.2 Pesticides

The diversity of crops produced in the catchment requires an extensive schedule of pesticide application, with spraying being concentrated during the citrus (April–September) and deciduous fruiting (September–March) seasons (Marriott 1998, Impson *et al.* 2007). Four pesticides used in the Suurvlei River were identified as highly toxic to fish: the insecticides Dursban (Chlorpyrophos), Gusathion (Azinphos-methyl), and Sanamectin (Abamectin) and the fungicide Dithane (Mankozeb) (Davies 2007). Of these, Gusathion was being phased out at the time. The majority of these pesticides are applied to the fruit trees by spray application, a method linked to spray-drift contamination of aquatic ecosystems (Schulz 2001). The impact of spraying is likely to be exacerbated by the close proximity of intensive agriculture to the rivers (often less than 10 m) (Marriott 1998, Impson *et al.* 2007). It is anticipated that the intensive agriculture has affected the water quality and aquatic fauna in the tributaries of the Twee River, although no study of the impact of these pesticides on the endemic fishes of the Twee River catchment has been conducted to date.

5.1.3 Native and Non-native Fish

The fish in the Twee River appear to have evolved in isolation from those in the remainder of the Olifants-Doring system. The two native species, the Twee River redfin *Barbus erubescens* and a recognised distinct taxon of Cape galaxias *Galaxias zebratus* Castelnau, 1861, are both endemic to this catchment (Hamman *et al.* 1984, Skelton 2001, Impson *et al.* 2007) and among South Africa's most endangered freshwater fishes (Impson *et al.* 2007). Twee River redfin are currently listed as Critically Endangered (Tweddle *et al.* 2009) and it is expected that the *Galaxias* taxon, currently listed as Data Deficient (Tweddle *et al.* 2009), will also be listed as Critically Endangered once it has been formally described. Threats to the Twee River redfin are listed as non-native fish, habitat destruction, pesticides, and pollution (Tweddle *et al.* 2009).

Five species of fish have been introduced into the Twee catchment (Hamman *et al.* 1984, Marriott 1998, Impson *et al.* 2007). These are three North American species: smallmouth bass, *Micropterus dolomieu* (Lacepède, 1802), rainbow trout, *Oncorhynchus mykiss* (Walbaum, 1792) and bluegill sunfish, *Lepomis macrochirus* Rafinesque, 1819; a translocated species endemic to the Cape Floristic Region, the Cape kurper *Sandelia capensis* (Cuvier, 1831); and the Olifants-Doring endemic species, the Clanwilliam yellowfish *Labeobarbus capensis* (A Smith, 1841). Of these, the Cape kurper had invaded the largest portion of the catchment by 1997 (Marriott 1998, Impson *et al.* 2007) - Figure 5.2.

5.1.4 Instream Habitat

The instream habitat characteristics of the Suurvlei tributary are significantly different from those of the Middeldeur (Davies 2007). The Suurvlei is fast-flowing, shallow, narrow, with cobble substrate and very little in-stream vegetation, whereas the Middeldeur is slow-flowing, deep, wide, with silt/mud substrate and dominated by instream vegetation, primarily palmiet *Prionium serratum* (Linnaeus, 1758) (Davies 2007). While the two tributaries appear to be very different, it is unclear whether instream habitat suitable for Twee River redfin or Cape galaxias is currently available in the Suurvlei. Adult Twee River redfin are thought to prefer sheltered areas in pools, particularly near overhanging vegetation, and caves under boulders, often in water deeper than 1.5 m, whereas juveniles school in the upper water column of pools near palmiet, or overhanging vegetation (Marriott 1998, Impson *et al.* 2007). Further, a slug of sand, resulting from the rupture of a dam in the Suurvlei catchment (Jannie Hanekom, Farm Ysterplaat, pers. comm. 2008), is moving down the Suurvlei, filling pools and smothering instream habitat (pers. obs.). The riparian zone of the Suurvlei tributary has also been invaded by a number of non-native plant species and areas that were historically wetland have been eroded into stream channels (Davies 2007).

5.1.5 Aims of this Chapter

Historical records indicate that Twee River redfin and Cape galaxias were present in the Suurvlei tributary, but the 1996/7 surveys only found a small population of Twee River redfin persisting above the Cape kurper distribution - Figure 5.2 (Marriott 1998, Impson *et al.* 2007). The accidental introduction of Cape kurper into the Suurvlei in the 1950s (Hamman *et al.* 1984, Marriott 1998, Impson *et al.* 2007) has resulted in the invasion of the Twee River system by Cape kurper and this has coincided with the disappearance of the two endemic species from the Suurvlei (Marriott 1998, Impson *et al.* 2007). However, both endemic species co-occur with Cape kurper in the Middeldeur. The remaining redfin population above the Cape kurper invasion in the Suurvlei should be a source population for the re-colonisation of the historical range in the Suurvlei, yet there is no evidence of this occurring.

The Suurvlei tributary has been included in the Cape Alien Fish Removal Project as a priority river for non-native fish eradication (Impson 2007). The project proposes the removal of the translocated Cape kurper from the Suurvlei tributary using the piscicide rotenone, thus facilitating the recovery of the Twee River redfin in the Suurvlei tributary [See Section 3.5.2 and Marr *et al.* (in press)]. This constitutes the first step in the proposed conservation plan for the endemic fish of the Twee River (R. Bills, South African Institute for Aquatic Biodiversity (SAIAB), 2006, pers. comm.). Following the successful removal of Cape kurper and the recovery of redfin in the Suurvlei River, the plan proposes to remove introduced fish from the remainder of the Suurvlei, Middeldeur, and Twee Rivers using piscicides. Natural re-colonisation of the treated reaches by redfin and galaxias would then occur from the upper Middeldeur. The treatment of the Twee could be completed in two phases with the area above Die Drift waterfall being treated first.

The Suurvlei tributary is highly modified through the majority of the intervention area proposed for the Cape Alien Fish Removal Project, and it has been questioned whether Twee River redfin and/or Cape galaxias would recolonize, or even survive, in the treated reaches following the proposed intervention (Davies 2007). It is therefore important to identify the stressors currently preventing the re-colonization of the Suurvlei (water quality, pesticide use, instream habitat degradation, or the presence of the non-native Cape kurper) in order to determine whether the proposed project would realize the intended outcome. This would also provide valuable data for the development of a management plan for the endemic fishes of the Twee River system.

5.2 Methods

This study typifies many freshwater fish studies in the Cape Floristic Region. No historical data describing the fish distributions, water quality, or instream habitat are available for the catchment, non-native fish have been introduced, and the landscape has been transformed. In the Twee River, the introduced species have not totally extirpated the native species, as has occurred in the majority of the streams in the Cape Floristic Region. Cape kurper have invaded the entire Suurvlei and the Middeldeur up to the Kunje waterfall (Figure 5.2). The Middeldeur above Kunje waterfall, including the Hex tributary, is less impacted than the remainder of the catchment, but are not comparable to those in the Suurvlei in terms of altitude, gradient, or stream order. Therefore, this study has been limited to a comparison between reaches in the Middeldeur where Cape kurper co-occurs with the two endemic species, and those in the Suurvlei where the two endemic species have been extirpated. The study is, therefore, a comparison between the Suurvlei and Middeldeur tributaries to determine whether differences in water quality, pesticide exposure, instream habitat, or the presence of non-native Cape kurper are preventing re-colonising of the Suurvlei by Twee River redfin. If no statistically significant differences can be found between these factors in the respective tributaries, we could eliminate one, or more, of these factors.

This work was conducted with the approval of the Animal Ethics Committee of the University of Cape Town (**2011/V12/SM**) and the necessary permit from CapeNature (permit no. 0035-AAA004-00800).

5.2.1 Fish Distributions

The survey was completed using fyke nets set overnight facing upstream with the wings spanning the width of the stream. For shallow habitats 600 mm D-ring twin-compartment fyke nets with 300 mm diameter hoops and 2 mm mesh were used. For deeper habitats, 700 mm square twin-compartment fyke nets with 2 mm mesh were used. The nets were set between 17h00 and 18h00 and cleared between 08h00 and 10h00 the following morning. Non-native fish were killed by a blow to the head; endemic fish were counted and released unharmed with minimal handling. An air-space was maintained in the final compartment to minimize the risk of drowning air breathing animals. The nets were aired and dried by day to reduce the risk of transferring organisms between sites. The co-ordinates of each sample site were recorded using a Garmin E-Trex Vista GPS. Fish distribution maps were plotted using the Arc View 3.2 GIS programme. Fish sampling took place in October 2006, February 2007, and May 2007. Additional data were collected in December 2008, November 2010, and April 2011.

5.2.2 Water Quality

Water quality samples were collected at eight sites within the Twee River catchment in April 2011 (Figure 5.3). Physical water chemistry measurements of pH, dissolved oxygen, conductivity and temperature were recorded in triplicate at each site and samples for the analysis of nutrients collected. Temperature and pH were measured using a YSI pH100 handheld meter (0.01 unit resolution, 0.1% accuracy), conductivity was measured using a YSI EC300 (0.1μS/cm resolution, 1% accuracy). Dissolved oxygen was measured using a YSI DO200 (0.1% resolution, 2% accuracy). The nutrient samples were frozen in the field for analysis of soluble reactive phosphate, nitrate, nitrite and ammonia at the Oceanography Department of the University of Cape Town (UCT). The mean and standard deviation of the respective water chemistry parameters were calculated. Two tailed t-tests with unequal variances were used to determine whether there was a significant difference in water chemistry and nutrient variables between the two tributaries. Principal Component Analysis (PCA) was performed to determine the main factors contributing to the differences between the Suurvlei and Middeldeur tributaries using PRIMER-E 6.1.5 (Clarke and Warwick 2001). Each variable was normalised (-1, 1) prior to the PCA. A distance-based test for homogeneity of multivariate dispersion was performed to test whether there was a statistically significant difference in multivariate dispersion for the water chemistry between the two tributaries using the PERMANOVA and PERMDISP routines in PRIMER-E 6.1.5 (Anderson 2001b, Anderson *et al.* 2008). These tests were performed on a resemblance matrix generated from the normalized data using Euclidian distance. The PERMDISP routine was used to determine whether the multivariate dispersion about the group centroid differed between the two tributaries, while the PERMANOVA routine was used to determine whether the position of the group centroids in multivariate space and/or multivariate dispersion about the group centroid differed between the tributaries (Anderson 2001b, Anderson *et al.* 2008). A significant result from the PERMDISP test reduces the interpretability of significant result from a PERMANOVA tests, because it is not possible to determine whether the significant result in PERMANOVA is due to multivariate dispersion, change in the position of the group centroids, or both (Anderson 2001b, Anderson *et al.* 2008). PERMANOVA tests were therefore only performed when the PERMDISP test returned a non-significant result; $p > 0.05$.

Aquatic macro-invertebrates are commonly used as bio-monitoring surrogates for water quality samples because they are visible to the naked eye, easy to identify to family level, have rapid life cycles, and are largely sedentary (Dickens and Graham 2002). Numerous bio-assessment techniques using macro-invertebrates have been developed, including the South African Scoring System (SASS) version 5, which is widely used in the bio-monitoring of rivers in South Africa (Dallas 1997, Dickens and Graham 2002, Dallas 2007, Dallas and Day 2007). Seven macro-invertebrate samples were collected from riffles at five of the eight water quality sites (Figure 5.3). No suitable riffle habitat was available at the other three sites – MID01, MID04, and SUUR04. Unlike the majority of the streams in the Cape Floristic Region, riffle habitat is rare in the Twee River system and all available riffle habitat in the Suurvlei and Middeldeur was sampled. Invertebrate samples were collected using a 400 mm x 400 mm Süber sampler with a 250 µm mesh. The substrate was agitated for a period of two minutes to dislodge invertebrates from the substrate. The macro-invertebrate samples were preserved in 70% ethanol and sorted to family level in the laboratory.

The taxa in each sample were aggregated to site-level. The SASS score for each site was calculated by summing the SASS 5 scores for the taxa identified at that site (Dickens and Graham 2002). The Average Score Per Taxon (ASPT) was calculated as the SASS score divided by the number of taxa present at the site (Dickens and Graham 2002). The SASS score and ASPT results were compared to biological classes proposed to delineate the ecological conditions of the rivers in the south-western Cape (Dallas and Day 2007). The biological bands are as follows: richer that reference (X, SASS > 166, ASPT > 9.0), reference (A, SASS > 137-166, ASPT > 8.2-9.0), below reference (B, SASS > 108-137, ASPT > 7.4-8.2), well below reference (C, SASS > 79-108, ASPT > 6.6-7.4), and impoverished (D, SASS < 79, ASPT < 6.6). A distance-based test for homogeneity of multivariate dispersion was performed to test whether there was a difference in multivariate dispersion of the invertebrate assemblages at the sites using the PERMDISP routine. The PERMANOVA routine was used to determine whether there were statistically significant differences in the centroids of the respective site invertebrate assemblages. These tests were performed on a resemblance matrix generated from the Log (x+1) transformed raw data using Bray-Curtis similarity. A SIMPER analysis was performed on raw data to identify the invertebrate families contributing most to the dissimilarity between the two tributaries using PRIMER-E 6.1.5.

5.2.3 Instream Habitat

Transects of the Suurvlei and Middeldeur Rivers were completed to measure the instream habitat and riparian zone condition during low flow in April 2011 (Figure 5.4). Selected habitat variables were recorded: average depth, width, percentage emergent or submerged vegetation, type of vegetation, and indicators of anthropogenic impact, including the state of the riparian zone following Ross *et al.* (2001). The emergent/submerged vegetation (*Palmiet*, *Potomageton*, *Isolepis*, and sedges/grasses) and substrate type (sand, gravel, cobbles, boulders, and bedrock) were calculated as the percentage of the total stream width. Stream velocity represented by the surface turbulence was scored from 0 to 2 (0 = glassy, 1 = moderate visual aberration, 2 = surface turbulence or "white water") following Ross *et al.* (2001). The state of the riparian zone was evaluated by estimating the extent of natural vegetation, recovering natural vegetation, burned, alien vegetation, felled alien vegetation, bulldozing, and bank erosion. Two additional variables, proximity of orchards and anthropogenic impact, were scored from 1 to 5 (Table 5.1).

Principal Component Analysis was performed on the instream habitat and riparian zone data to identify the main factors contributing to the differences between the Suurvlei and Middeldeur tributaries. The data were normalised (-1, 1) prior to the PCA.

5.2.4 Pesticides

The conventional whole organism toxicity trails *sensu* Marking and Bills (1976) would require more than 300 fish [10 individuals per replicate, 5 replicates at 5 concentrations plus control] per species per pesticide evaluated. This method was not considered suitable in this case, because of the large number of fish required to provide defensible results and the critical endangered status of the Twee River species. An alternative could be to measure the exposure of fish in each tributary to the pesticides and their breakdown products. The presence of pesticides in rivers is difficult to evaluate, as they break down quickly and are usually present below detection levels, but still impact aquatic biota (Fulton and Key 2001). However, exposure to organophosphate or carbamate pesticides, and their breakdown products, have been linked to the inhibition of acetylcholine esterase in the brain and muscle tissue of fish (Fulton and Key 2001). Acetylcholine esterase inhibition can therefore be used as a surrogate to evaluate pesticide exposure between two sites. Inference of differences in pesticide exposure can therefore be conducted using a low number of individuals of a single species; in this case a non-native species. Cape kurper from three sites (three in the Middeldeur and one in the Suurvlei – Figure 5.3) were captured using fyke nets in November 2011 and used in a pilot study to evaluate the acetylcholine esterase inhibition.

The fish were killed by severing the spinal cord. Brain tissue samples were extracted and stored in 1.5 mL eppendorf tubes containing 0.1M phosphate saline buffer, pH 8.0. The samples were frozen in the field and stored at -80 °C prior to enzyme assays. In the laboratory, the samples were thawed over ice, the tissue homogenized in 0.1M phosphate buffer, pH 8.0 (20 mg tissue in 1mL of the buffer) and centrifuged at 1200 rpm. The supernant was decanted for acetylcholine esterase analysis. Acetylcholine esterase activities in the supernant were determined colorimetrically, following the method of Ellman *et al.* (1961) using a spectrophotometer (Novaspec II, Pharmacia LKB, Biochrom Ltd., Cambridge, UK) housed in the Molecular and Cell Biology Department of UCT. The supernant (100 µL) was added to a glass cuvette containing 2.9 mL of 0.1 M phosphate buffer pH 8.0 and 100 µL of 0.01M 5, 50-dithiobis-2-nitrobenzoic acid. The contents were mixed and the absorbance read at 412 nm in 10 s intervals for one minute. Substrate (20 µL of 0.075M acetylthiocholine iodide) was added to the reaction mixture and the absorbance read at 10 s intervals for one minute. The reaction was carried out at 25 °C. Three supernant samples were assayed for each tissue sample. The absorbance values were corrected for non-enzymatic reaction and the acetylcholine esterase activity was calculated using the extinction coefficient of the 5thio-2-nitrobenzoate ion (13,600 Mcm^{-1}) and expressed in µmoles per minute per mg tissue.

An ANOVA was used to determine whether the differences in acetylcholine esterase absorbance within the samples were greater than the differences between the samples at each site. A second ANOVA was used to determine whether differences between the sites can be identified. Tukey's honest significant difference test was used, *post hoc*, to identify the sites contributing to the significant result. The ANOVA and Tukey tests were conducted in the *R* package (R Development Core Team 2011).

5.2.5 Interactions Between Native and Non-Native Fish

Laboratory experiments examining interactions between the non-native fishes and endemic species of the Twee River catchment, including impacts on growth and habitat use, following the experimental design of Marchetti (1999), were planned but could not be completed due to funding limitations and the permit application being declined. The planned laboratory experiments could have provided valuable insights into the nature of the interactions between the endemic species and the introduced Cape kurper, and should be considered in future studies.

<u>Predation or competition for resources</u>

Analyse of gut contents is commonly used to determine the diet of fish (Hyslop 1980, Cortes 1997, Rindorf and Lewy 2004, Layman *et al.* 2005). From the gut content, diet breadth and dietary overlap between native and non-native species can be determined. These studies may not be conclusive where the native species has been largely extirpated by the non-native species, however, as a result of the low probability of finding prey fish in the gut of non-native predators when the prey availability has been reduced (Meffe 1985). Stable-isotope data can be used to construct trophic webs to represent the importance of specific prey items in a fish's diet (Fry 1991, Vander Zanden *et al.* 1997, Vander Zanden and Rasmussen 1999, 2001, Clarke *et al.* 2005, Saito *et al.* 2007). Stable-isotope analyses have been used to demonstrate food competition between native and non-native fishes that has resulted in diet shifts in the native species with detrimental fitness costs (Vander Zanden *et al.* 1999, Vander Zanden and Rasmussen 2002, Vander Zanden *et al.* 2004, Clarke *et al.* 2005, Cucherousset *et al.* 2007). An enrichment of $3.4^{0}/_{00}$ in N^{15} has been estimated to be the difference between trophic levels (Vander Zanden *et al.* 1997) and can be used as a guideline to determine whether a native species is likely to be part of the diet of the non-native species.

The depth and cover of the large stands of Palmiet in the Middeldeur preclude fish capture using back-pack electroshocking or seine netting. Fish were, therefore, captured using fyke nets. Fish captured overnight in fyke nets cannot be used for gut content analysis. The extent of diet overlap between the Twee River redfin, Cape galaxias and Cape kurper was therefore carried out using stable-isotope analysis. The fish were captured using fyke nets, killed by severing the spinal cord, samples of muscle tissue extracted for stable light isotope analysis - 20 Cape galaxias, 10 Twee River redfin, and 40 Cape kurper (20 from the Middeldeur and 20 from the Suurvlei), and frozen in the field. Muscle tissue was freeze-dried and analysed for carbon 13 and nitrogen 15 isotopes in the Archaeology Department, UCT by combustion in a Thermo 1112 Elemental Analyser (Germany, Italy) interfaced via a Thermo Conflo II to a Thermo Delta XP Plus stable light isotope mass spectrometer. Results are reported relative to international standards.

A distance-based test for homogeneity of multivariate dispersion was performed to test whether there was a difference in multivariate dispersion of the isotope signatures of the fish species and populations using the PERMDISP routine. The PERMANOVA routine was used to determine whether there were statistically significant differences in the position of the centroids of the respective species/population isotope signatures. These tests were performed on a resemblance matrix generated from the raw data using Euclidian distance.

<u>Cape kurper age, length, and morphometrics</u>

During the fish survey, large numbers of Cape kurper were captured. Since this species is invasive in this catchment, most of the specimens captured were sacrificed. These fish were not suitable for diet evaluation because they had been captured using fyke nets, but a number of parameters pertaining to the Cape kurper population were measured. Length, gape and weight were recorded in the field and five scales from below the dorsal fin were removed for age estimation. In addition, a subsample of fish was used to record a range of morphometric parameters (Figure 5.5). Cape kurper are from the order Perciformes and have ctenoid scales suitable for use for ageing the individuals (Anderson and Neumann 1996). The scales were mounted between two microscope slides and photographed on a Nikon stereoscopic zoom microscope SMZ1500 with Nikon DS camera control unit DS-U2 and DS-5M camera head. The number of year rings on the scales were counted from the images and used to estimate age of the fish (Anderson and Neumann 1996). The median age was used as the estimated age of the individuals. Age was plotted against average length per age class to determine growth rate of the individuals using the von Bertalanffy growth model.

The length-weight relationship was determined by plotting the weight against the standard length and fitting a power curve to the data points (Anderson and Neumann 1996). Two dimensions of gape (horizontal and vertical) were measured using Vernier callipers. The vertical gape was plotted against standard length and a linear regression fitted to the data. The standard length, body depth and body width of ten Twee River redfin and ten Cape galaxias were measured using a Vernier calliper to determine whether they might besusceptible to being ingested by Cape kurper. A relationship between the size of redfin at risk of ingestion by Cape kurper aand the gape of the Cape kurper was then established.

A sub-sample of 26 Cape kurper were used to determine morphometric relationships within the species. Ten morphometric parameters were measures in the field. These were standard length, body depth, head length, head height, orbit diameter, snout length, lower jaw length, lower jaw width (Figure 5.5), body mass, and eviscerated mass. The morphometric measurements were plotted against standard length and the relationship between the respective morphometric measures and standard length explored.

5.3 Results

The fish distribution survey has been published in the journal *African Journal of Aquatic Sciences* (Marr *et al.* 2009), but has been expanded here to include additional data from subsequent surveys. The remainder of the results have not previously been published.

5.3.1 Fish Distributions

Maps showing the distribution of each species found in the Twee River system are presented in Figure 5.2. Distributions records from Marriott's 1996/7 survey are presented for comparison. Filled circles indicate sites at which the respective species were recorded, whereas open circles indicate sites where the respective species were not found. A total of six freshwater fish species were collected in the Twee River System: the endemic Twee River redfin (Figure 5.2a) and Twee River galaxias (Figure 5.2b); the translocated Cape kurper (Figure 5.2c) and Clanwilliam yellowfish (Figure 5.2d); and the introduced North American bluegill sunfish (Figure 5.2e) and smallmouth bass (Figure 5.2f). The North American rainbow trout, recorded in Marriott's 1996-7 survey, was not collected during the current survey and seems to have failed to establish in the Twee Catchment (Figure 5.2g).

Specimens of Twee River redfin were collected from the Hex, Middeldeur, Twee and Suurvlei rivers (Figure 5.2a). Specimens were collected only from the upper reaches of fish distribution in the Suurvlei and immediately below the Suikerbossie Bridge, where one gravid female was collected in October 2006, indicating that the species might use the lower Suurvlei for breeding. No redfin were found in the Buffelshoek River. A survey of the Hex tributary in November 2010 extended the known range of redfin in the Hex tributary. The Hex population appears to be safe at present. No juvenile redfin were recorded in areas invaded by Cape kurper while Cape galaxias, which are of a similar size to juvenile redfin, were captured. From the available evidence, redfin recruitment in reaches invaded by Cape kurper appears to be very limited. Further investigation is required to evaluate the recruitment of the redfin population in the Twee River catchment. In April 2011, more than 200 Twee River redfin were captured in a 50 m reach in the Middeldeur above the Cape kurper distribution, a large proportion of which were young-of-the-year.

Cape galaxias were present in large numbers in the upper Middeldeur and Hex Rivers, with isolated populations in the lower Middeldeur (Figure 5.2b). This species was not collected from the Suurvlei River or its tributaries.

Cape kurper were the most widespread and numerous fish species in the Twee River system (Figure 5.2c). Their range extended from the Kunje waterfall on the Middeldeur River, and from a raised pipe culvert on the Buffelshoek River, down into the Twee River. No Cape kurper were observed below Little Augrabies waterfall, where smallmouth bass were present.

Clanwilliam yellowfish were present in large numbers (adults and juveniles) below the Die Drift waterfall and downstream towards the confluence with the Leeu River (Figure 5.2d). One individual was collected above Die Drift waterfall in April 2007. Marriott found yellowfish above Die Drift waterfall, but at a low density. Below Die Drift yellowfish dominated the fish assemblages except below Little Augrabies waterfall where smallmouth bass were present.

Bluegill sunfish were only collected in the lower Middeldeur, in the Twee just below the confluence of the Middeldeur and Suurvlei Rivers, and in the lower Twee River below the Little Augrabies waterfall (Figure 5.2e). In December 2008, a number of bluegill were found between the De Straat and Kunje waterfalls above the Cape kurper distribution. Subsequent surveys did not find bluegill in this reach and invasion of this reach appears to be transient. The source of bluegill in the Twee system was not known until two farm dams on a tributary of the Middeldeur, upstream of the De Straat waterfall, were found to hold large populations of bluegill. These dams, on the farm Tandfontein, pose a threat to the entire upper Middeldeur, an area identified by Marr *et al.* (2009) as an important sanctuary area for the Twee river endemic fishes.

Smallmouth bass were collected only below Little Augrabies waterfall in the Twee River system (Figure 5.2f). In November 2010, bass (*Micropterus spp.*) were recorded in the uppermost of the bluegill dams on the farm Tandfontein. The farm manager confirmed that bass had been introduced in 2010 and offered to reduce the level in the dam to 40% of capacity such that the bass can be removed.

5.3.2 Water Quality

Significant differences were found between the Suurvlei and Middeldeur for all water chemistry parameters evaluated (t-test, $p < 0.05$). The water in the Suurvlei has a higher pH, lower conductivity, higher dissolved oxygen, higher temperature and lower turbidity than the Middeldeur (Table 5.2). The PCA showed that the first two principal components (PCs) accounted for 75% of total variation between the tributaries. All the water chemistry variables included contributed to the differences between the tributaries (Figure 5.6). The PERDISP result for multivariate dispersion of the water chemistry data for the two tributaries was not significant ($p = 0.09$). The PERMANOVA routine returned a significant result ($p < 0.001$) indicating that there was a significant difference in the position of the centroids for the respective tributaries. The nutrient concentrations on the two tributaries was only significantly different for nitrite (Table 5.2: t-test, $p = 0.02$). The PERMDISP and PERMANOVA analyses both returned non-significant results for the nutrient data indicating that neither the dispersion about the respective centroids, nor the position of the respective centroids, were significantly different between the two tributaries.

The assessment of water quality using the SASS 5 scores for macro-invertebrates present at the sampling sites showed the three sites (SUUR01, MID02, and MID03) could be considered as reference sites, or better (Table 5.3). The remaining sites in the Suurvlei River were both below reference condition, with the SUUR02 site well below reference condition (Table 5.3). These results indicate that there is a drop in water quality between the confluence of the Suurvlei and Buffelshoek Rivers and a point below the intensive fruit orchards of the farm Tuinskloof. This was not evident from the water chemistry and nutrient data. The NMDS plot of macro-invertebrate data shows a clear separation between the sites in the two tributaries (Figure 5.7). Sites SUUR02 and SUUR03 appear to have similar invertebrate assemblages as the data groups for these sites overlap on the NMDS plot. This is confirmed by the average similarity between the SUUR02 and SUUR03 sites being similar to that within the SUUR02 and SUUR03 sites (Table 5.4). The PERDISP result for multivariate dispersion of the aquatic invertebrate data for the two tributaries was not significant (p = 0.16). The PERMANOVA routine returned a significant result ($p < 0.001$) indicating that there was a significant difference in the position of the centroids for the respective tributaries. When evaluated at the site level, there was no significant difference in dispersion (PERMDISP, p = 0.25) but a significant difference in the group centroids for each site (PERMANOVA, $p < 0.001$). A pair-wise PERMANOVA between the sites was performed to determine which sites contributed to the significant difference in the position of the group centroids. A significant result ($p < 0.001$) was found for all pair-wise comparisons with the exception of the SUUR02-SUUR03 comparison (p = 0.06), thereby supporting the inference from the NMDS plot and average site similarity (Figure 5.7 and Table 5.4).

The SIMPER analysis showed that nine taxa contribute to the differences between the Suurvlei and Middeldeur Rivers: Baetidae, Hydropsychidae, Leptoceridae, Chironomidae, Simulidae, Elmidae, Hydraenidae, Amphipoda, and Oligochaeta (Table 5.5).

5.3.3 Instream and Riparian Habitat

The survey of instream and riparian habitat was conducted during the period of low flow just prior to the winter rainfall season. The data collected for each site are presented in Appendix A.5. The two tributaries were shown to have distinctly different characteristics for both instream and riparian zone habitats. The PCA showed that the PC contributing most to the differences in instream habitat between the two tributaries was PC2. The first PC accounted for the differences within each tributary (21% of total variation), particularly for the Middeldeur (Figure 5.8a). The second PC contributed to the differences between the two tributaries (20% of the total variation). The major variables contributing to the differences are the presence of palmiet, average depth, stream width, flow and proportion of open water. The sites in the Suurvlei had a higher flow, shallower average depth, higher proportion of open water, and narrower stream width than the sites in the Middeldeur. The habitat in the Middeldeur was dominated by deep (>1m depth) pools with extensive areas of palmiet. Although a number of pools are present in the Suurvlei, they are less common, less extensive, shallower, narrower and do not have extensive beds of palmiet. The instream habitat in the Suurvlei is less variable than that of the Middeldeur (Figure 5.8a). The water velocity in the Suurvlei was mostly moderate visual aberration. This could be largely the result of the shallow water depth in this tributary rather than as a result of higher flow. The Middeldeur is the larger of the two tributaries and is expected to contribute a higher volumetric flow than the Suurvlei.

For the riparian zone, the PCA showed that the PC contributing most to the differences between the two tributaries was PC1, accounted for more than 50% of total variation between the tributaries (Figure 5.8b). The major variables contributing to differences between the tributaries are the presence of natural vegetation in the Middeldeur, whereas the Suurvlei is defined by the presence of orchards, alien trees, felled alien trees, recovering natural vegetation and anthropogenic impacts. The riparian zone in the Middeldeur is less variable than that of the Suurvlei (Figure 5.8b). Two sites in each tributary were outliers and were defined by the second PC. The Middeldeur outlier sites had been severely burned in 2009 and showed no signs of recovery, whereas the Suurvlei sites had been bulldozed (channel straightening using a bulldozer or back-actor). Although there are extensive orchards on the banks of the Middeldeur, these are mostly more than 20 m away from the river because the bank consists of a series of river terraces.

5.3.4 Pesticides

Samples of brain tissue were obtained from 34 Cape kurper at three sites in the Middeldeur and one site in the Suurvlei in November 2011. The acetylcholine esterase activity was determined within two weeks of the samples being taken. The ANOVA within sites returned a significant result ($p < 0.05$) for sites M1, M3, and S1, and a result of $p = 0.09$ for site M2. This indicates that the variation between the samples at each site is greater than the variation within the samples. The ANOVA between the sites returned a significant result ($p < 0.001$), but the Tukey honest significant difference test indicated that site M1 was the only site contributing to the significant ($p < 0.001$). The results indicate that there is no difference in pesticide exposure between sites in the Suurvlei, where redfin are absent, and in the Middeldeur, where redfin are present. However, anomalies in the weights used during the laboratory analyses were detected and the validity of the results is questioned. The pilot study should, therefore be repeated to verify the results. A long-term study to evaluate seasonal variations in pesticide exposure could then be planned. The average acetylcholine esterase activity rates for each site are presented in Table 5.6.

5.3.5 Interactions Between Native and Non-Native Fish

The stable light isotope signatures of the respective species present in the Middeldeur and Suurvlei Rivers are summarised in Figure 5.9. There are no significant differences in the dispersion between the three species (PERMDISP p = 0.47), but there are significant differences between the location of the centroids for the respective species (PERMANOVA p < 0.001). There was also a significant difference in the position of the group centroids for the two populations of Cape kurper. The difference in δN^{15} is between the galaxias and the two Cape kurper populations is greater than 3.4 $^{0}/_{00}$, providing evidence that galaxias could form a part of the diet of Cape kurper. No ontogenetic shifts in stable isotope signatures were evident for all species. Interestingly, the Cape kurper in the Suurvlei River recorded a higher δN^{15} value than the population on the Middeldeur. This could be related to the large number of predatory invertebrates in the Suurvlei River in comparison to those in the Middeldeur. It is possible that Cape kurper in the Middeldeur are preying on redfin, but they would form only a small portion of the diet.

Scales from 215 Cape kurper were used to determine the length at age relationship for the Twee River population. A von Bertalanffy relationship between age and mean length at age was found (**SL** = 115.7673*(1-exp(-0.2001*(**Age**+1.9451))), Figure 5.10). The length-weight relationship based on 316 Cape kurper was a power relationship: **Weight** = 3.38 x 10^{-5} ***SL**$^{2.956}$, adjusted R^2 = 0.987, p < 0.0001 (Figure 5.10).

There was a linear relationship between standard length and each of the eight morphometric parameters measured for a sub-sample of 26 Cape kurper (Figure 5.11). All linear relationships were statistically significant (p < 0.0001) and are summarised in Table 5.7. The relationship between gape and standard length for Cape kurper was **Gape** = 0.0844 ***SL**+0.3941, adjusted R^2 = 0.970, p < 0.0001, n = 175. That for the body depth and standard length for redfin was **Effective Body Depth** = 0.176 ***SL**+2.605, adjusted R^2 = 0.889, p < 0.0001, n = 10. Note, the body depth was increased by 10% to represent an effective body depth of a prey item being ingested by Cape kurper. A relationship between the effective body depth of redfin and the gape limitation of Cape kurper was established the risk of redfin predation by Cape kurper (Figure 5.12). The results indicate that only redfin smaller than 50 mm are at risk of predation by large Cape kurper.

5.4 Discussion

This study of the Twee River catchment is a compilation of a number of smaller studies aimed at determining what factors might be preventing the re-establishment of the Twee River redfin in the Suurvlei River. Each study would benefit from being extended over several seasons, or years, but that was not possible due to funding and time limitations. Although further work is required to fully identify the factors preventing the re-establishment of Twee river redfin in the Suurvlei, some conclusions can be drawn.

5.4.1 Fish Distributions

The distribution of the two species endemic to the Twee River system appears not to have changed over the ten years since Marriott's 1996/8 survey, although both appear to have become less abundant and more localized in the lower Middeldeur and upper Twee Rivers.

<u>Endemic Species</u>

Both species appear to have a sanctuary above the Kunje Waterfall in the upper Middeldeur River. While galaxias were frequently detected in large numbers in this area, numbers of redfin were small except in the reaches between the De Straat and Kunje Waterfalls and in the Hex tributary. This area is clearly an important conservation area for the two endemic species, and it is recommended that this area should be developed into a sanctuary as part of a conservation plan for the Twee River endemics.

In the lower Middeldeur and Twee Rivers mature adult redfin of 70-80 mm were frequently captured in the fyke nets, albeit in low numbers. No juveniles were captured in these reaches and it is not clear if redfin in these reaches are recruiting. Juveniles were collected in the pool between the two sections of the Middle Waterfall in the lower Twee River, where Clanwilliam yellowfish dominate the fish assemblage. Redfin were introduced into a dam in the Suurvlei catchment as a conservation measure by CapeNature, but the persistence of this population has not been confirmed. Redfin were also introduced to a 100 m reach in the Suurvlei River above their natural range by CapeNature and have established a recruiting population in this reach.

Galaxias appear to be locally common at specific sites in the lower Middeldeur, but were detected at fewer sites than during the 1996/7 survey. Further surveys are required to confirm the distribution and size of the galaxias populations in the lower Middeldeur and upper Twee and to establish whether these populations are recruiting. No conservation actions have been initiated for the galaxias. The establishment of sanctuary populations in the Buffelshoek tributary and farm dams could be considered as an initial conservation initiative for galaxias.

Introduced Species

The Cape kurper appears to have reached the full extent of its invasion. The population is currently bounded by the Kunje Waterfall on the Middeldeur River, a pipe culvert on the Buffelshoek River, and a bedrock step on the Suurvlei River. Cape kurper may extend their range downstream into the Leeu River, but the presence of smallmouth bass below the Little Augrabies waterfall will prevent their establishment in the Doring system (Impson *et al.* 2007). Cape kurper are rare in the pools, where Clanwilliam yellowfish dominate the fish assemblage.

Bluegill sunfish do not appear to have established recruiting populations in the Middeldeur and Twee Rivers. Bluegill have proven to be invasive elsewhere in the world (Marchetti *et al.* 2004c), South Africa (de Moor and Bruton 1988), and in the Cape Floristic Region (Impson *et al.* 2002b) and could become problematic should they establish a recruiting population. At present bluegill numbers are low, probably being outcompeted by the large population of Cape kurper present in the system. The establishment of a large population of bluegill in the Twee system could lead to the extirpation of the endemic species due to predation on eggs, larvae and juveniles, as found in California by Marchetti (1999) for the Sacramento perch *Archoplites interruptus* (Girard, 1854).

The current survey identified the source of bluegill sunfish in the Twee River system as two large dams on the farm Tandfontein. These dams are situated on a tributary of the Middeldeur in the area proposed above as a sanctuary for the Twee River endemics. Bluegill appear to be periodically flushed from the dams into the Middeldeur. Reducing the risk of future introductions of bluegill from these dams could lead to the extirpation of bluegill from the Twee River system.

Only one Clanwilliam yellowfish was captured above Die Drift waterfall during the surveys. Below Die Drift waterfall, yellowfish have formed a large, self-sustaining population that is currently a source population for yellowfish in the remainder of the Doring system. Above Die Drift waterfall, it appears that Clanwilliam yellowfish are very rare and it is likely that they have not formed a naturalised population in these reaches. The presence of yellowfish in the reaches below Die Drift waterfall is a matter of concern since these reaches contain what is considered prime redfin habitat, with the highest population densities of redfin being observed in these reaches during Marriott's, and subsequent, surveys (R. Bills, SAIAB, pers. comm.). It has been suggested that yellowfish out-compete, and possibly prey on, redfin in this area (R. Bills, SAIAB, pers. comm.), although this hypothesis is untested.

Rainbow trout were not detected and appear to have failed to form a naturalised population in the Twee River. Trout were introduced illegally below Die Drift waterfall in an area which does not contain suitable breeding habitat and is sub-optimal for trout (ND Impson, CapeNature, 2009, pers. comm.). The absence of suitable breeding habitat is considered the primary reason for the failure of trout to establish in the Twee River.

<u>A note on the species of bass in the lower Twee River</u>

Bass (*Micropterus* spp.) collected in the lower Twee River were difficult to identify because some specimens showed the characteristics of both smallmouth and spotted bass. Smallmouth bass *Micropterus dolomieu* and spotted bass *Micropterus punctulatus* (Rafinesque, 1819) co-occur in the lower Twee, where the two species appear to be hybridising, although this has not yet been confirmed by genetic analysis. Hybrids between smallmouth and spotted bass have been recorded in the USA (Koppelman 1994, Cofer 1995) and it has been suspected that these species have hybridised elsewhere in the Olifants–Doring system (Bills 1999). Bass were found downstream of the natural distribution range of the Twee River redfin. Bass, of a species yet to be confirmed, were introduced into the upper of the two dams on the farm Tandfontein in 2010.

5.4.2 Instream and Riparian Habitat

It is clear that the instream and riparian zone habitats in the Suurvlei and Middeldeur are very different. The habitat available in the Middeldeur is complex and consists of a series of deep pools fringed with palmiet. The palmiet encroaches on the main channel, possibly as a result of reduced flows from excessive water abstraction. In areas, palmiet has encroached to form a bank-to-bank bed. The submerged stems, about 100 mm in diameter, provide adequate cover for fish to avoid predation.

A number of changes have occurred in the Suurvlei catchment, including increase in area under orchards, increase in production from these orchards, rupture of the dam that has resulted in the slug of sand moving down the river, and degradation of the riparian zone. Many of the areas where large scale bank works have taken place, such as removal of grey poplars using a back-actor or bulldozing to remove the sand slug, appear to be recovering, with natural riparian and instream vegetation re-establishing. As clearing of alien vegetation is completed, and a natural riparian zone re-established, the instream habitat in the Suurvlei could improve and some of the deep pools reported by long-term residents could be cleared of sand.

There are, currently, a number of pools close to the Suurvlei population of redfin that might constitute suitable habitat for redfin, but they have not being colonised. The present understanding of what constitutes adequate redfin habitat is incomplete. Although the stream gradient of the Hex River is greater than that of the Suurvlei, redfin are present in pools of similar depth and size to those found the Suurvlei. Habitat suitability criteria could be developed for both Twee River endemic species, based on their occurrence in the upper Middeldeur and Hex Rivers, using standard methods (Bovee 1986, Heggenes 1991, 1996, Bovee *et al.* 1998). Riparian zone management programmes can then be instituted to promote the natural generation of suitable habitat in the Suurvlei River. At present, there does not appear to be a large amount of suitable habitat for Twee River redfin in the Suurvlei, but in time, with continued riparian zone rehabilitation, it is possible that suitable habitat could be re-created for the Suurvlei. What remains evident is that redfin can co-occur with Cape kurper in the complex habitat and deep pools of the Middeldeur, but not in the simple shallow habitat of the Suurvlei.

5.4.3 Water Quality

The analysis of water chemistry and nutrients appear to indicate that the water quality in the Suurvlei is slightly better than that that of the Middeldeur. However, the macro-invertebrate assemblages in the Suurvlei, particularly at sites SUUR 02 and SUUR 03, indicate that factors impacting the Suurvlei were not detected through water chemistry and nutrient analysis. Neither river is limited in the aquatic invertebrate food available. The loss of sensitive invertebrate species at sites in the Suurvlei suggests that the presence of orchards alongside the river, and bank destabilisation, are having an impact on the aquatic assemblage not detected in the water chemistry. Longer-term monitoring is required to detect the source of these differences. At present it would appear the water quality is not limiting the re-establishment of redfin in the Suurvlei.

5.4.4 Interactions Between Native and Non-Native Fish

Both Twee River redfin and Cape kurper are primarily opportunistic insectivores with similar diet composition, although Cape kurper show evidence of piscivory (Marriott 1998). The stable isotope analysis confirmed similarities in diet of redfin and Cape kurper. There was no clear evidence of predation on redfin. The gape analysis indicates that large Cape kurper are capable of ingesting redfin of up to about 50 mm. The absence of smaller redfin in the areas where redfin and Cape kurper co-occur may be attributable to predation on juvenile redfin, although no evidence to support this hypothesis has been collected. The greatest intensity of predation of redfin would have occurred shortly after colonisation by Cape kurper, and the probability of detecting predation when the redfin may form only a small portion of the Cape kurper diet, even when using stable isotopes, is low. Low predation rates on eggs, larvae or juveniles may still have a large impact on the native species if it has a relatively low fecundity, or the predator species is abundant (Meffe 1985).

Cape kurper are known to be aggressive (Cambray 2004) and it is suspected that they are preying on galaxias and the juvenile stages of redfin. The stable isotope evidence suggests that redfin and Cape kurper compete for food resources and that the kurper is the dominant competitor. In captivity, Cape kurper have eaten or fatally harassed all other fish in aquaria, even other Cape kurper (pers. obs.). It is important that the interactions between the Twee River endemic species and Cape kurper be documented, although such experiments may be ethically questionable, as the outcome can be predicted with confidence. The sacrifice of either endemic species in laboratory or field cage experiments cannot be justified considering their current conservation status.

5.4.5 Concluding Remarks

While this study has not conclusively shown that Cape kurper are preventing redfin re-colonising the Suurvlei River, or reducing recruitment of redfin in the Middeldeur and Twee Rivers, there is evidence supporting these conclusions. The conclusion that redfin are not able to co-occur with Cape kurper in the absence of the complex habitat provided by palmiet is supported. It is possible to test whether redfin would survive in the Suurvlei using appropriately sized enclosures, or by removing Cape kurper from the Tuinskloof reaches of the Suurvlei and Buffelshoek Rivers. These reaches are bordered by the highest intensity of fruit farming in the Twee catchment and show a gradient of water quality degradation (SUUR01 through SUUR03). If redfin can survive and establish a recruiting population in these reaches in the absence of Cape kurper, it can be concluded that Cape kurper were preventing the re-establishment. If the redfin fail to establish a recruiting population in the absence of Cape kurper, the inclusion of the Suurvlei River in the GEF Alien Fish Eradication Project should be questioned. Further work would be required to identify the factor preventing the re-establishment of redfin in the Suurvlei River.

5.5 Management of Non-Native Fish in the Twee River

The priority non-native fish management action is the removal of bass and bluegill from the dams on the farm Tandfontein. The owners of the farm have shown a desire to conserve nature on their property, and the farm manager has offered to reduce the level of the dams so that they can be treated with a piscicide for the removal of non-native fish. However, the risk of re-introduction of bluegill, and in particular bass, needs to be prevented. The two dams receive water from the Leeu River, which holds populations of bass and bluegill. The dams only spill during the winter rainfall season and their water levels could be managed to reduce the risk of flushing fish into the Middeldeur. The spill-over of bluegill into the Middeldeur could be further reduced by frequently reducing, or removing, the population of bluegill in the dams. Alternatively, the lower dam could be held fishless to buffer any spill-over of fish from the upper dam. Removing the stochastic introduction of bluegill into the Middeldeur from the Tandfontein dams could result in the extirpation of bluegill from the Middeldeur and Twee. All other off-stream populations of bluegill in the Twee catchment would need to be removed to reduce the risk of future introduction.

Bluegill appear to have established in the Tandfontein dams through the water transfer network. Bass, however, were intentionally introduced with the knowledge of the farm manager. Since bass, particularly smallmouth bass, are a greater threat to the endemic species of the Twee River than bluegill, preventing the re-introduction of bass once they have been removed is an essential long-term conservation project. In the Twee below Little Augrabies waterfall, the presence of smallmouth bass is preventing Cape kurper establishing in the Leeu River. Cape kurper have been extirpated by bass wherever bass have been introduced into the Cape kurper's natural range.

Following the removal of bass and bluegill from the Tandfontein dams, managing the Cape kurper population is the second priority intervention. The Suurvlei has been included in the GEF Alien Fish Eradication Pilot Project and the removal of Cape kurper from this tributary would be the first logical step in the management of Cape kurper. However, the questions raised in this study need to be addressed before the implementation of the Suurvlei treatment. Cage experiments, or preferably removal of Cape kurper from the upper portion of the proposed intervention area, should be considered to confirm that redfin will survive and re-establish once the Cape kurper have been removed. Once it has been confirmed that redfin will re-establish in the Suurvlei, the GEF Alien Fish Eradication Project intervention proposed for this river can be implemented.

Removal of Cape kurper from the Suurvlei will be considerably easier than removing the species from the Middeldeur and Twee above Die Drift waterfall. Two difficulties arise: the depth of the pools and the encroachment of palmiet. The Middeldeur and Twee consists largely of deep pools connected with short, shallow sections. Some pools are in excess of five metres deep. Palmiet is present in these reaches and severely encroaches on the river channel. In some areas the river channel disappears below dense stands of palmiet. Removal of Cape kurper from these reaches would be difficult, even using a piscicide. Piscicides are probably the only viable consideration here, although dosage rates would need to be high due to the high organic content of the palmiet stems resulting in the increased rate of degrading of rotenone.

The Clanwilliam yellowfish appear to be in low abundance in the reaches above Die Drift waterfall, possibly indicating that the population is in decline due to the absence of juvenile yellowfish. Any yellowfish captured in these reaches should be removed. The population below Die Drift waterfall is a matter of contention among fish researchers. The researchers at SAIAB hold that the population originated from the Clanwilliam hatchery and are of dubious genetic origin and therefore should be removed (ER Swartz and R Bills, SAIAB, pers. comm.). CapeNature retorts that the population is a source population for the Doring River and is believed to be the origin of many of the yellowfish currently in the Doring River (ND Impson, CapeNature, pers. comm.). The population could be used as a source for stocking of dams for recreational angling. The Twee River yellowfish population is one of the remaining recruiting yellowfish populations in the Olifants-Doring system and is likely an important contributor to the yellowfish population in the Leeu-Matjies-Groot sub-catchments of the Doring River. Whether the SAIAB researchers approve, the Twee yellowfish are currently in the Doring system and could potentially be managed as an important source population in the system. The population could be periodically reduced by moving fish to other locations, or downstream below the Little Augrabies waterfall. In fact, there is no reason why the Twee River below Die Drift waterfall could not be used as a breeding area for the two endangered large cyprinids of the Olifants-Doring system: sawfin *Barbus serra* Peters, 1864 and sandfish *Labeo seeberi* Gilchrist & Thompson, 1911. This can be considered when the long-term persistence of the Twee River redfin and galaxias have been secured.

It is suspected that rainbow trout never established in the Twee River and are thus do not require further management. It is unlikely that stocking of rainbow trout would be legally permitted into the Middeldeur or Suurvlei Rivers, but education of land owners to prevent illegal stocking is a necessary precautionary step.

Figures

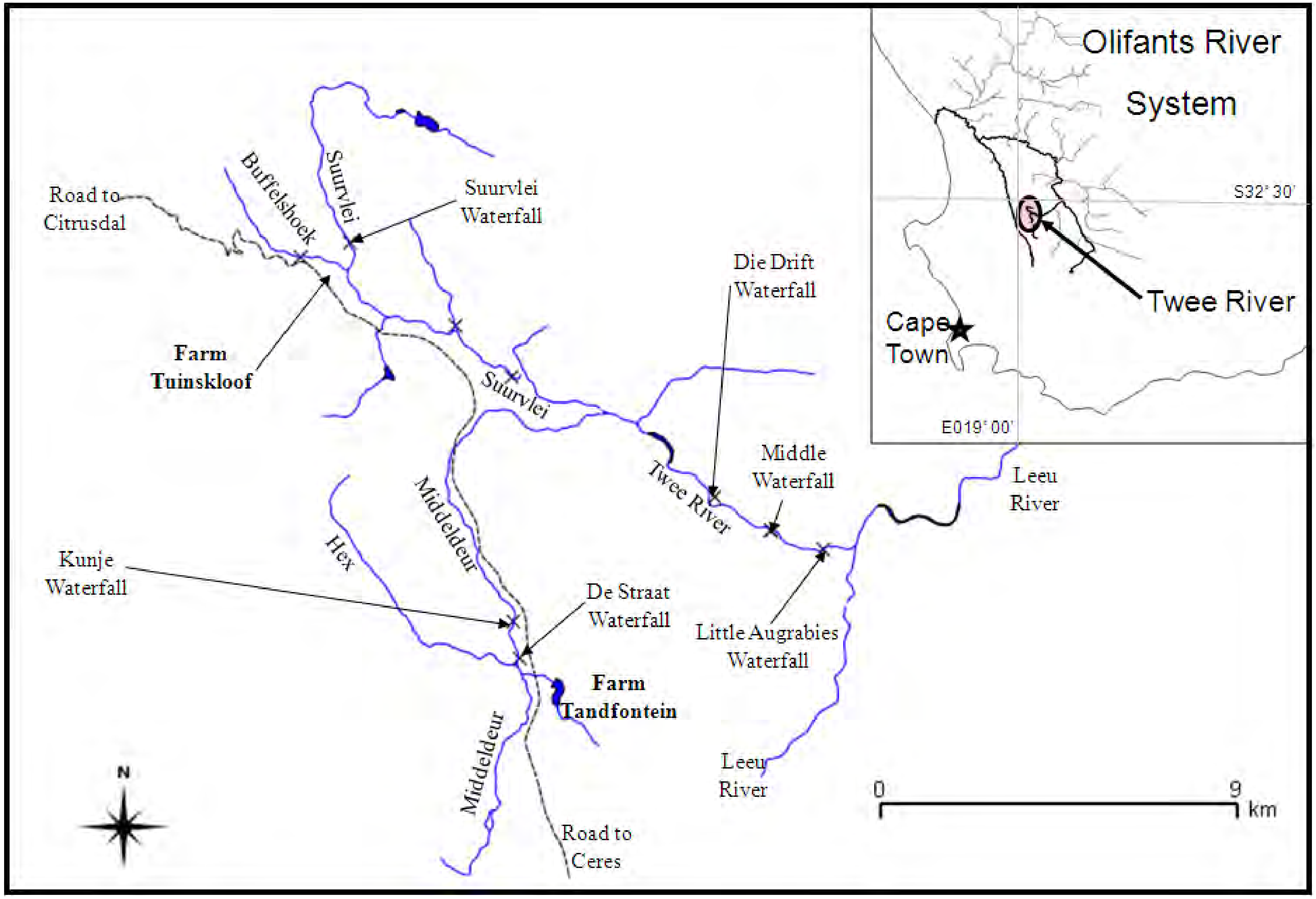


Figure 5.1: Twee River catchment showing the tributaries of the Twee River and the location of important waterfalls, roads and landmarks.

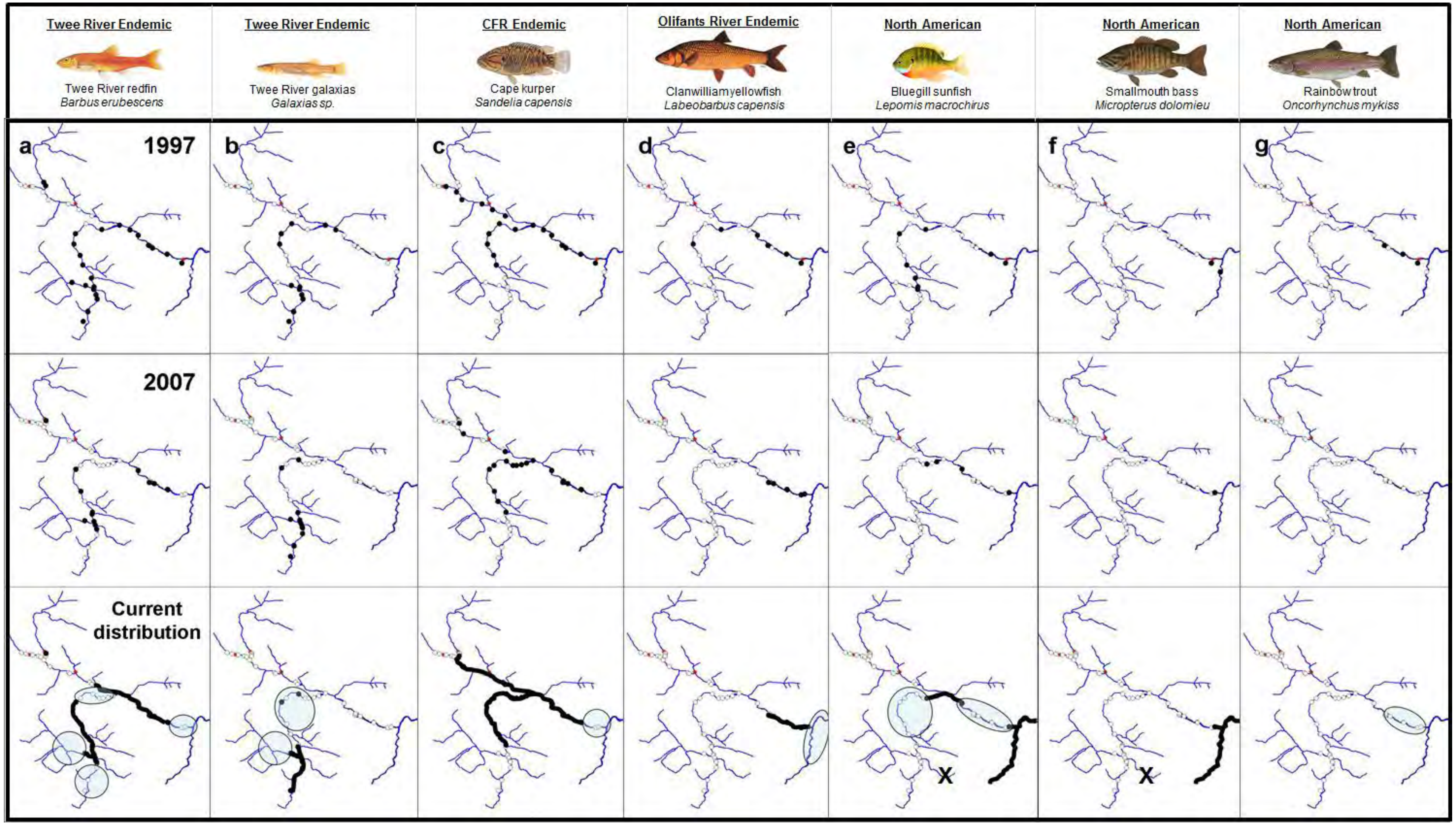


Figure 5.2 a-g: Site records from Marriott's (1997) and current surveys, and current distribution as inferred from the current survey for the endemic Twee River redfin (*Barbus erubescens*) (a) and Twee River Galaxias (*Galaxias zebartus*) (b), and introduced species Cape kurper (*Sandelia capensis*) (c); Clanwilliam yellowfish (*Labeobarbus capensis*) (d); bluegill sunfish (*Lepomis macrochirus*) (e); smallmouth bass (*Micropterus dolomieu*) (f); and rainbow trout (*Oncorhynchus mykiss*) (g). Areas circled indicate river reaches requiring further surveys. Black dots indicate sites where the species was captured. Open dots indicate that the species was not captured at the site. The X in the bluegill and bass maps indicates the location of the dams on the farm Tandfontein.

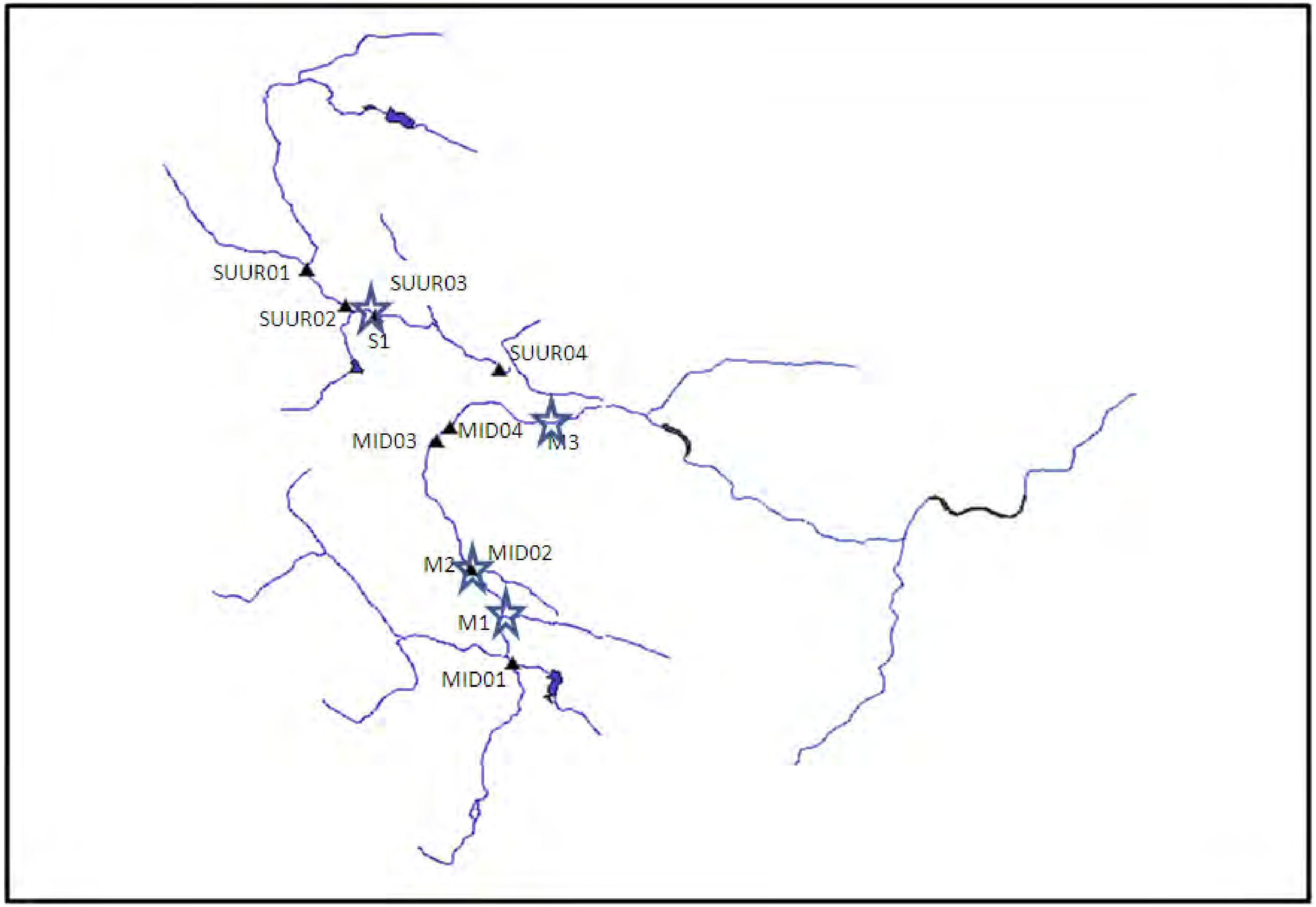


Figure 5.3: Location of the water chemistry (▲) and pesticide exposure (star) sampling sites in the Suurvlei and Middeldeur tributaries of the Twee River system.

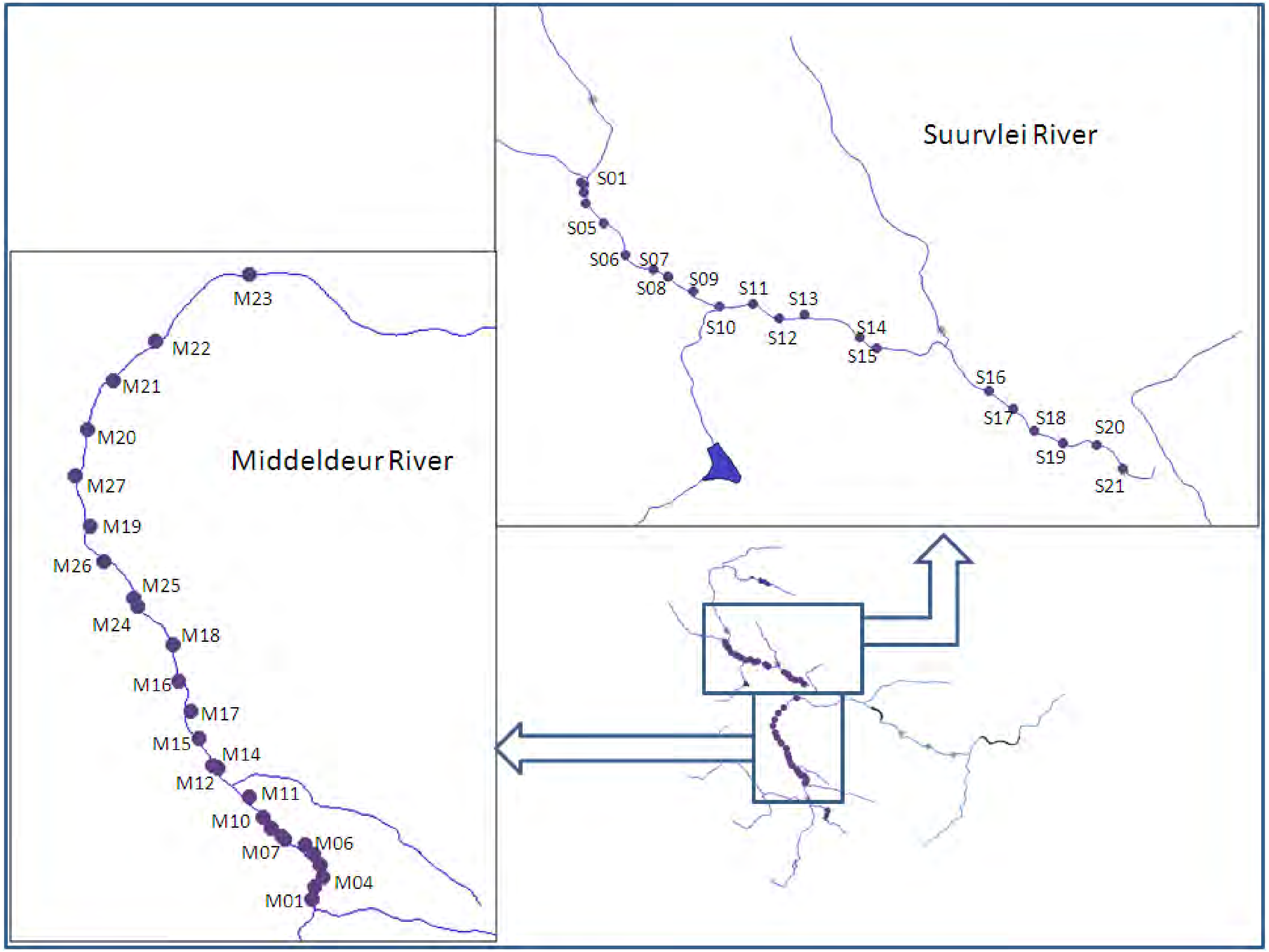


Figure 5.4: Location of the instream habitat sampling sites in the Suurvlei and Middeldeur tributaries of the Twee River system. See Appendix A.5.6 (included on the CD) for the data for each transect.

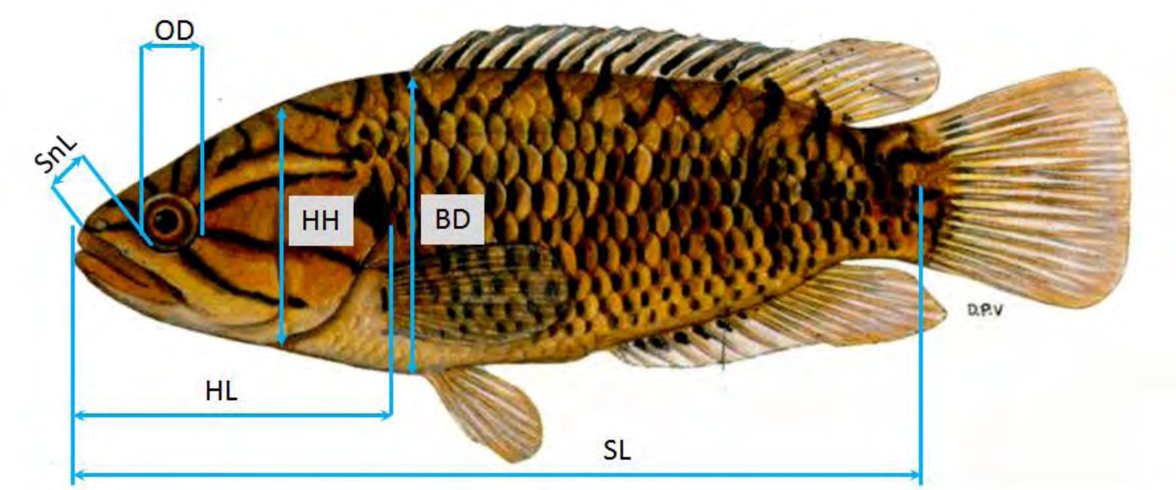


Figure 5.5: Morphometric measurements recorded for Cape kurper *Sandelia capensis* from the Twee River catchment: OD – orbit diameter; BD – body depth; HH – head height; HL – head length; SL – standard length; and SnL – snout length. (LJL – lower jaw length and LJW – lower jaw width not shown).

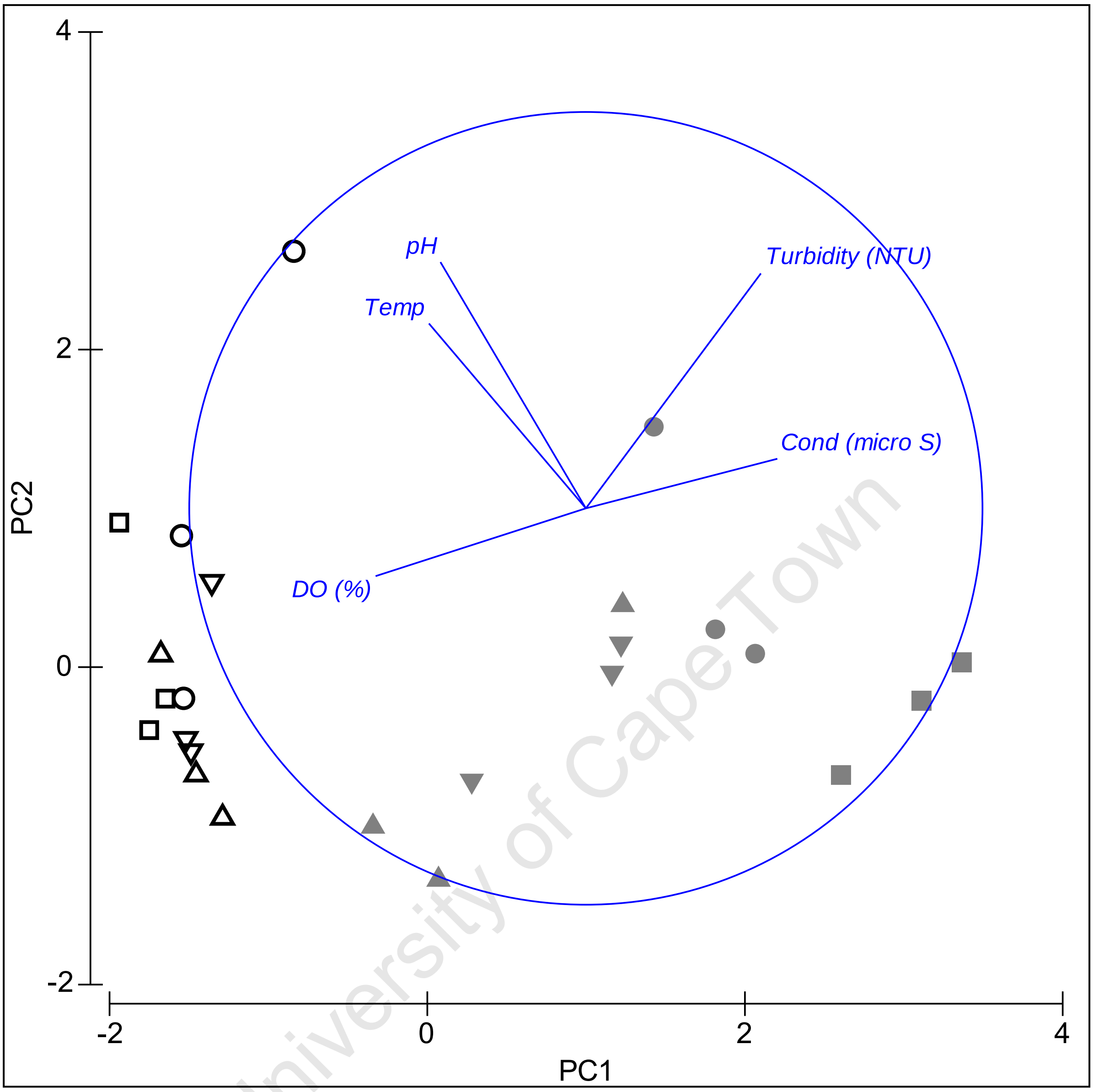


Figure 5.6: Principal Component Analysis of the water chemistry data for the Suurvlei and Middeldeur tributaries of the Twee River system. The sites in the Suurvlei are represented by open symbols (SUUR01 Δ, SUUR02 , SUUR03 □, SUUR04○) and those in the Middeldeur by solid triangles (MID01 ▲, MID02 ▼, MID03 ■, MID04 ●). The major variables contributing to the first two principal components are overlain as vectors in the PCA plot.

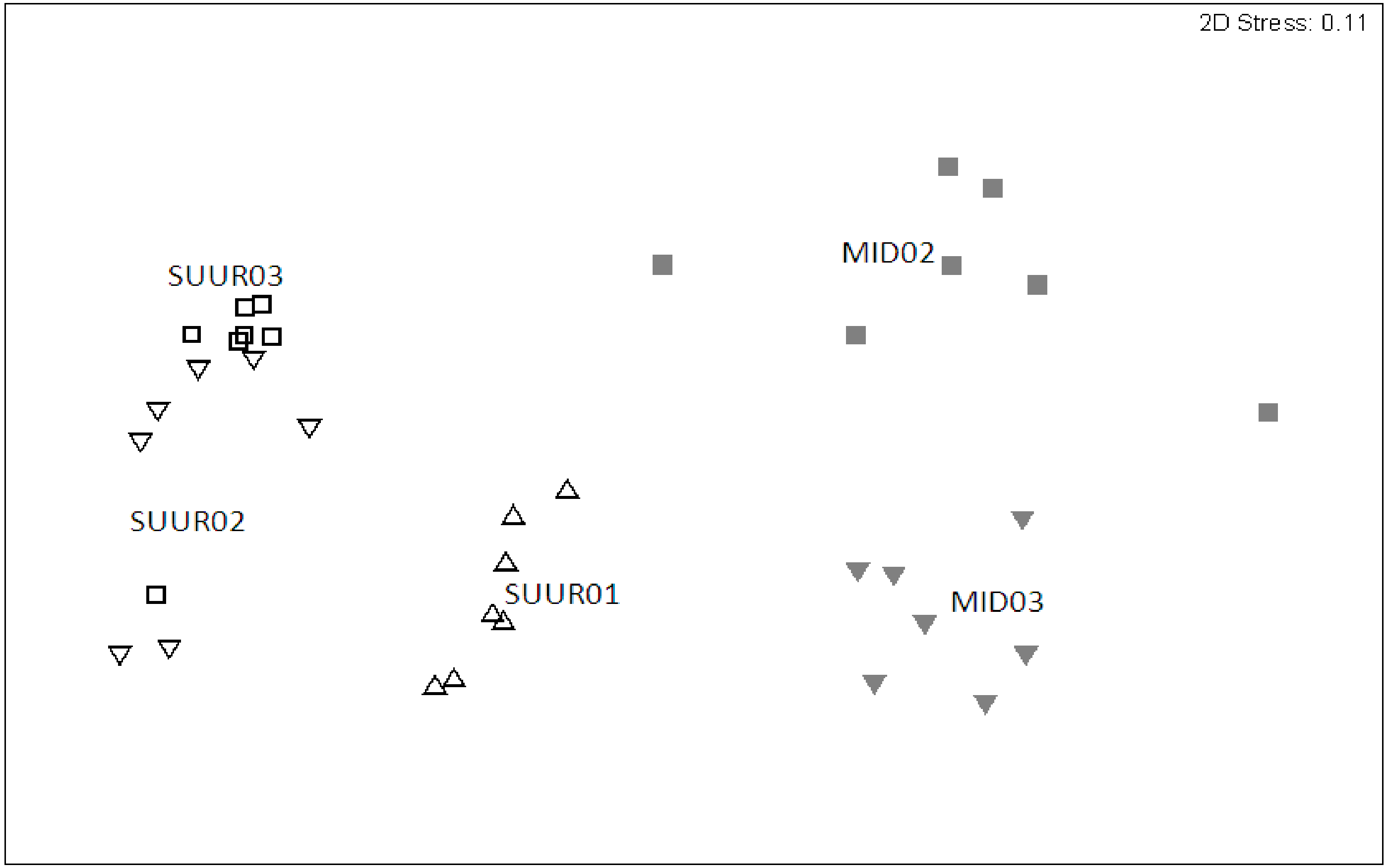


Figure 5.7: Non-Metric Dimensional Scaling plot of the macro-invertebrate families present at the water quality sites in the Suurvlei and Middeldeur tributaries of the Twee River system. The sites in the Suurvlei are represented by open symbols (SUUR01 Δ, SUUR02 , SUUR03 □) and those in the Middeldeur by solid triangles (MID02 ▼, MID03 ■).

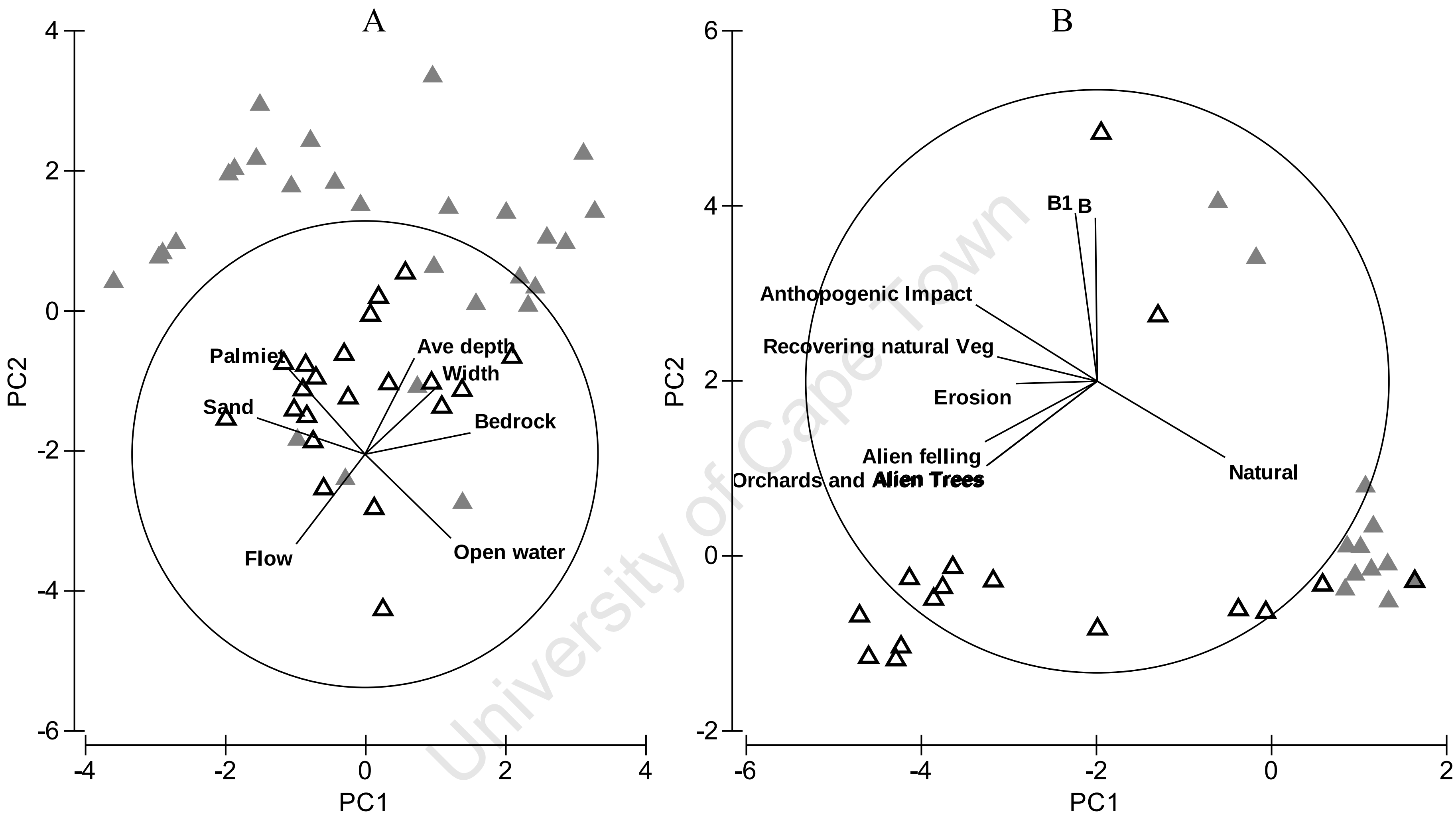


Figure 5.8: Principal Component Analysis of the instream habitat (A) and riparian habitat (B) for the Suurvlei and Middeldeur tributaries of the Twee River system. The sites in the Suurvlei are represented by open triangles (Δ) and the sites in the Middeldeur by solid triangles (▲). The major variables contributing to the first two principal components are overlain as vectors in the PCA plot. In the riparian habitat plot, B represents burned sites and B1 represents bulldozed sites.

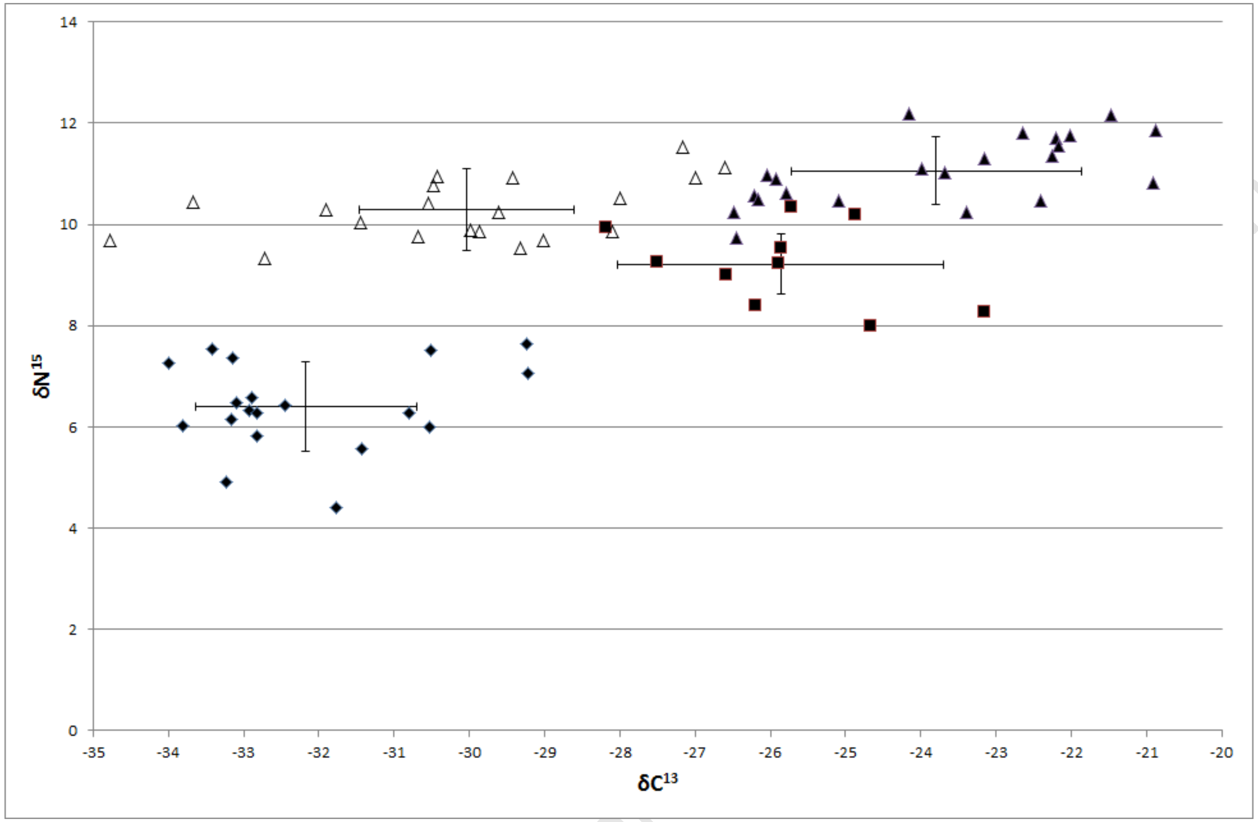


Figure 5.9: Stable light isotope signatures for Twee River redfin (■), Cape galaxias (♦) and Cape kurper (Middeldeur (Δ) and Suurvlei (▲) populations) from the Twee River system. The centroid for each species/population is plotted with error bars of one standard deviation.

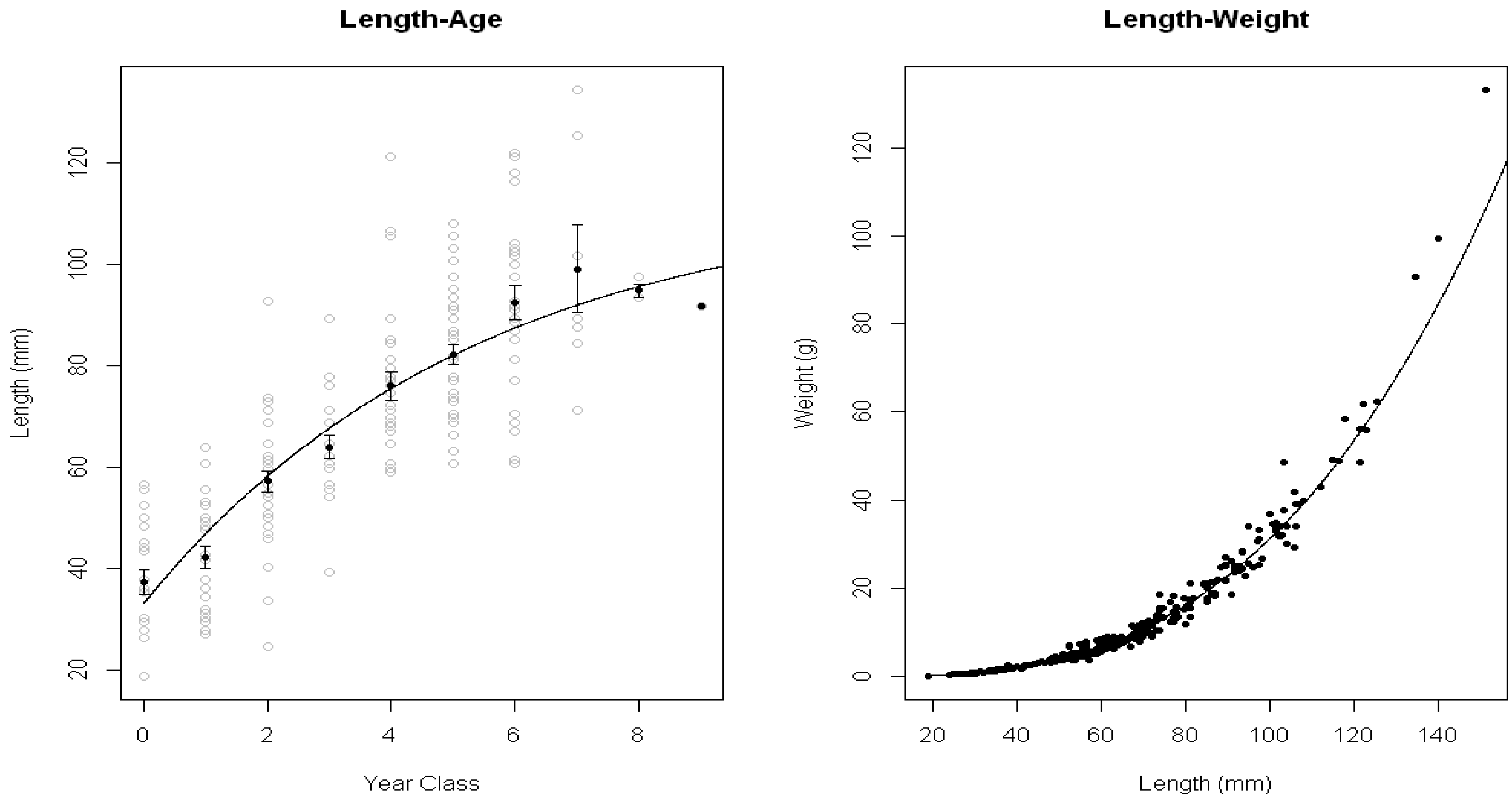


Figure 5.10: Age Length and Length-Weight relationships for Cape kurper in the Twee River system. A total of 316 individuals were measured for the length weight analysis and 215 for the age length analysis. Standard length was used for both analyses. The average length per age class was used for the Age-Length analysis and fitted to the von Bertalanffy model.

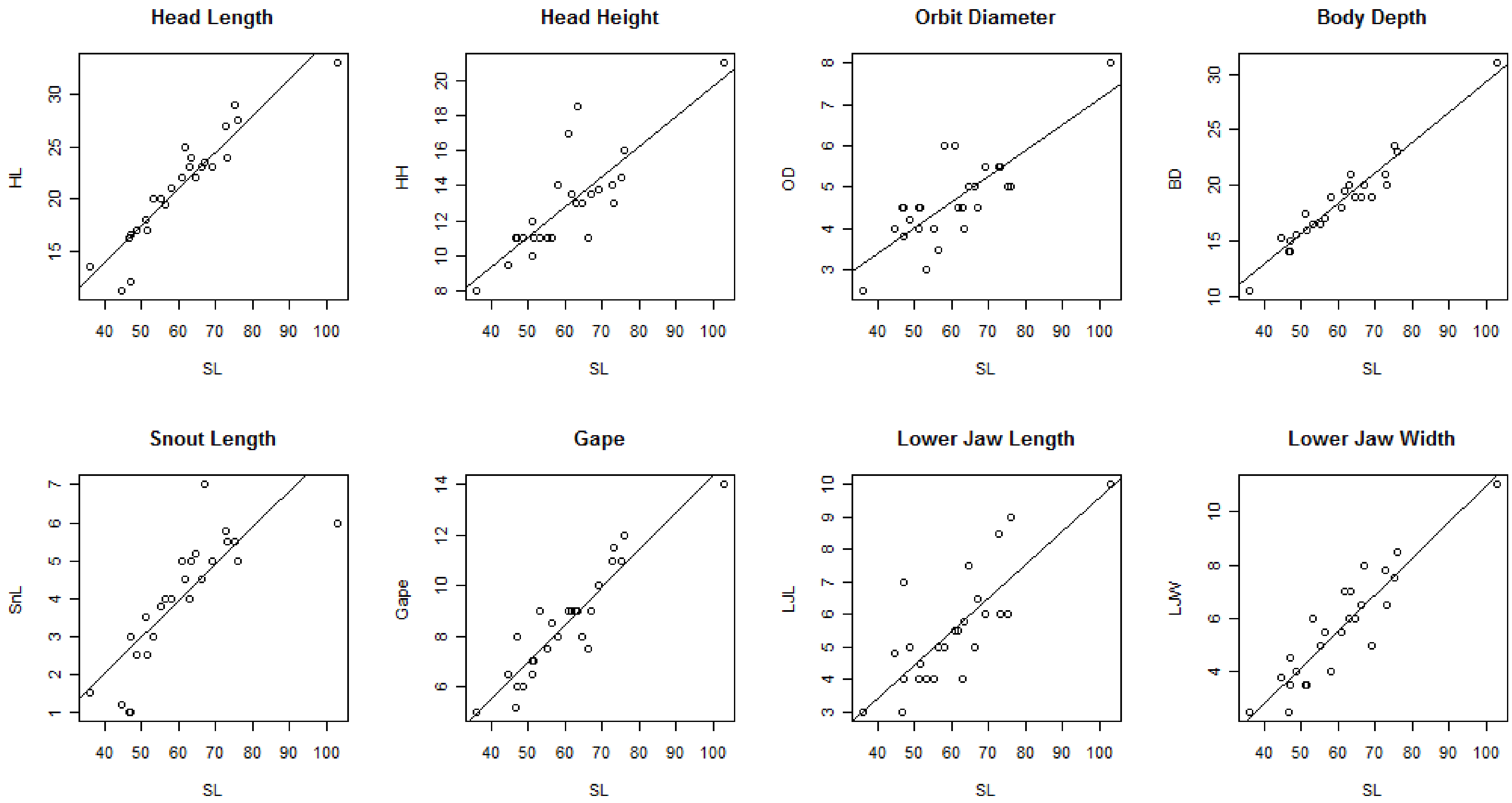


Figure 5.11: Summary of morphometric parameters plotted against the standard length for 26 Cape kurper in the Twee River system. All parameters expressed in mm. See Table 5.8 for linear regression lines and correlation coefficients.

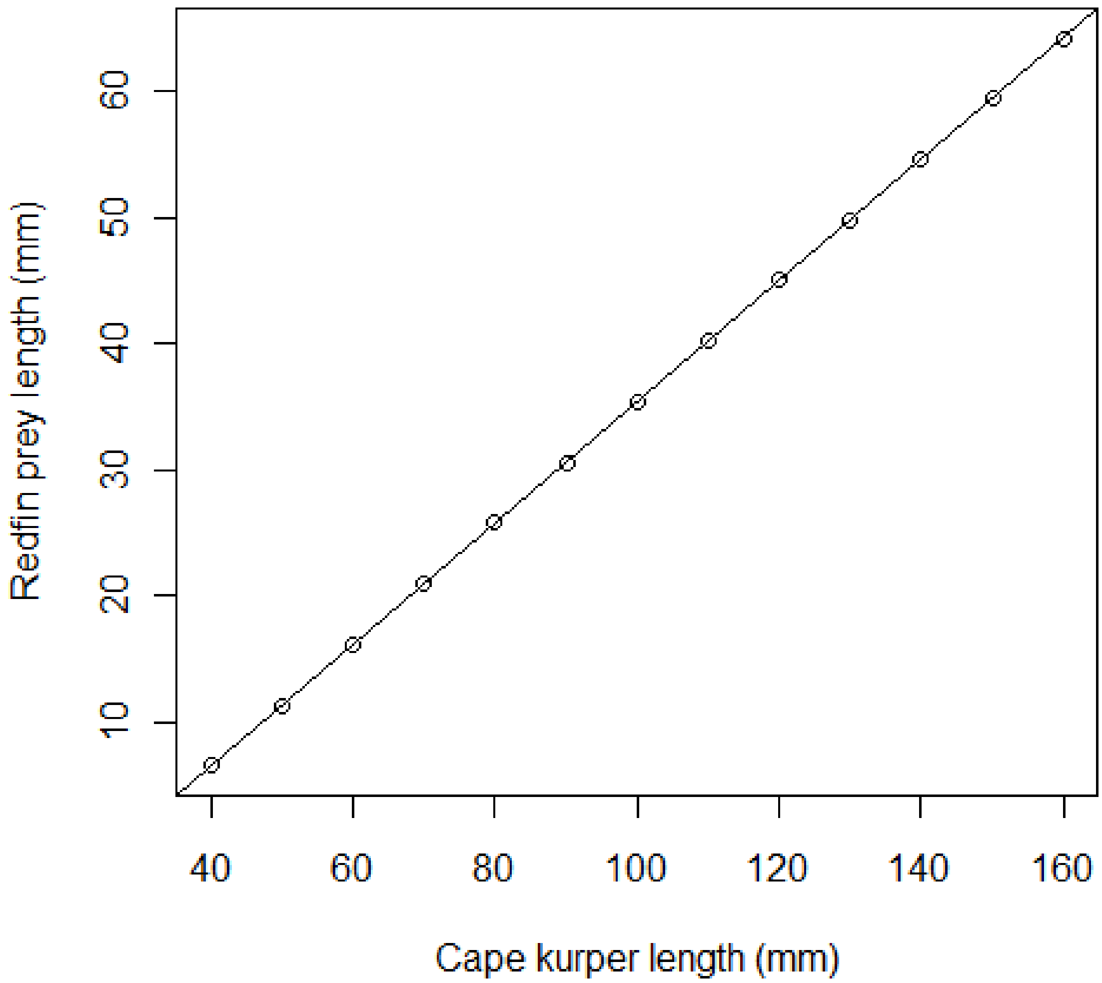


Figure 5.12: Theoretical length of Twee River redfin at risk of being ingested by Cape kurper as a function of the standard length of Cape kurper based on regression relationships body depth-length of redfin and gape-length of Cape kurper.

Tables

Table 5.1: Levels used in the riparian habitat evaluation for the anthropogenic impact and proximity to orchards.

Anthropogenic Impact	
1	No impact - near natural
2	Low impact e.g. small number of alien trees, reduced riparian zone, agricultural activities in close proximity, low level of burning, bridge
3	Medium level impact e.g. severely reduced riparian zone, alien trees, evidence of alien clearing with recovering vegetation, agricultural activities alongside river, high level of recent burning
4	High level impact e.g. no riparian zone, dense alien trees, evidence of recent alien clearing, evidence of bank erosion, totally burned with recovery of natural vegetation, evidence of river engineering (bulldozing) with recovering vegetation
5	Very high level impact e.g. no riparian zone, dense alien trees, current alien clearing, severe bank erosion, totally burned with no recovery, recent river engineering (bulldozing) with no recovering vegetation
Orchards	
1	>50 m away both banks
2	>50 m away one bank, <50 m other bank
3	>20 m away both banks
4	>20 m away one bank, < 20m away other bank
5	<20 m away both banks

Table 5.2: Water chemistry summary for the Suurvlei and Middeldeur tributaries of the Twee River (mean ± standard deviation)

	Suurvlei	Middeldeur	t-Test
pH	6.2 ± 0.50	5.2 ± 0.52	$p < 0.001$
Conductivity (µS/cm)	21.2 ± 1.88	30.1 ± 3.63	$p < 0.001$
Dissolved Oxygen (%)	105.0 ± 8.87	79.6 ± 15.62	$p < 0.001$
Temperature (°C)	19.8 ± 2.01	17.4 ± 1.72	$p < 0.005$
Turbidity (NTU)	0.9 ± 0.20	1.3 ± 0.58	$p < 0.001$
Soluble Reactive Phosphate (µM/L)	0.06 ± 0.00	0.05 ± 0.04	$p = 0.789$
Nitrate (µM/L)	16.9 ± 5.7	16.4 ± 3.4	$p = 0.900$
Nitrite (µM/L)	0.10 ± 0.04	0.27 ± 0.01	$p = 0.002$
Ammonium (µM/L)	0.17 ± 0.33	0.29 ± 0.32	$p = 0.615$

Table 5.3: SASS 5 scores for the sites on the Suurvlei and Middeldeur tributaries of the Twee River including the biological classes *sensu* Dallas and Day (2007)

	SASS Score	ASPT	No. of Taxa	Biological class
SUUR 01	230	7.42	31	X
SUUR 02	92	6.13	15	C
SUUR 03	113	6.28	18	B
MID 02	186	7.75	24	X
MID 03	162	7.36	22	A

Table 5.4: Average Bray-Curtis similarities in water quality variables between, and within (diagonal), water quality sites on the Suurvlei and Middeldeur tributaries of the Twee River

	SUUR 01	SUUR 02	SUUR 03	MID 02	MID 03
SUUR 01	74.53				
SUUR 02	60.05	72.25			
SUUR 03	57.19	72.11	78.41		
MID 02	52.74	38.39	40.85	73.60	
MID 03	46.80	38.25	40.65	55.70	66.87

Table 5.5: Results of Simper analysis of invertebrate assemblages of the Suurvlei and Middeldeur tributaries of the Twee River using Bray-Curtis dissimilarities on raw data. Average dissimilarity = 81.43.

Species	Suurvlei Ave. Abundance	Middeldeur Ave. Abundance	Contribution to dissimilarity (%)	Cumulative dissimilarity (%)
Elmidae	22.43	351.93	32.18	32.18
Hydropsychidae	145.33	9.50	12.60	44.78
Chironomidae	147.67	57.36	11.21	55.99
Simulidae	123.10	6.07	10.18	66.17
Leptoceridae	1.33	80.29	7.78	73.96
Baetidae	87.19	10.93	7.60	81.55
Amphipoda	0.00	43.07	4.28	85.84
Oligachaeta	39.29	11.36	3.22	89.06
Hydraenidae	7.57	26.21	2.51	91.57

Table 5.6: Average Bray-Curtis similarities between, and within (diagonal), invertebrate communities at the water quality sites on the Suurvlei and Middeldeur tributaries of the Twee River. Data was Log(x+1) transformed prior to analysis.

	SUUR 01	SUUR 02	SUUR 03	MID 02	MID 03
SUUR 01	74.527				
SUUR 02	60.051	72.245			
SUUR 03	57.188	72.112	78.412		
MID 02	52.741	38.388	40.853	73.599	
MID 03	46.797	38.245	40.646	55.698	66.87

Table 5.7: Preliminary results of acetylcholine esterase (AChE) activity rate in the brain tissue of Cape kurper (*Sandelia capensis*) measured at sites in the Middeldeur and Suurvlei tributaries in the Twee River system.

Site	Mean AChE rate (µmoles/min/mg)	Variance	Max AChE rate (µmoles/min/mg)	Min AChE rate (µmoles/min/mg)	*n*
M1	6.065	1.435	6.913	5.218	2
M2	1.980	0.682	3.337	0.929	6
M3	2.344	0.895	3.942	1.146	13
S	2.165	0.431	3.091	1.042	13

Table 5.8: Summary of the relationship between selected morphometric parameters and standard length (SL) for 26 Cape kurper from the Twee River catchment. All parameters expressed in mm.

Parameter	Relationship	Adjusted R^2	*p*
Head height (HH)	HH = 0.355***SL** – 0.297	0.868	< 0.0001
Head length (HL)	HL = 0.173***SL** + 2.410	0.663	< 0.0001
Orbit diameter (OD)	OD = 0.063***SL** + 0.896	0.615	< 0.0001
Body depth (BD)	BD = 0.275***SL** + 1.907	0.927	< 0.0001
Snout length (SnL)	SnL = 0.097***SL** + 1.894	0.656	< 0.0001
Gape	Gape = 0.147***SL** – 0.387	0.849	< 0.0001
Lower jaw length (LJL)	LJL = 0.103***SL** – 0.720	0.622	< 0.0001
Lower jaw width (LJW)	LJW = 0.137***SL** – 2.658	0.822	< 0.0001

CHAPTER 6. PERCEPTIONS OF ANGLERS

The social component of conservation is vitally important in determining the outcome of any conservation programme. Freshwater anglers, riparian landowners and conservation agencies need to work together towards safeguarding the future of the native fish populations of the Cape Floristic Region. To achieve this, awareness programmes are required to inform the respective parties of the situation and to maintain open dialogue between all stakeholders. Before an awareness programme can be devised, the perceptions, drivers and loyalties of the respective stakeholders need to be established. Freshwater anglers have been identified as a high-risk vector for the future spread of non-native fish in South Africa (Richardson *et al.* 2004). It is, therefore, important to understand the perceptions of freshwater anglers in order to communicate the importance of native fish conservation to them effectively. In this chapter, the perceptions of freshwater anglers in the Western Cape are evaluated using the Conversion Model^{TM}, a proprietary marketing research tool that evaluates the loyalty of customers to brands, institutions and ideals (Hofmeyr 1990).

6.1 Introduction

Recreational angling is an important leisure activity with an estimated 10.6% of the population of industrialised countries participating (Arlinghaus *et al.* 2002). A number of authors (Cooke and Cowx 2004, 2006) have expressed concerns regarding the impact of recreational angling on fish stocks and their aquatic habitat. The majority of these studies report on the impact of recreational angling on native species and the associated conservation concerns (Arlinghaus *et al.* 2002, Arlinghaus and Mehner 2003, Cowx and Gerdeaux 2004, Arlinghaus 2005, Arlinghaus and Mehner 2005, Cooke and Cowx 2006, Arlinghaus and Cooke 2009, Arlinghaus *et al.* 2010, Cowx *et al.* 2010, Cowx and Van Anrooy 2010, Danylchuk and Cooke 2011). However, these works do not adequately discuss the complexities encountered when recreational angling is based almost exclusively on non-native species, such as is the case in New Zealand, south western USA, and South Africa (Cambray 2003b, Chadderton 2003, Clarkson *et al.* 2005, McDowall 2006). The species targeted in these recreational fisheries are among the most widely introduced species and many are recognised for their detrimental impacts on native aquatic biodiversity (Cambray 2003b). The existence of the fishery is a direct threat to the native fish populations, because, in these regions, the native species are not able to co-exist with the non-native species (Skelton 2000, Impson *et al.* 2002b, Skelton 2002, Cambray 2003b, Marsh and Pacey 2005, Woodford *et al.* 2005, McDowall 2006, Impson 2007). Fisheries based exclusively on non-native species in these regions will, therefore, always be in direct conflict with conservation initiatives for native species. Justification for the conservation of small native species, not attractive to angling community, is difficult, and the anglers do not fully recognise the obligation to protect native biodiversity (Cambray 2003b). Non-native species are better known due to their established economic value and global literature whereas little is known of the native species, which are often poorly studied (Cambray 2003b). In South Africa, there is a growing trend towards angling for native species (Cambray 2003a). It is, however, important that these native species are not stocked beyond their natural ranges.

6.1.1 Importance of Freshwater Angling

Freshwater angling is an important recreational activity in South Africa. Even though the participation in sport and recreational angling in South Africa is low, about 5% of the population, the total economic impact of recreational angling, including deep sea, rock and surf and freshwater, is worth almost three times that of rugby and cricket combined, an estimated R18.8 billion (Leibold and van Zyl 2007). The term "economic impact" is poorly defined by Leibold and van Zyl (2007), but appears to encompass angling and subsidiary income, such as travel, accommodation, retail shops and equipment. The Leibold and van Zyl (2007) report has been criticised (J. Turpie, Anchor Environmental, pers. comm.), but has been used in the absence of published data. Freshwater angling contributes almost half of this value (Table 6.1). Bank angling (including carp and match angling) contribute 46% of the total economic impact, although the economic impact per participant is the lowest at R 2,770 per person per annum (pp pa) due to the estimated 1.5 million informal participants – participants not affiliated to organised clubs or societies including subsistence anglers. Subsistence angling is a minor component of angling in the Western Cape in comparison to other provinces, particularly Kwazulu-Natal and the Orange Free State northwards. The economic impact per participant was highest for fly angling (R 77,780 pp pa) followed by artificial lure (artlure) angling (R 45,850 pp pa), together accounting for more than 54% of the total economic impact. The average economic impact per participant for freshwater angling in South Africa is estimated to be R 5,770 pp pa. The Leibold and van Zyl (2007) survey did not, however, present these figures by province and it thus difficult to report economic impact of freshwater fishing for the Western Cape. Because of high the economic impact per participant of freshwater angling, especially bank (bait), artlure, and fly angling, it is important that the perceptions of these anglers towards the management of non-native fish are taken into account, especially since the fish they target are invasive and not native to the Western Cape.

6.1.2 Conservation versus Angling

The aims of native fish conservation are in direct conflict with those of non-native based recreational angling. Native fish conservation aims to conserve native fish populations by reducing the threats acting on the remaining populations. The greatest threat to the majority native fish in the Cape Floristic Region is the presence of piscivorous non-native fish; the species targeted by the recreational angling groups. As a result, angling groups feel threatened by proposals to conserve native fish populations because these programmes are likely to threaten angling species. This is particularly acute in the Cape Floristic Region largely as a result of the history of fish conservation action in the region.

Historically, the native species of the Cape Floristic Region were not considered suitable angling targets (Coke 1988, Skelton 2000) and, as a result, popular cold and warm water angling species from overseas were introduced to provide recreational angling [see Section 3.2]. In the 1920s, provincial Departments of Inland Fisheries were formed specifically to develop the angling potential of the streams in South Africa by filling the “vacant niches” with suitable angling species (Coke 1988, Skelton 2000). This led to the founding of a number of hatcheries around South Africa to provide angling species for stocking. In the 1960s, provincial Departments for Inland Fisheries became the provincial conservation authorities (Coke 1988, Skelton 2000). Therefore, those formerly responsible for the spread of the majority of non-native fish species were now entrusted with the responsible for the conservation of the native species. This forced change in roles did not result in an immediate change of policies and it was not until the 1980s that conservation authority in the Western Cape changed its policies and started taking its mandate to conserve the native fishes of the region seriously (Coke 1988, Skelton 2000).

The first actions towards the conservation of native fish were declaration of a moratorium on the stocking of non-native species; declaring open season for all non-native species; and the removing of restrictions on angling methods and bag limits (Coke 1988, Skelton 2000). A catch-and-kill policy was encouraged for all non-native species. All angling groups were threatened by these policies and an acrimonious relationship developed between anglers and the conservation authority. The anglers had the perception that the conservation authority planned to eradicate all non-native species from the province and felt that their sport was, therefore, directly threatened. The conservation authority was soon to moderate its stance on non-native species as a result of the public outcry.

The underlying acrimony towards the conservation authority resurfaced when the CAPE Alien Fish Project to eradicate non-native fish using piscicides [see section 3.2] was proposed by the conservation authority in the Western Cape. The media publicized the project and were initially very critical of it, with negative articles appearing in newspapers, e.g. Cape Argus - Bamford (2008), magazines, e.g. Flyfishing - Thorpe (2008), and Farmers Weekly - Steyn (2010), and several fishing web-sites. To a degree, the conservation authority was at fault by not embarking on a pro-active campaign to diffuse the potential media storm (Marr *et al.* in press). Although concern was also expressed regarding the use of piscicides and their impacts on non-target fauna, especially aquatic invertebrates (Marr *et al.* in press), the underlying mistrust of the conservation authority's intentions resurfaced. The absence of a clear plan detailing how the proposed project would fit into the long-term conservation of native fish populations increased the angler concerns that the conservation authority harboured hidden agendas to destroy recreational angling for non-native fish in the province.

6.1.3 Determining social perceptions

Since recreational anglers, along with riparian landowners, are the most important stakeholders in native fish conservation, it is imperative that conservation planners understand their commitment to the non-native species they target, and their perceptions of the need to conserve native fish. Many survey philosophies have been developed by social scientists to measure the perceptions of study groups. Contingent valuation is used to determine the economic value of a resource to the users and non-users through philosophies such as "willingness-to-pay". Similarly, behavioural measures of loyalty could be used to determine indices based on frequency of use, consistency of selection, and similar metrics derived from, or described by, Jacoby and Chestnut (1978) and Jacoby *et al.* (1978). However, these measures do not provide insight into the motivation underpinning behaviour, nor the levels of commitment involved i.e. the psychological relationship between the consumer and the available products (Ceurvorst 1993).

6.1.4 The Conversion Model™

The patience and stillness required for angling is an important mechanism that allows the participant to process the physical and psychological stresses of modern occupations and lifestyles. Recreational angling thus brings balance to many participants and has become an important component of their sense of well-being. Because recreational angling is voluntary, not motivated by economic gain, re-connects the participant to nature, and has deeply psychological benefits (Arlinghaus *et al.* 2002, Arlinghaus and Cooke 2009), there are parallels between recreational angling and religious involvement. It is therefore unreasonable to expect an angler to forego his angling for the benefit of species he may not have any affiliation to. However, it may be possible to “convert” an angler from a non-native species to a native species. To achieve this conversion, an understanding of the angler’s loyalty and commitment to his current target species is required, in conjunction with his perceptions of the attractiveness of the alternate native species. The underlying philosophy of the Conversion Model™ was developed by evaluating religious conversion (Hofmeyr 1986) and subsequently customized as a marketing tool to evaluate brand switching (Hofmeyr 1990, Rice and Hofmeyr 1990, Hofmeyr and Rice 1995). The Conversion Model™ is thus an appropriate tool to evaluate the perceptions of freshwater anglers towards the available species and the conservation of native fishes.

The Conversion Model™, a proprietary marketing research tool, has been extensively used to gauge the strength of a consumer's psychological commitment, or loyalty, to a product (Hofmeyr 1990). In this context, a product may be an ideology, pastime, political party, or commercial brand. The consumers are classified according to how committed they are to using a product, whereas non-users are classified according to how likely they are to be converted to using it (Rice and Hofmeyr 1990, Hofmeyr and Rice 1995). Four psychological dimensions underpin the Conversion Model (Hofmeyr 1990, Rice and Hofmeyr 1990, Hofmeyr and Rice 1995). The first identifies the degree of satisfaction a person experiences in the fit between the product and his/her needs and values: the better the fit, the higher the level of commitment. It is known, however, that dissatisfied consumers may persist with their current product whereas satisfied consumers may defect to rival products. The Conversion Model, therefore, goes beyond the measurement of satisfaction by evaluating a customer's commitment to a specific product. In the second dimension of the model the consumer's psychological and material involvement in the product is measured - the extent to which the consumer identifies with the product. The higher the level of involvement, or identification, the consumer has with the product, the lower the likelihood that s/he will switch to an alternative product. The third dimension is the relative attraction power of alternatives to the product currently supported by the consumer, as this commitment to the product can be undermined by the appeal of competitors’ products (Hofmeyr and Rice 1995). The final dimension is the ambivalence of the consumer towards the products available - a consumer's inclination to be a "fence sitter" when confronted with appealing alternative to a product to which they are committed.

A number of outputs are generated by the Conversion Model. The commitment model segments consumers according to how strongly they are committed to their preferred product. The four segments of the "Strength of Commitment" output are (Rice and Hofmeyr 1990):

- **Entrenched -** staunchly loyal users who are not available for conversion to an alternative product
- **Average -** quite staunchly loyal; not immediately available for conversion, but potentially available.
- **Shallow -** weakly loyal users; some will be immediately available for conversion.
- **Convertible -** marginal users; small negative changes in their feelings about current usage causes switching to alternative products.

The underlying disposition model segments non-users of a particular product in terms of how strongly they are attracted to that product in comparison with the one they are currently using. The four segments of the "Balance of Disposition" model are (Rice and Hofmeyr 1990):

- **Available -** non-users of the alternative product who prefer the alternative product to their current one although they have not yet switched.
- **Ambivalent -** non-users of the alternative product who are attracted to both the alternative product and their current one simultaneously.
- **Weakly Unavailable -** non-users of the alternative product whose preference lies with their current product, but not strongly.
- **Strongly Unavailable -** non-users of the alternative product whose preference is strongly for their current product.

The "States of Mind" output is not product specific. It looks at how the person responded to all the products they use, and are aware of, and categorises them as either (Rice and Hofmeyr 1990):

- **Single-minded**: Strong attachment to one product. Committed to one, may be uncommitted to others. Unlikely to switch from what they are using.
- **Passive**: Committed, but less involved (i.e. they're happy with the product they use but the brand of product is not as important, if they find another product equally appealing they could switch but they are not looking to switch.
- **Shared**: Equally attracted to two or more products. Can either be committed or uncommitted. Generally happy with the product but find a number of different types appealing.

- **Seekers**: Unhappy with current product. Unhappy with the product they currently use. Looking for an alternative but either have not found one or there are barriers to use.
- **Uninvolved**: Don't care which product they use. Availability and convenience are key here – they won't go out of their way or spend more to use anything specific.

6.1.5 Aims of this Chapter

The purpose of this chapter is to better understand the commitment of freshwater anglers to their preferred species. The angler's knowledge of freshwater fish conservation in the Western Cape was measured to evaluate their awareness of the need for conservation action and the management of non-native fishes in the Cape Floristic Region. A questionnaire-based social survey was conducted of freshwater anglers residing in Greater Cape Town. The survey was designed using the philosophy of the Conversion Model.

6.2 Methods

Permission was gained from TNS Global, one of the world's leading market research groups and the owner of the Conversion Model, to use the Conversion Model in this survey. Discussions were held with TNS Global staff in Cape Town, and a questionnaire developed in accordance with the guidelines for the Conversion Model provided by TNS Global.

6.2.1 Development of Survey Questions

The questionnaire contained two main components: Angling and Conservation (Appendix A.6.1). The Angling Questionnaire was designed to determine the loyalty of the respondent to their preferred target species using the Conversion Model whereas the Conservation Questionnaire was designed to measure the awareness of the respondent of the conservation issues facing native fish in the Cape Floristic Region. A common set of 18 fish species found in the Western Cape was used for both questionnaires. Nine of the species are native to the Cape Floristic Region. Named pictures of the fish species referred to in the questionnaires were made available to the respondent while they were completing the questionnaires (Appendix A.6.1), but the respondents were not informed which of the species were native to the region. In addition, two questions were included to elicit the respondent's perception of the conservation efforts of the conservation authority and eradication techniques for non-native fishes.

6.2.2 Collection of Data

The survey was restricted to freshwater anglers based in Cape Town. An attempt was made to cover all facets of freshwater angling in the Western Cape by engaging all the major disciplines of freshwater angling: Western Province Bank Anglers, Western Province Bass, and the Cape Piscatorial Society. In addition, several fishing tackle stores in the greater Cape Town area were visited and a random sample of the patrons approached to participate in the survey. The majority of the questionnaires were answered in face-to-face interviews, supplemented with telephonic interviews where face-to-face interview were not possible. The anglers were designated to one of fly fishing, bank angler, or art lure groups based on their indicated preferred angling technique.

6.2.3 Analysis of Survey Responses

A database was compiled from the completed questionnaires and the angling component presented to TNS to run the Conversion Model. The conservation component of the survey was not suitable for the Conversion Model because no single favourite species had been elicited in the responses. TNS provided the results from the Conversion Model and these were analysed in conjunction with the raw angling and conservation data. Summary statistics were compiled for the respective sections of the questionnaires and Conversion Model results. A total of 70 anglers were interviewed, 36 fly, 15 bank (bait), and 19 artlure anglers. Differences in awareness between the respective angling groups regarding species present in the Western Cape, species native to the Western Cape, and species endangered within the Western Cape were evaluated using PERMANOVA routine in PRIMER-E 6.1.5 (Anderson 2001a, b, 2006, Anderson *et al.* 2008). These tests were performed on a resemblance matrix generated from the raw data using the Bray-Curtis similarity. Differences in the perception of the respective angling groups regarding the species threatened by habitat degradation, water pollution, water abstraction, pesticides, non-native plants, and non-native fish in the Western Cape were evaluated using PERMANOVA.

6.3 Results

The results of the social survey show that the different angling groups differ in their levels of awareness of the freshwater fishes in the Western Cape. Of the angling groups, the fly anglers showed the greatest awareness of the native species of the Western Cape and the greatest support for the conservation of the native fishes of the region.

6.3.1 Angler Awareness

Respondents from all angling groups showed a greater awareness of non-native species than of native species (Table 6.2). Of the non-native species, respondents were least familiar with banded tilapia *Tilapia sparrmani*, Mozambique mouthbrooder *Oreochromis mossambicus*, and bluegill sunfish *Lepomis macrochirus*. Of the native species, respondents were most familiar with Clanwilliam yellowfish *Labeobarbus capensis*, Cape kurper *Sandelia capensis* and redfin *Pseudobarbus spp.* whereas rock catlets *Austroglanis spp.* and sandfish *Labeo sebeeri* were the less well known (Table 6.2). The PERMANOVA analysis revealed a significant difference between the knowledge of the angling groups when all species were considered and for the non-native species (all species $p = 0.023$, non-native species $p = 0.008$) but no significant difference was found for the native species ($p = 0.181$). Pair-wise tests show that there were no significant differences in awareness of all species between bank and artlure anglers ($p = 0.341$) and artlure and fly anglers ($p = 0.084$) but a significant difference between the awareness of bank and fly anglers ($p = 0.001$). For the non-native species, there was no significant differences in awareness of all species between bank and artlure anglers ($p = 0.123$), but significant differences between the awareness of fly anglers and the other two groups (bank $p = 0.005$, artlure $p = 0.032$). For the native species, there were no significant differences in awareness of all species between bank and artlure anglers ($p = 0.796$) and artlure and fly anglers ($p = 0.345$), but a significant difference between the awareness of bank and fly anglers ($p = 0.038$).

The fly anglers displayed a superior knowledge of the native species with more than 70% of the respondents correctly identifying native species. Only 40% of the bank and artlure anglers correctly identified native species. Moggel *Labeo umbratus*, rock catlets and sandfish were most frequently considered not to be native to the Western Cape (Table 6.2). The poor knowledge of moggel could be explained by most respondents not considering the Gouritz River system as being included in the Western Cape. Bluegill, sharptooth catfish *Clarias gariepinus* and rainbow trout *Oncorhynchus mykiss* were the non-native species most commonly thought to be native. The PERMANOVA analysis revealed a significant difference in the perception of which species were native to the Western Cape between the angling groups when all species were considered (all species $p = 0.002$), but no significant difference was found when only native species were considered ($p = 0.064$). Pair-wise tests show that there was no significant differences in ability to identify native species between bank and artlure anglers ($p = 0.194$) when all species were considered, but significant differences between the awareness of fly anglers and the other two groups (bank $p = 0.003$, artlure $p = 0.002$). Considering only native species, there were no significant differences between bank and artlure anglers (p = 0.099) and bank and fly anglers (p = 0.332).but a significant difference between artlure and fly anglers (p = 0.043).

Some respondents included non-native species among the species they considered to be endangered. The most commonly included were rainbow trout and brown trout *Salmo trutta*. Yellowfish were considered to be endangered by the highest percentage of respondents, even though they are currently listed as Vulnerable in the 2007 IUCN Red Data assessment (Tweddle *et al.* 2009). Redfin, a composite group including species with a range of IUCN threat statuses from Near Threatened to Critically Endangered, were the next most common native species considered to be endangered. Moggel (Least Concern) were considered endangered by the lowest percentage of respondents, followed by rock catlets (Vulnerable and Endangered) and the Data Deficient species: Cape kurper and Cape galaxias *Galaxias zebratus*. The PERMANOVA analysis revealed a significant difference in the perception of which species were endangered between the angling groups (p = 0.028). Pair-wise tests show that there were no significant differences in perceptions of endangerment between bank and artlure anglers (p = 0.820) and bank and fly anglers (p = 0.069), but a significant difference between artlure and fly anglers (p = 0.009).

The species targeted most often were the species usually associated with the respective angling groups. Fly anglers predominantly targeted rainbow trout (>80%) and artlure anglers preferred bass *Micropterus spp.* (>80%) while bank anglers preferred carp (~75%). The only native species to be included among the preferred target species by a small percentage of each group was Clanwilliam yellowfish, (Table 6.2).

6.3.2 Support for Fish Conservation

Fly anglers showed the greatest support for the conservation of native species (84% of respondents) but showed strong support for the conservation of both trout species (Table 6.3). Bank anglers showed a strong support for native species (62%) but showed no strong preference for the conservation of non-native species. Artlure anglers were the least supportive of native fish conservation (53%) and showed strong support for the conservation of both trout and both bass species (57%). Eighty percent of bank anglers indicated that the conservation of native fishes was extremely important whereas 72% of fly anglers and 47% of artlure anglers expressed the same level of commitment to the conservation of native species (Table 6.4).

6.3.3 Conversion Model

The output of the Conversion Model shows that the majority of anglers from each group (about 60% for bank and artlure anglers, and > 80% for fly anglers) have an average commitment to their preferred target species (Table 6.5). An average commitment indicates that the support for the target species is quite staunch, and the anglers are not immediately available for conversion to an alternate species, but potentially available. More than 30% of the artlure anglers were entrenched indicating that they are strongly loyal to their preferred species and not available for conversion to an alternate species. About one third of bank anglers had shallow or convertible commitments to their preferred species whereas only 10% of the other two groups had shallow commitments. An ANOVA showed that there was a significant difference in the state of commitment between bank anglers and the other two groups (Artlure p = 0.019, Fly p = 0.044), but no significant difference between artlure and fly anglers (p = 0.064).

The "state of mind" output presents a slightly different perspective. In this analysis, 57% of bank anglers, 47% of artlure, and 42% of fly anglers are described as single-minded, indicating that they had a strong attachment to the preferred species and were unlikely to switch from that species (Table 6.6). The next highest group was the shared group (29% of bank anglers, 37% of artlure, and 44% of fly anglers) indicating that they equally attracted to two or more species and generally happy with their angling and find a number of different species appealing. An ANOVA showed that there was no significant difference in the state of mind between the angling groups.

The segmentation output of the Conversion Model indicates that the majority of the native species are of no interest to the freshwater anglers. The bank anglers were committed to carp, artlure anglers to bass species and the fly anglers to the trout species (Table 6.7). The only native species of any interest to anglers are the larger cyprinids Clanwilliam yellowfish, sawfin *Barbus serra*, whitefish *Barbus andrewii*.

6.3.4 Threats

The respondents were asked to indicate the species they believed were threatened by habitat destruction, water pollution, water abstraction, pesticides, non-native plants, and non-native fish. Many of the respondents indicated during the interviews that they did not believe that non-native fish were a threat to the native fishes of the Western Cape. Respondents from each angling groups indicated that native fish were threatened by all the threats included (Table

6.8). A greater percentage of fly anglers believed that native fish were threatened by the respective threats than the other groups. Artlure anglers' participation the threat analysis was lower than that of the other groups. The PERMANOVA analysis of the threats indicates that there is no significant difference between the respondents from the respective angling groups, or the threats they indicated threatened the native fishes (Anglers p = 0.289, Threats p =0.151, Angler x Threats p = 0.751).

6.3.5 CapeNature and Piscicides

The respondents were required to indicate whether they believed that the conservation authority for the Western Cape, CapeNature, was doing a good job of conserving the native freshwater fishes in the region. Each angling group showed a distinct perception of CapeNature. The bank anglers were largely supportive of CapeNature with 60% indicating that they believed that CapeNature were doing a good job. The artlure anglers were undecided with 50% of the respondents supportive of CapeNature. The fly anglers were distinctly unimpressed with CapeNature, 73% indicating that they believed the CapeNature was not doing a good job of conserving the native freshwater fishes of the Western Cape.

Piscicides are an important tool for the management of non-native fish populations. The respondents were asked to indicate whether they were aware of piscicides, believed that they were effective, or found their use acceptable. Fly anglers were the most informed about piscicides with 97% of the respondents aware of piscicides, 75% believing them to be effective and 36% finding their use acceptable. Artlure anglers were aware of piscicides with 74% of the respondents aware of piscicides, 47% believing them to be effective, and 5% finding their use acceptable. Bank anglers were the least supportive of the use of piscicides with 67% of the respondents aware of piscicides, 27% believing them to be effective and none finding their use acceptable.

6.4 Discussion

The results of the Conversion Model show that the majority of anglers from the respective angling groups are single-mindedly dedicated to their target species, mostly non-native species. The remainder of the anglers have shared commitment to a range of species, mostly including the native yellowfish. The segmentation output of the Conversion Model indicates clearly that the anglers display no interest in moving to angling for the smaller native species. Most anglers are satisfied with their target species and, although the majority of respondents showed average commitment to their preferred target species, they were also unavailable for conversion to other species, particularly native species. The only native species with any potential are the yellowfish. The outputs from the Conversion Model have provided valuable data allowing a better understanding of the perceptions of the freshwater anglers of the Western Cape.

This survey has clearly indicated that non-native species are very important to recreational angling groups in the Western Cape. The majority of the native species of the region hold little interest to angling groups. The lack of suitable angling species was the reason for the original introduction of the non-native species to the region. Of the native species, only the Clanwilliam yellowfish, and to a lesser degree whitefish and sawfin, hold any attraction for the anglers. However, Clanwilliam yellowfish and sawfin are only found in the Olifants-Doring catchment, a three hour drive from Cape Town. Further, many populations of these species are not openly available to anglers. Whitefish have largely been extirpated from the riverine habitat and only persist in a few impoundments. The recent confirmation of the introduction of non-native smallmouth yellowfish *Labeobarbus aeneus* into the Breede River (K. Broos, Cape Piscatorial Society, 2011, pers. comm.) is indicative that anglers would like more access to fishing for the larger cyprinids of the Western Cape. It has been suggested (L. Fleming, Yellowfish Working Group, 2011, pers. comm.) that smallmouth yellowfish have already been introduced into the Berg River indicating the desire of fly anglers for yellowfish angling close to Cape Town. The Berg and Breede Rivers are the natural distribution range of whitefish, which has largely been extirpated from the Berg River and from riverine habitat in the Breede River. It has to be questioned why anglers would introduce a new species, rather than promote conservation programmes for whitefish.

Because recreational angling is an important contributor to the local economy, and an important pastime for many people, it is clear that the conservation of the native fishes of the Western Cape needs to proviside for recreational angling. Conservation projects requiring the sacrifice of populations of angling species require clear communication of the need for the project, and justification for the loss of the angling opportunity. Without the support and buy-in of anglers, irresponsible and spiteful introduction of non-native species could sabotage well-meaning conservation projects (Johnson *et al.* 2009). Respondents from all angling groups acknowledged the need to conserve the native fishes of the region, but would clearly like to continue angling for their preferred non-native species. The conservation authority could use the anglers desires to fish for the larger cyprinids to generate greater support for native fish conservation projects. Conservation projects using the larger cyprinids as flagship of umbrella species should be used to provide angling opportunities closer to Cape Town.

The use of piscicides to eradicate non-native fish has been proposed in the CAPE GEF Alien fish project. Respondents from all angling groups expressed concern about CapeNature's desire to use piscicides to remove non-native fish. Averaged over the groups, fewer than 15% of the anglers supported the use of piscicides. Whether this is an expression of the concern that once CapeNature begins eradicating non-native fishes, they will continue and destroy all recreational angling in the province, or ethical objections to the use of piscicides, needs to be determined. This concern was greatest amongst fly anglers who recognised that trout populations were in the best habitat for the rehabilitation of native fishes. During the survey, it was clear than many of the respondents were not supportive of the GEF Alien Fish Eradication Project, contrary to the belief of the conservation authority (Marr *et al.* in press).

Anglers are concerned that the conservation authority is planning to eradicate all non-native fishes. In fact, the conservation authority is not in a position to eradicate all non-native fish populations and is merely trying to reduce the non-native fish invasion, say from 90% to 75 or 80%, such that the native fish populations can be secured for future generations. There is a place for recreational angling based on non-native fish in the Western Cape, but sacrifices will have to be made on both sides. Conservation has to work with angling groups to ensure that they support conservation initiatives, and although conservation has no obligation to provide for recreational angling, they should include treats for anglers in their conservation planning.

More than half of the anglers expressed concern that CapeNature is not doing a good job of conserving the native fishes of the Western Cape. Most respondents expressed a desire to see more active conservation for native fishes by CapeNature. The need for a clear conservation plan detailing the conservation actions planned, how the conservation outcomes would be achieved, and the sacrifices required from the respective angling groups was expressed. Clear communication of CapeNature's stance on non-native species was also called for. CapeNature have been developing a non-native fish policy for more than three years, but this policy has not even been released for comment by researchers.

The lack of awareness among angler groups of which species are native to the Western Cape, and which species are endangered, is indicative that the education of anglers, and the general public, is required. Increasing awareness of the native freshwater fishes of the region, and the threats acting on the remaining populations, will garnering greater support for conservation projects, especially those requiring the removal of non-native species. At present, a portion of anglers, mostly fly anglers, are very concerned about the lack of conservation action and expressed a willingness to be involved in native fish conservation projects. Although many anglers are vocal in their support for non-native angling species, there is a growing support for native fishes and the call to conserve the native species is growing. Greater support among the general public could result in the call for the conservation of native species to grow louder than the calls to maintain the *status quo*.

The lack of a documented conservation plans for the native fishes in the Western Cape and management plans for non-native species has reduced angler confidence in CapeNature, as has the lack of conservation action for the native freshwater fishes of the region. The proposed use of piscicides has further reduced angler support for CapeNature. CapeNature's failure to communicate adequately with the angling groups by providing justification for the proposed piscicide projects and educational material detailing how piscicide applications are planned and executed, the risks involved in the administration of piscicides and how they can be managed, has resulted in misinformation about the use of piscicides. In Australia, the use of piscicides has been prevented by pro-trout lobby groups (Lintermans 2000, 2004). Many anglers feel that CapeNature has bullied its way to the approval of the proposed piscicide projects and are not supportive of the concept of using piscicides. This survey has made it clear that CapeNature have considerable work to do to build a working relationship with the angling groups.

Tables

Table 6.1: Participation in and the total economic value of the respective facets of freshwater angling in South Africa, compiled from Leibold and van Zyl (2007). Formal participants include all anglers affiliated to registered clubs and societies. The informal participants include non-affiliated anglers and subsistence anglers.

Facet	Formal participants	Informal participants	Total participants	Total economic impact (per annum)	Economic impact (per person per annum)
Bank angling	6,500	1,502,530	1,509,030	R 4.2 billion	R 2,770
Fly angling	4,500	40,500	45,000	R 3.5 billion	R 77,780
Artlure angling	3,200	28,800	32,000	R 1.5 billion	R 45,845
Total	14,200	1,571,830	1,586,030	R9.15 billion	R5,770

Table 6.2: Summary of angling group's awareness of the freshwater fish species present in the Western Cape, species native to the Western Cape, and their level of endangerment. Results are expressed as percentages of the respondent within the respective groups.

		Non-native									Native										
		Rainbow trout	Brown trout	Smallmouth bass	Largemouth bass	Bluegill sunfish	Carp	Sharptooth catfish	Mozambique mouthbrooder	Banded tilapia	Sawfin	Yellowfish	Whitefish	Sandfish	Moggel	Cape kurper	Cape galaxias	Redfin	Rock catlets	Non-native	Native
Aware	Bank	100.0	93.3	93.3	93.3	73.3	100.0	80.0	66.7	66.7	66.7	86.7	66.7	53.3	60.0	80.0	53.3	66.7	53.3	85.2	65.2
	Artlure	100.0	100.0	100.0	100.0	100.0	94.7	78.9	78.9	73.7	52.6	94.7	68.4	52.6	63.2	94.7	68.4	78.9	52.6	91.8	69.6
	Fly	100.0	100.0	100.0	100.0	97.2	100.0	97.2	97.2	86.1	83.3	100.0	91.7	77.8	77.8	94.4	80.6	94.4	61.1	97.5	84.6
	Average	100.0	97.8	97.8	97.8	90.2	98.2	85.4	80.9	75.5	67.5	93.8	75.6	61.2	67.0	89.7	67.4	80.0	55.7		
Consider to be native	Bank	13.3	13.3	0.0	0.0	26.7	13.3	6.7	6.7	20.0	40.0	46.7	53.3	40.0	20.0	53.3	40.0	46.7	46.7	11.1	43.0
	Artlure	26.3	10.5	10.5	15.8	15.8	5.3	21.1	0.0	5.3	26.3	73.7	42.1	31.6	10.5	63.2	52.6	47.4	26.3	12.3	41.5
	Fly	2.8	2.8	0.0	0.0	2.8	2.8	11.1	0.0	0.0	69.4	94.4	80.6	61.1	33.3	88.9	77.8	86.1	52.8	2.5	71.6
	Average	14.1	8.9	3.5	5.3	15.1	7.1	12.9	2.2	8.4	45.3	71.6	58.7	44.2	21.3	68.5	56.8	60.0	41.9		
Consider to be endangered	Bank	13.3	0.0	0.0	0.0	0.0	0.0	0.0	6.7	6.7	40.0	60.0	20.0	33.3	13.3	26.7	26.7	40.0	20.0	3.0	31.1
	Artlure	5.3	5.3	5.3	0.0	0.0	5.3	5.3	0.0	0.0	15.8	42.1	21.1	26.3	5.3	31.6	21.1	47.4	21.1	2.9	25.7
	Fly	11.1	11.1	0.0	0.0	0.0	2.8	0.0	0.0	0.0	66.7	66.7	66.7	55.6	11.1	36.1	52.8	75.0	44.4	2.8	52.8
	Average	9.9	5.5	1.8	0.0	0.0	2.7	1.8	2.2	2.2	40.8	56.3	35.9	38.4	9.9	31.5	33.5	54.1	28.5		
Target most often	Bank	0.0	0.0	0.0	13.3	0.0	73.3	6.7	0.0	0.0	0.0	6.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0		
	Artlure	5.3	0.0	47.4	36.8	0.0	0.0	0.0	5.3	0.0	0.0	5.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0		
	Fly	80.6	5.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	13.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0		

Table 6.3: Commitment of angling group's towards the conservation of the freshwater fish species present in the Western Cape. The respondents were asked to indicate whether they could find "many good reasons" (Mgr), "few good reasons" (Fgr), or an intermediate level (Int) to justify the conservation of the respective species. Results are expressed as percentages of the respondent within the respective groups. Not all participants provided a response. The average section presents the average across the angling groups.

		Non-native									Native										
		Rainbow trout	Brown trout	Smallmouth bass	Largemouth bass	Bluegill sunfish	Carp	Sharptooth catfish	Mozambique mouthbrooder	Banded tilapia	Sawfin	Yellowfish	Whitefish	Sandfish	Moggel	Cape kurper	Cape galaxias	Redfin	Rock catlets	Non-native	Native
Bank Anglers	Mgr	40.0	40.0	40.0	40.0	46.7	40.0	40.0	40.0	40.0	60.0	73.3	60.0	60.0	60.0	53.3	60.0	60.0	66.7	40.7	61.5
	Int	33.3	33.3	20.0	26.7	6.7	26.7	13.3	6.7	6.7	13.3	13.3	6.7	6.7	6.7	20.0	13.3	13.3	6.7	19.3	11.1
	Fgr	26.7	26.7	33.3	26.7	20.0	33.3	33.3	26.7	26.7	0.0	0.0	6.7	6.7	0.0	0.0	0.0	0.0	0.0	28.1	1.5
Artlure Anglers	Mgr	84.2	78.9	73.7	73.7	47.4	36.8	26.3	42.1	36.8	47.4	73.7	63.2	47.4	47.4	52.6	52.6	47.4	47.4	55.6	53.2
	Int	10.5	10.5	21.1	21.1	15.8	21.1	5.3	5.3	10.5	10.5	15.8	15.8	21.1	15.8	15.8	15.8	21.1	21.1	13.5	17.0
	Fgr	5.3	5.3	5.3	5.3	10.5	36.8	47.4	21.1	21.1	0.0	0.0	0.0	0.0	5.3	5.3	5.3	0.0	0.0	17.5	1.8
Fly Anglers	Mgr	55.6	58.3	16.7	22.2	16.7	11.1	11.1	13.9	13.9	86.1	94.4	88.9	83.3	75.0	77.8	83.3	88.9	77.8	24.4	84.0
	Int	33.3	30.6	41.7	33.3	13.9	25.0	11.1	27.8	25.0	0.0	2.8	2.8	5.6	5.6	5.6	2.8	2.8	11.1	26.9	4.3
	Fgr	11.1	11.1	38.9	41.7	61.1	58.3	69.4	44.4	44.4	0.0	0.0	0.0	0.0	5.6	5.6	2.8	0.0	0.0	42.3	1.5
Average	Mgr	59.9	59.1	43.5	45.3	36.9	29.3	25.8	32.0	30.2	64.5	80.5	70.7	63.6	60.8	61.2	65.3	65.4	63.9	40.2	66.2
	Int	25.7	24.8	27.6	27.0	12.1	24.2	9.9	13.2	14.1	8.0	10.6	8.4	11.1	9.3	13.8	10.6	12.4	12.9	19.9	10.8
	Fgr	14.3	14.3	25.8	24.5	30.5	42.8	50.0	30.7	30.7	0.0	0.0	2.2	2.2	3.6	3.6	2.7	0.0	0.0	29.3	1.6

Table 6.4: Degree of importance of conserving native freshwater fishes, as determined by respondents from the respective angling groups of the Western Cape. Results are expressed as percentages of the respondent within the respective groups.

Importance	Bank	Artlure	Fly
Extremely	80.0	47.4	72.2
Very	20.0	36.8	22.2
Moderately	0.0	10.5	5.6
Not at all	0.0	5.3	0.0

Table 6.5: Strength of commitment of respondents from the respective angling groups towards their preferred angling species, as determined by the Conversion Model. Results are expressed as percentages of the respondent within the respective groups.

	Bank	Artlure	Fly
Entrenched	7.1	31.6	5.6
Average	57.1	57.9	83.3
Shallow	21.4	10.5	11.1
Convertible	14.3	0.0	0.0

Table 6.6: "State of mind" of respondents from the respective angling groups regarding their preferred angling species, as determined by the Conversion Model. Results are expressed as percentages of the respondent within the respective groups.

	Bank	Artlure	Fly
Single Minded	57.1	47.4	41.7
Passive	0.0	5.3	11.1
Shared	28.6	36.8	44.4
Seekers	0.0	10.5	0.0
Uninvolved	14.3	0.0	2.8

Table 6.7: Conversion Model Segmentation between the strength of commitment and balance of disposition models for angling groups by available freshwater fish species. The results are expressed as percentage of respondents within the respective groups.

		Non-native									Native								
		Rainbow Trout	Brown trout	Smallmouth bass	Largemouth bass	Bluegill Sunfish	Carp	Sharptooth catfish	Mozambique mouthbrooder	Banded tilapia	Sawfin	Yellowfish	Whitefish	Sandfish	Moggel	Cape kurper	Cape galaxias	Redfin	Rock catlets
Bank Anglers	Entrenched	0.0	0.0	0.0	0.0	0.0	7.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Average	0.0	0.0	0.0	14.3	0.0	42.9	7.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Shallow	0.0	0.0	7.1	7.1	0.0	28.6	21.4	7.1	0.0	0.0	14.3	7.1	0.0	0.0	0.0	0.0	0.0	0.0
	Convertible	28.6	21.4	21.4	35.7	28.6	21.4	28.6	7.1	0.0	0.0	14.3	21.4	0.0	7.1	7.1	0.0	0.0	0.0
	Available	7.1	0.0	0.0	0.0	7.1	0.0	0.0	7.1	0.0	21.4	14.3	0.0	14.3	0.0	0.0	0.0	0.0	0.0
	Ambivalent	14.3	14.3	14.3	7.1	0.0	0.0	0.0	28.6	21.4	21.4	7.1	7.1	21.4	21.4	21.4	7.1	7.1	14.3
	Weakly Unavailable	7.1	14.3	7.1	0.0	14.3	0.0	28.6	0.0	28.6	7.1	0.0	21.4	14.3	7.1	7.1	7.1	14.3	7.1
	Strongly Unavailable	42.9	50.0	42.9	35.7	50.0	0.0	14.3	50.0	50.0	50.0	50.0	42.9	50.0	64.3	64.3	85.7	78.6	78.6
Artlure Anglers	Entrenched	5.3	0.0	15.8	0.0	0.0	0.0	0.0	5.3	0.0	0.0	5.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Average	0.0	0.0	42.1	52.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Shallow	0.0	5.3	21.1	15.8	0.0	5.3	0.0	0.0	0.0	0.0	10.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Convertible	31.6	10.5	10.5	21.1	10.5	26.3	15.8	10.5	5.3	0.0	0.0	0.0	0.0	0.0	5.3	0.0	0.0	0.0
	Available	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	10.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Ambivalent	36.8	36.8	5.3	5.3	15.8	5.3	5.3	21.1	0.0	10.5	21.1	15.8	0.0	5.3	0.0	0.0	0.0	0.0
	Weakly Unavailable	10.5	15.8	0.0	0.0	15.8	26.3	10.5	21.1	31.6	15.8	15.8	26.3	21.1	26.3	15.8	10.5	15.8	10.5
	Strongly Unavailable	15.8	31.6	5.3	5.3	57.9	36.8	63.2	42.1	63.2	73.7	36.8	57.9	78.9	68.4	78.9	89.5	84.2	89.5
Fly Anglers	Entrenched	2.8	2.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Average	66.7	44.4	5.6	2.8	0.0	0.0	0.0	0.0	0.0	5.6	25.0	2.8	0.0	0.0	0.0	0.0	0.0	0.0
	Shallow	22.2	19.4	16.7	19.4	0.0	0.0	0.0	0.0	0.0	5.6	19.4	2.8	0.0	0.0	0.0	0.0	0.0	0.0
	Convertible	5.6	13.9	25.0	22.2	5.6	19.4	5.6	5.6	0.0	11.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Available	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	5.6	8.3	5.6	2.8	2.8	2.8	0.0	0.0	0.0
	Ambivalent	0.0	16.7	13.9	11.1	0.0	8.3	5.6	16.7	5.6	13.9	16.7	41.7	16.7	13.9	11.1	5.6	13.9	5.6
	Weakly Unavailable	2.8	2.8	5.6	13.9	16.7	11.1	22.2	27.8	25.0	25.0	13.9	22.2	13.9	22.2	16.7	11.1	13.9	11.1
	Strongly Unavailable	0.0	0.0	33.3	30.6	77.8	61.1	66.7	50.0	69.4	33.3	16.7	25.0	66.7	61.1	69.4	83.3	72.2	83.3

Table 6.8: Perceptions of angling groups regarding which freshwater fish species are threatened by common threats to freshwater fish populations. The results are expressed as percentage of respondents within the respective groups. The “Respondents” column reports the percentage of each group that indicated that at least one species was threatened by the respective threats.

		Sawfin	Yellowfish	Whitefish	Sandfish	Moggel	Cape kurper	Cape galaxias	Redfin	Rock catlets	Respondents
Habitat degradation	Bank	33.3	46.7	33.3	40.0	26.7	33.3	33.3	40.0	40.0	53.3
	Artlure	42.1	42.1	42.1	42.1	26.3	42.1	31.6	42.1	42.1	47.4
	Fly	75.0	80.6	77.8	72.2	61.1	66.7	69.4	80.6	77.8	83.3
	Average	50.1	56.4	51.1	51.4	38.0	47.4	44.8	54.2	53.3	
Water pollution	Bank	53.3	73.3	53.3	53.3	46.7	53.3	53.3	46.7	46.7	73.3
	Artlure	57.9	63.2	63.2	57.9	52.6	57.9	63.2	63.2	63.2	68.4
	Fly	75.0	75.0	75.0	69.4	69.4	75.0	75.0	75.0	72.2	80.6
	Average	62.1	70.5	63.8	60.2	56.2	62.1	63.8	61.6	60.7	
Water abstraction	Bank	46.7	53.3	46.7	46.7	40.0	40.0	40.0	40.0	40.0	53.3
	Artlure	31.6	36.8	31.6	31.6	26.3	26.3	31.6	31.6	31.6	36.8
	Fly	80.6	83.3	80.6	72.2	63.9	69.4	69.4	69.4	72.2	88.9
	Average	52.9	57.8	52.9	50.2	43.4	45.3	47.0	47.0	47.9	
Pesticides	Bank	66.7	73.3	66.7	73.3	60.0	66.7	73.3	73.3	73.3	73.3
	Artlure	52.6	52.6	57.9	52.6	52.6	52.6	57.9	57.9	57.9	63.2
	Fly	75.0	77.8	77.8	69.4	69.4	72.2	75.0	77.8	69.4	83.3
	Average	64.8	67.9	67.4	65.1	60.7	63.8	68.7	69.7	66.9	
Non-native plants	Bank	20.0	26.7	20.0	26.7	13.3	26.7	26.7	26.7	26.7	33.3
	Artlure	26.3	31.6	26.3	26.3	21.1	31.6	26.3	26.3	26.3	31.6
	Fly	61.1	61.1	58.3	58.3	55.6	58.3	61.1	63.9	55.6	69.6
	Average	35.8	39.8	34.9	37.1	30.0	38.9	38.0	39.0	36.2	
Non-native fish	Bank	26.7	33.3	46.7	33.3	33.3	33.3	40.0	40.0	33.3	60.0
	Artlure	31.6	42.1	36.8	26.3	10.5	42.1	42.1	36.8	26.3	47.4
	Fly	77.8	83.3	75.0	61.1	47.2	69.4	75.0	83.3	72.2	88.9
	Average	45.3	52.9	52.8	40.3	30.4	48.3	52.4	53.4	44.0	

CHAPTER 7. MANAGEMENT OF NON-NATIVE FISH

7.1 Introduction

The overarching goals of conservation are to ensure that the biosphere can continue to renew itself and provide the means for all life; to ensure human survival and well-being; and to retain potential evolutionary pathways (Siegfried and Davies 1982). Conservation, therefore, traditionally focuses on the indefinite retention of the structure, function and composition of biomes at landscape, community or ecosystem, species or population, and genetic levels (Noss 1990). Protected areas have been the core element of terrestrial conservation efforts for well over a century (Abell *et al.* 2007). A fundamental assumption of protected areas is that they will afford protection to the biota within their boundaries. For many fish species, components of their life cycles (eggs, larvae, juveniles, or adults) may not remain within the boundary of the freshwater protected area (Koehn 2003). Boundary-based conservation zones traditionally applied in terrestrial conservation are not appropriate to protect rivers, freshwater fishes and other aquatic organisms, because fish move within river systems and are largely dependent on upstream supplies of water and food (Koehn 1995, Cowx and Collares-Pereira 2002, Crivelli 2002). Large-scale catchment management approaches are required in river and freshwater fish conservation, since river reaches are substantially impacted by changes in their upstream catchments (Koehn 1995, Cowx and Collares-Pereira 2002, Crivelli 2002, Abell *et al.* 2007, Nel *et al.* 2007, 2009a, b, 2010)

The conservation of the native freshwater fishes of the Cape Floristic Region is the central theme of this thesis. Freshwater fish populations are susceptible to extirpation through habitat degradation; water pollution; over-abstraction of water; habitat fragmentation by obstructions in the river (e.g. weirs and dams) and the presence of non-native fish species within the water body, particularly piscivores (Bruton 1995, Maitland 1995, Richter *et al.* 1997, Cowx 2002, Skelton 2002). In the Cape Floristic Region, there is a strong correlation between the presence of predatory non-native fish and the absence of native species (Impson 2007, Tweddle *et al.* 2009). The native freshwater fishes are generally small and adapted evolutionary to variations in water flow and quality driven by the variability of the surrounding Mediterranean-climate. They are, however, poorly adapted to the presence of non-native piscivorous fish. Due to the nature of the rivers in the Cape Floristic Region and their native fishes, addressing habitat degradation without suitably addressing the presence of non-native fishes will not contribute to the conservation of the native fishes. The extent of non-native fish invasion is so large and their presence so damaging to the native fish populations that the first step in any native fish conservation programme in the Cape Floristic Region has to be to adequately manage and reduce the populations of non-native fishes. Researchers and conservation officials in the Cape Floristic Region agree that non-native fish are the most significant threat to the long-term survival of their native freshwater fish assemblages (Barnard 1943, Coke 1988, Skelton *et al.* 1995, Impson and Hamman 2000, Skelton 2000, Impson *et al.* 2002a, b, Skelton 2002, Cambray 2003a, b, Impson 2007, Tweddle *et al.* 2009). Because the native fishes of the Cape Floristic Region cannot persist in the presence of predatory non-native fish, the remaining native fish populations can only be conserved, and their ranges increased, through the control, management or elimination of the non-native fish species (Impson 2007). This thesis, therefore, concentrated on the management of non-native freshwater fishes as a mechanism for the conservation of the native freshwater fishes in the Cape Floristic Region.

7.1.1 Aims of the Thesis

This thesis evaluates the conservation of native freshwater fishes in the Cape Floristic Region through the management of non-native fish by engaging with the following questions:

- What is the status of the non-native fish invasion in the Cape Floristic Region, South Africa?
- Are there commonalities in the freshwater fish introductions in Mediterranean-climate regions and the outcomes of these introductions?
- Does a species introduction contribute more to the decline of an endemic-range restricted fish than habitat degradation?
- Can freshwater angling groups specifically targeting non-native species be convinced to target native fish?
- What mechanisms for the management of non-native fish could be adopted for the Cape Floristic Region?

7.2 Contributions from this Thesis

A number of studies have been presented in this thesis and the lessons learned from these studies for the conservation of native fishes and the management of non-native fishes in the Cape Floristic Region is summarised here.

In Chapter 3, a meta-analysis estimated that more than 90% the river habitat in the three major catchments has been invaded by non-native fish and that catchments covering less than 1% of the area of the Cape Floristic Region have no recorded non-native fish introductions.

The majority of catchments contain four or more non-native species, while the major river systems of the Olifants-Doring, Berg, Breede and Gouritz containing 10 or more non-native species. This large-scale invasion has resulted in the fragmentation of the native fish populations. As a result, the majority of the native fishes of the Cape Floristic Region continue to be threatened by the presence of non-native fish.

In Chapter 4, profound taxonomic and functional changes to the freshwater fish assemblages of the Mediterranean-climate regions as a result of the introduction of non-native fish were identified. The species selected for introduced exhibited phylogenetic preferences with more than 90% of species introduced originating from five taxonomic orders (Cypriniformes, Cyprinodontiformes, Perciformes, Salmoniformes and Siluriformes). The pathways for introductions were consistent across all Mediterranean-climate regions. The results show strong evidence of on-going taxonomic and functional homogenization of the freshwater fish faunas. Characteristics suitable for inclusion in risk assessment databases as criteria that predicting successful establishment in Mediterranean-climate regions were identified.

In Chapter 5, the difficulties associated with attempting to identify the reasons for the decline of a critically endangered fish, even for a relatively simple system, were demonstrated. Water quality, pesticide exposure, instream and riparian zone habitat, and dietary overlap between native and non-native species were explored. The results of the study were inconclusive. The results do indicate that native and non-native species can co-occur in the complex habitat of one tributary, but not in the simple habitat of the other.

In Chapter 6, a social survey of freshwater anglers showed that angling for non-native species is important to the anglers and that they are not likely to switch to angling for native species, with the possible exception of Clanwilliam yellowfish. The anglers considered the conservation of native fishes extremely important and less than half believed that the conservation authority were doing a good job in conserving native fishes. The results indicate that support for the conservation authority’s proposed piscicide project is low but that it could increase support for conservation projects by using larger cyprinids as flagship or umbrella species.

7.3 Management of Non-Native Freshwater Fish

The National Environmental Management: Biodiversity Act (NEM:BA) provides enabling legislation, promoting the management of non-native species by designating non-native species into three categories: **prohibited species** which may not be imported into the country, **exempted species** which may be freely traded within the country without permit, and **invasive non-native species** which must be controlled by means of a management plan (van Rensburg *et al.* 2011). At national level, critical biodiversity areas for native freshwater organisms have been mapped through the National Freshwater Ecosystem Priority Areas (NFEPA) project (Driver *et al.* 2011, Nel *et al.* 2011). Section 28 of South Africa's National Environmental Management Act (NEMA) obliges conservation authorities to remedy degradation of environments harmed by non-native and/or invasive species. Yet, native fish populations continue to decline, and non-native fish are continually being introduced into new areas. In order to begin addressing the management of non-native fish in the Cape Floristic Region, it is important to understand why there has been so little progress made in conserving the native freshwater fishes of the Cape Floristic Region when the conservation of other taxa has advanced since the completion of the CAPE Project.

7.3.1 Why is the conservation authority failing?

Many reasons for the lack of progress in the conservation of freshwater fishes in the Cape Floristic Region have been offered, including the lack of capacity, lack of funding, lack of public awareness, inadequate reserve networks, and legislative deficiencies (Impson *et al.* 2002b). These inadequacies have affected the operational capabilities of the provincial conservation authority by reducing its ability to: undertake regular survey work, undertake priority research projects, purchase land to conserve freshwater aquatic systems, undertake or implement species or habitat recovery plans (Impson *et al.* 2002b). Poor communication and co-operation with riparian landowners and angling clubs, insufficient public awareness campaigns, and a poor enforcement capability are further reducing the effectiveness of the conservation authority (Impson *et al.* 2002b). In the five years following the above analysis, the situation had not changed (Impson 2007). The only significant advance has been in the communication and co-operation between the conservation authority and other stakeholders, riparian landowners, and angling groups (Impson 2007). The public participation process for the Environmental Impact Assessment of the CAPE Alien Fish Eradication Project has contributed largely to this improvement (Marr *et al.* in press), but see Section 6.4.

The question remains: Why is there no active attempt to conserve native fish species in the Cape Floristic Region? The situation is complicated by the fact that the biome stretches over two provinces, each with its own conservation authority. For the Western Cape, the conservation authority is CapeNature, which has been a partner in the CAPE Project: their fisheries scientists have been prominent role players in the CAPE Alien Fish Eradication

Project (Impson 2007), and previously in the River Health Programme. In the Eastern Cape, conservation is managed by three entities: South African National Parks manage conservation within the province's national parks (e.g. Addo Elephant Park), Eastern Cape Parks Board manage conservation within other parks and protected areas, while the Eastern Cape Department of Environmental Affairs manage conservation in all areas outside of parks and protected areas. Overall, the Eastern Cape conservation authorities are not well equipped to implement freshwater fish conservation measures. Although the situation in the Eastern Cape should not be ignored, the following discussion is biased towards the Western Cape due to a greater familiarity with this region.

There is a clear lack of capacity for freshwater fish conservation in CapeNature. This can be attributed to a lack of funding for freshwater fish conservation, but is more likely as a result of CapeNature having no clear documented conservation plan for the native freshwater fishes. A conservation plan has been outlined by Impson *et al.* (2002a), but it has many deficiencies, such as the absence of defined goals or no assigned responsibilities for implementing the plan. The absence of clearly defined responsibilities has resulted in a lack of accountability within freshwater fish conservation. A lack of definition of roles within freshwater fish conservation in the Western Cape is an issue that needs to be addressed. Who is responsible for developing conservation plans - the South African Biodiversity Institute (SANBI), South African Institute for Aquatic Biodiversity (SAIAB), or provincial conservation authorities? What role do research institutions, such as universities, play? What about private consultants? It is important that the roles of the respective organizations are defined such that progress towards the conservation of native fish can be measured and the respective role players held accountable for their responsibilities.

7.3.2 Roles and Responsibilities

The mandate for conserving and managing freshwater ecosystems is shared between the Department of Water Affairs (DWA), the national custodian of inland water resources, and the Department of Environmental Affairs (DEA), the national custodian of biodiversity (Driver *et al.* 2011). The DWA responsibilities include incorporating freshwater ecosystem priorities through integrated water resource management and instituting programmes to monitor the condition of freshwater ecosystems (Driver *et al.* 2011). The DEA's responsibilities include co-ordinating the implementation of the National Biodiversity Framework, leading the process of listing of threatened or protected ecosystems, supporting the publication of bioregional plans in terms of the Biodiversity Act, supporting the publication of biodiversity management plans in terms of the Biodiversity Act, leading the development and review of the National Protected Area Expansion Strategy, supporting the application of biodiversity stewardship tools and methods to conserving freshwater biodiversity as well as terrestrial biodiversity, and leading the development of regulations for non-native and listed invasive species in terms of the NEM:BA (Driver *et al.* 2011). These regulations deal with preventing the unauthorised introduction and spread of non-native species, managing and controlling invasive species to prevent, or minimise harm to the environment, and where possible and appropriate eradicating invasive species that may cause such harm (van Rensburg *et al.* 2011).

Under the Biodiversity Bill, the South African Biodiversity Institute (SANBI) has been appointed to monitor, report on, and coordinate and promote taxonomy of South Africa's biodiversity. SANBI's role and responsibilities include: co-ordinating a Freshwater Programme, reporting on the state of biodiversity (high-level national reporting: SANBI is not involved in on-the-ground monitoring efforts.), providing technical support to DEA in developing the list of threatened freshwater ecosystems and species, providing technical support to provinces involved in developing provincial spatial biodiversity plans, providing technical support to DEA and others involved in the development of biodiversity management plans for species and ecosystems, and providing technical support to DEA in developing regulations for non-native and listed invasive species (Driver *et al.* 2011). For freshwater fish, SANBI have delegated their responsibilities to the South African Institute for Aquatic Biodiversity (SAIAB) (P.H. Skelton, SAIAB, pers. comm.).

Provincial conservation authorities, such as CapeNature, are the major line managers responsibility for ecosystem management and conservation. Their responsibilities include: monitoring the condition of freshwater ecosystems in collaboration with the River Health Programme, filling in gaps in knowledge of freshwater ecosystems and species, for example properly surveying the distribution of threatened fish populations, initiating and/or participating in the development of biodiversity management plans in terms of the NEM:BA, for priority freshwater ecosystems and species, contributing a leading role in the work of Catchment Management Agencies, commenting on development applications, including environmental impact assessments, mining and prospecting applications, and recreational fishing and aquaculture permit applications, participating in the determination ecological flow requirements and the classification of water resources, consolidation and expansion of the provincial protected area network through biodiversity stewardship programmes, interacting with Working for Water, Working for Wetlands, and LandCare to direct these programmes towards rehabilitating freshwater ecosystem priority sites (Driver *et al.* 2011).

The roles and responsibilities of Catchment Management Agencies in managing and conserving freshwater ecosystems include: ensuring that freshwater ecosystem priorities are reflected in the development and implementation of Catchment Management Strategies, and contributing to the monitoring of freshwater ecosystems (Driver *et al.* 2011).

7.3.3 Institutional Limitations

Most provincial conservation authorities lack the financial resources and are inadequately staffed to fulfil their roles and responsibilities in freshwater fish and ecosystem conservation (Driver *et al.* 2011). Each provincial conservation authority ideally requires at least six to eight aquatic scientists and technicians, with expertise in limnology, hydrology, fish biology, aquatic invertebrate biology, aquatic plant biology and other aspects of aquatic ecology, in order to play an effective role in managing and conserving freshwater fish and ecosystems (Driver *et al.* 2011). This shortage can be alleviated by promoting freshwater fish research and conservation as a career in the Cape Floristic Region. An alternative to fill the short-term need for freshwater fish researchers in South Africa could be to establish training or internship arrangements with US, Australian or European organizations (e.g. through Fulbright Scholarships or internship agreements with the American Fisheries Society, or the US Fish and Wildlife Services). Should such arrangements be established, international aquatic scientists and technicians could be drafted in to help overcome the shortage in trained scientists in the CapeNature, or conservation authorities elsewhere in South Africa.

7.3.4 Non-native Fish Management Goals

The management of non-native fishes requires a clear goal that defines the problem, delineates the intervention area, identifies the major stakeholders, considers the factors influencing the problem, and presents the desired outcome (Bomford and Tilzey 1997). Such a goal is invaluable in directing the efforts of implementing authorities, funding agencies, and stakeholders towards an agreed outcome (Bomford and Tilzey 1997, Wittenberg and Cock 2001, Hulme 2006). For example, the 10-year goal of non-native fish management in the Murray-Darling Basin in Australia is to reduce the distribution and abundance of all non-native species by 30% (Murray-Darling Basin Commission 2004). Once a goal has been agreed upon, the management options available can be identified and the most appropriate strategy for achieving the goal selected considering the technical options available, the ease with which specific species can be targeted, the risks associated with the management options, and the likelihood of success (Bomford and Tilzey 1997, Wittenberg and Cock 2001, Hulme 2006). The performance of the project can thus be measured, evaluating the efficiency and effectiveness of the techniques employed in achieving the stated goals (Bomford and Tilzey 1997).

Fisheries researchers and conservation officials need to agree on an achievable goal and time frame to achieve the goal. This goal would then be negotiated with the major stakeholders, such as representative of recreational angling aquaculture and the ornamental fish trade. An example of such a goal could be: to prevent the establishment of any further populations of non-native fishes in natural watercourses in the Cape Floristic Region and to reduce the abundance and distribution of all non-native species by 30% in twenty years. Maps outlining the areas designated for the removal of non-native fishes could be presented with intermediate goals and timelines such that the respective stakeholders understand where sacrifices may need to be made. Each intervention should be justified by the listing benefits that would accrue for the conservation of native species as a result of the implementation. Prioritising of the interventions is recommended.

7.3.5 Non-native Fish Management Policy

A non-native fish management policy details the position of the conservation authority on the management of non-native fish, the beneficial use of non-native fishes, and the consequences of not adhering to the policy, for example, the Upper Colorado River Endangered Fish Recovery Program (Upper Colorado River Endangered Fish Recovery Program 2004) or the Queensland Pest Fish Strategy (MacKenzie 2003). The policy could outline the extent to which non-native species would be permitted, and which non-native species should be removed from all waters if possible. The role of recreational angling and aquaculture in the management of non-native fishes could be outlined. “Codes of Conduct” could be outlined to provide consistency across angling codes and aquaculture. The policy could delineate protected non-native fish populations, such as important angling populations. CapeNature is in the process of drafting policies for the management of non-native fishes, but, after three years, nothing has been produced for comment by stakeholders. The zones established for non-native fish through the NEM:BA mapping process have been adopted by CapeNature to inform the issue of permits.

7.3.6 Prevention of Introductions

Preventing the entry of known, or potentially, invasive non-native species is widely accepted as being more effective than attempting to manage species once they have been introduced

(Park 2004, Hulme 2006, Finnoff *et al.* 2007). The precautionary principle could be invoked and the entry of any species that might establish in the country should be prohibited on the basis of defendable risk assessments (Leung *et al.* 2002, McDowall 2004). Preventing a potentially invasive species entering a country is difficult and the majority of custom officials are not adequately trained to identify fish species (Lintermans 2004, McDowall 2004). Identifying the primary vector and pathways for species introductions of entry is key to preventing future introductions (Simberloff *et al.* 2005, Hulme 2006).

An important component in a prevention strategy is risk assessment. Risk assessment is used prior to introduction to assess the potential impacts of the proposed species introduction (Townsend and Winterbourn 1992). A risk evaluation strategy, such as that employed in the United Kingdom, usually encompasses four components: risk identification, risk assessment, risk management and communication, and risk review and reporting (Copp *et al.* 2005b, c). The Fish Invasiveness Screening Kit (FISK) was developed to provide a screening tool for freshwater fish introductions. Continued development has resulted in FISK becoming a tool to aid decision- and policymakers in assessing and classifying freshwater fishes according to their potential invasiveness (Copp *et al.* 2009). Many countries, including South Africa, use black and white lists for prohibited and permitted species. These are updated periodically using expert opinion. Many of the species included on the lists have not been subjected to formal risk assessments. The recent revision of the freshwater fish black and white lists was completed without any risk assessment being formally conducted. Implementing a formal risk assessment tool such as FISK will increase the rigor of risk assessments and reduce the risk of permitting the trade or introduction of potentially invasive freshwater fishes.

NEM:BA requires the listing of permitted and prohibited species for South Africa (van Rensburg *et al.* 2011). Frequent revision to the list of permissible species is required, with newly-identified potentially invasive species being included on the lists. However, the process of listing a new species on either the prohibited or permitted lists is arduous. Species not included on either list require a risk assessment before their import into the country will be permitted. An alternative philosophy could be to scrap the permitted and prohibited lists and demand a risk assessment for all taxa prior to import being permitted. The results of the risk assessment would then be used to allocate the taxa to either the permitted or prohibited lists.

Many potentially harmful non-native species have established populations outside of their native ranges, yet may be far from achieving their potential geographic distribution. Preventing or slowing the secondary spread of known and established non-native species requires different set of goals, strategies, and target audiences than measures required to restrict the import of new species (Vander Zanden and Olden 2008).

Once non-native fish have established in a catchment, they are usually spread illegally to neighbouring catchments (Clavero and García-Berthou 2006). Anglers have been guilty of being major vector for the secondary spread of non-native fishes (Rahel 2004, Johnson *et al.* 2009) and this trend is unlikely to be stopped unless the risk of being caught, and the consequences of being caught, are sufficient to deter the activity. Education will prevent the spread of non-native fishes by the naive, but is unlikely to deter the malicious. Most anglers involved in the illegal spread of non-native fishes are aware of the consequences of their actions (Johnson *et al.* 2009). The current NEM:BA legislation states that persons responsible for the introduction of non-native species are liable for the costs of eradicating them (Beamesderfer 2000). This may be suitable for plant infestations, but for fish introductions, this may be difficult to enforce. If the liability for eradication of introduced fish can be enforced, this may be sufficient a deterrent to prevent anglers engaging in the illegal introduction of non-native fishes.

7.3.7 Early Detection-Rapid Response Management Plans

No matter how effective prevention programmes are, there is a high probability that one or more non-native species will be introduced. The precautionary principle requires that action be taken to eradicate potentially invasive species as soon as they are detected (Wittenberg and Cock 2001, McDowall 2004). Detection of new introductions at low population levels is often difficult, especially in aquatic systems (Collares-Pereira and Cowx 2004, Hulme 2006, Mehta *et al.* 2007) and the costs associated with detection programmes are high (Finnoff *et al.* 2007). Further, there is often a substantial time lag between introduction and detection (Crooks and Soule 1999, Crooks 2005). In principle, early detection and rapid response should be straightforward. In practice, for all but economic pests and vectors of disease, the required rapid response is often glacially paced. Rapid response management entails an assessment of the ecological and economic risks as soon as an introduced species has been detected. The deployment of remedial actions, if warranted, including initial containment of the individuals, followed by steps taken to eradicate them. An effective rapid response system requires a sound scientific basis upon which to plan actions, the tools and protocols with which to respond, and the capacity and resources to achieve its goals (Hulme 2006, Thomas *et al.* 2009). Ultimately, successful non-native species management also requires appropriate enabling legislation to authorize, implement, and fund detection and rapid response (Thomas *et al.* 2009).

For fish introduction into river networks, early detection is unlikely, or highly improbably (Collares-Pereira and Cowx 2004). However, rapid response is a requirement once a new introduction has been detected. Britton *et al.* (2011a) outlined a modular assessment tool for the management of non-native fish populations under an early detection-rapid response framework. The assessment scheme consists of four modules:

1) Prioritisation of introduced fish according to risk and distribution. Does the prioritisation suggest further management action is required?
2) Assessment of the ecological and socio-economic risk to the receiving ecosystem. Determine the required management action.
3) Assessment of the impacts of the required management action, mitigation steps and assessor confidence. Are these acceptable for the action to proceed?; and
4) Assess long-term cost-benefit of the management action compared to alternatives. Is the intervention an effective long-term use of conservation resources?

Clear guidelines for acceptable intervention strategies and techniques are required. Rapid-response management should be free of bureaucratic red tape with clear rubric for decision-making, selection of intervention technique, responsibilities, and liabilities. The modular approach outlined by Britton *et al.* (2011a) could be modified for application in South Africa.

7.3.8 Review of Eradication/Control Options

Where non-native species have already become established, active management needs to focus on reducing their impacts and preventing further spread (Saunders *et al.* 2002, Britton *et al.* 2011b). Eradication is preferable and is usually more cost-effective than long-term mechanical control (Bomford and O'Brien 1995, Bomford and Tilzey 1997, IUCN 2000) particularly for recent introductions, or where the species has been spatially constrained (Britton *et al.* 2011b). Eradication should be attempted only where it is ecologically feasible and has the necessary financial and political support, however (IUCN 2000). Sustained mechanical removal has been successful only at small scales (Britton *et al.* 2011b). Where eradication is not feasible, control is the next best alternative. Control programmes using mechanical removal techniques (e.g. electrofishing or netting) are generally effective in suppressing the target population, or reducing their recruitment (Britton *et al.* 2011b). Control programmes should be implemented in areas of highest value for native biodiversity (IUCN 2000). Eradication and control of non-native fishes remain constrained by their lack of selectivity and the challenges of treating large spatial scales effectively (Britton *et al.* 2011b). While large-scale eradication of non-native fish may not be technically feasible at present, small-scale projects to eradicate non-native fish from native fish sanctuaries, or to expand the ranges of critically endangered species, can be successfully completed with the current available technologies.

Habitat Restoration

Habitat restoration is often proposed as a means of restoring native fish assemblages and of restricting invasions by non-native fish species (Goodrich and Buskirk 1995). Habitat restoration, including re-instituting natural flow regimes, can alter fish community structure, population demographics, and the relative abundance of species (Bain *et al.* 1988, Gido and Propst 1999), creating conditions that promote the survival of native fishes over non-natives (Kennedy *et al.* 2005, Scoppettone *et al.* 2005). Habitat restoration should address the bottlenecks in the life cycles of native fish species (Cowx and van Zyll de Jong 2004). Habitat alone, however, will not ensure the recovery of native species where non-native species are the dominant driver of the native species' decline (Townsend and Crowl 1991, Chadderton 2003, Marsh and Pacey 2005). In many regions, native fishes are able to survive and successfully complete all their life functions in altered and managed habitats, provided that they are free of introduced predatory fishes (Marsh and Pacey 2005).

Barriers

Barriers have frequently been used to prevent non-native species spreading in aquatic systems (Moyle and Sato 1991, Rinne and Turner 1991). However, ensuring long-term survival of native species requires that sufficient ecological and genetic resources are present in populations living above the barrier to sustain or enhance the populations (Saunders *et al.* 1991, Harig and Fausch 2002). Therefore, the use of barriers requires a detailed knowledge of habitat requirements and movement patterns of the native species (Harig and Fausch 2002, Novinger and Rahel 2003, Peterson *et al.* 2008b, Peterson *et al.* 2008a). The decision to install a barrier may require a trade-off between the threats posed by the non-native species and those resulting from isolation above the barrier (Fausch *et al.* 2009).

Mechanical Removal

Mechanical removal techniques constitute a wide suite of capture methods (such as netting, electrofishing and angling) that are labour-intensive and costly. Some mechanical removal techniques, such as gill nets, can be fatal, whereas electrofishing can cause physical damage such as internal haemorrhaging and spinal damage if used inappropriately (Snyder 2003). The majority of techniques, such as trap nets and angling, are non-fatal, allowing non-target species to be returned to the river.

Gill nets and traps designed for fish ways have been used to remove non-native species in California and Australia (Knapp and Matthews 1998, Stuart *et al.* 2006, Knapp *et al.* 2007). Electrofishing is popular in the USA and Europe, achieving a small measure of success in fish control programmes (Kulp and Moore 2000). With the exception of very short, narrow streams with simple habitat, electrofishing is unlikely to be successful in completely eradicating non-native fish (Shepherd 2005, Meyer *et al.* 2006). Eradication of non-native fish using electrofishing is expensive because of high labour requirements (Meyer *et al.* 2006). Although angling is acknowledged as a potential mechanism in controlling species distributions (Beamesderfer *et al.* 1996), its success has rarely been assessed in the literature (Paul *et al.* 2003).

The effectiveness of all mechanical removal techniques is density-dependent, increasing exponentially with density. However, the effort required to remove additional individuals increases exponentially as densities tend towards zero (Bomford and Tilzey 1997). Mechanical removal is a long-term approach and requires a long-term commitment, political support and funding. As an eradication strategy, mechanical removal needs to create a bottleneck in the life cycle of the non-native species, only attained at high levels of effort. As a population control mechanism, mechanical removal techniques can be very effective and can maintain some non-native species below levels at which they become problematic. Understanding the response of the target species to mechanical removal is important, because removal may result in the rebound of the suppressed population (Goodrich and Buskirk 1995). Mueller (2005), in reviewing the mechanical removal of predatory non-native fish from the Colorado River (USA) concluded that, after 10 years of effort, involving the removal of more than 1.5 million predators at a cost of more than US$ 4.4 million, no positive response in the native fish assemblage could be detected. The author estimated that a reduction in non-native fish density of greater than 80% would be required before benefits for the native fish would be realised. However, the non-native fish would need to be maintained below these levels to prevent them rebounding; an expensive long-term commitment.

<u>Chemical Removal</u>

The use of piscicides such as Rotenone or Antimycin A to eradicate non-native fish species has been employed in the USA and the UK to successfully eradicate non-native or undesirable fish species (Finlayson *et al.* 2005, Britton and Brazier 2006, Britton *et al.* 2010b, Finlayson *et al.* 2010, Britton *et al.* 2011b), especially in large or complex habitats (Peterson *et al.* 2008a). Despite the importance of piscicides for the conservation of native fish, public misconception of piscicides and social resistance to toxicants can be significant because of the perceived danger to human and animal life (Finlayson *et al.* 2005). The use of a piscicide can have direct effects on aesthetics, air quality, the biota (invertebrates, amphibians, and fish), water quality, and recreation (Finlayson *et al.* 2005, 2010, Vinson *et al.* 2010). The magnitude of these impacts is often dependent on the piscicide selected, the treatment rate, the project size, and site-specific variables, such as water chemistry, flow, dissolved oxygen levels (Finlayson *et al.* 2005, 2010a, b). These impacts are typically of short duration, can be mitigated against, and are usually off-set by the long-term benefits to the native populations (Finlayson *et al.* 2005, 2010a, b).

Biological Control

Biological control consists of a suite of techniques designed to use biological agents to execute control over problem species (Simberloff and Stilling 1996). Techniques proposed for controlling or eradicating non-native fish include the introduction of predators (Carlander 1958, Santos *et al.* 2009); using pheromones as spawning disrupters or species-specific attractants or repellents (Sorrensen and Vrieze 2003, Sorrensen and Stacey 2004); viruses and pathogens (Crane and Eaton 1997, Thresher 2008); immuno-contraception (Hinds and Pech 1997); the release of sterile males (Bergstedt *et al.* 2003) and the use of recombinant genetics including sex- or stage-specific lethargy/sterility; gender distortion (daughterless or sonless genes); inducible mortality; pleitropic (having more than one effect); “Trojan” and “selfish” genes (Thresher 2008). The genetically based techniques are referred to as autocidal techniques (Thresher 2008).

Three genetically feasible trial projects are currently in development for fish: “daughterless” genes, female-specific sterility, and pleiotropic “Trojan” genes (Kapuscinski and Hallerman 1991, Kapuscinski and Patronski 2005, Cotton and Wedekind 2007, Thresher *et al.* 2007, Thresher 2008). Autocidal techniques may offer possibilities for the control, and possible extirpation, of non-native species where none currently exists (Thresher 2008). Theoretically, the genes can be species-specific; target particular life-history stages or one sex, and their effects can potentially be reversed if necessary (Thresher 2008). However, autocidal control programmes are inherently slow-acting, expensive and require high stocking rates over extended periods (> 10% of population stocked annually for > 50 years for carp) (Thresher and Kuris 2004, Thresher 2007, 2008, Thresher *et al.* 2009) and may raise ethical objections (Bax and Thresher 2009). Although these techniques are promising, they have only been developed for carp and have not been tested other than in theoretical studies or simulation models (Thresher and Kuris 2004, Thresher 2007, 2008, Thresher *et al.* 2009) and, therefore, cannot be considered viable options at present.

7.3.9 Species-Specific Management Plans

Species-specific management plans for non-native fishes are rare, because most water bodies contain more than one species of non-native fish. In Australia, the fisheries research community has focussed on the control of carp and a national management strategy has been developed for carp in Australia (Carp Control Coordinating Group 2000). Carp have been studied in detail and management strategies discussed to reduce their impacts (Koehn *et al.* 2000, Koehn 2004). In Alaska, northern pike *Esox lucius* Linnaeus, 1758 have been introduced into the south-central region where they did not historically occur. This is of concern to the local conservation authorities, as pike have the potential to impact on the economically valuable salmonid fishery of the area. A dedicated species management plan has therefore been prepared (Southcentral Alaska Northern Pike Control Committee 2007). Species management plans have been prepared for the ruffe (Ruffe Control Committee 1996) and Asian carp (Conover *et al.* 2007) in the USA.

Species-specific Management Plans are required under NEM:BA for invasive non-native species (van Rensburg *et al.* 2011). The non-native species targeted most by anglers all fall into this category. Guidelines for the development of species management plans are in the process of being developed by the DEA. Objectives outlined in typical species management plans include: coordination and implementation of a comprehensive management plan; prevention of further introductions; detection, monitoring and eradication of newly established populations; where feasible, control and eradication of established populations; mechanisms for informing stakeholders about the risks and impacts of the species; increasing and disseminating knowledge of the species through compiling of data and conducting research. Plans generally cover the current and past two years of efforts, as well as projections for the next five years. Annual revisions are recommended. The responsibilities for the development of species management plans have not been clearly stipulated to date.

Species-specific management plans are urgently needed for the species that pose the greatest threat to native fish stocks. Sharptooth catfish, smallmouth bass, spotted bass, bluegill sunfish, tench as well as smallmouth yellowfish, particularly require urgent attention. Anglers have recently introduced smallmouth yellowfish into the Breede River and rumours of introduction into the Berg River have been detected.

7.3.10 Recovering Reach By Reach

The simple management strategy for non-native freshwater fishes proposed by Clarkson *et al*. (2005) for the Colorado River could be applied to the rivers in the Cape Floristic Region. First, conservation authorities and other stakeholders, including SAIAB, need to identify rivers and catchments to be devoted exclusive to the conservation of native fishes, and those for non-native fish utilisation e.g. aquaculture and angling. Beginning in headwaters of rivers identified as conservation priorities, where non-native species exist, temporary barriers are installed and any native species above the barrier caught and retained in holding facilities. Non-native species above the barrier are chemically, or mechanically, removed, after which the native species are returned or introduced from appropriate stocks. The process is then successively repeated downstream, removing the upstream temporary barrier after each step. Through this process, interconnection between native fish populations is restored. The extent of the non-native fish eradication and the location of the lowest barrier would be determined on a case-by-case basis. Benefits will be temporary unless the intervention is sustained (Beamesderfer 2000). Non-native fish eradication programmes must ensure that the probability of re-invasion from up- or downstream reaches is significantly reduced with suitable barrier(s) in place to prevent re-invasion. Further, the risk of illegal release of non-native fish should be minimised. Education and publicity initiatives, in conjunction with simple and inexpensive monitoring protocols to detect non-native species in restored reaches, should be established to ensure long-term accomplishment of the programme's goals. Establishing privately owned protected areas as part of South Africa's Protected Areas Act, Act 57 of 2003, through stewardship initiatives is especially necessary. Through this process, the native freshwater fish assemblages in the rivers of the Cape Floristic Region, and other parts of southern Africa, could be restored reach by reach.

The first step has been accomplished in the Cape Floristic Region through the National Freshwater Ecosystem Priority Areas (NFEPA) project's mapping of critical biodiversity areas for freshwater fishes (Driver *et al.* 2011, Nel *et al.* 2011) and the zoning of catchments for listed invasive non-native fish species such as rainbow trout, carp and largemouth bass to accommodate the requirements of aquaculture and sports angling organisations under the auspices of NEM:BA. The catchments of the Cape Floristic Region have been zoned into three categories: zoned areas (where non-native fish can be stocked and used), amber areas (where risk assessments are required before non-native fish may be introduced or utilised), and no-go areas (where non-native fish may neither be stocked nor utilised). The zoned catchments are delineated on a species-by-species basis.

The CAPE Alien Fish Eradication Project is the next step in the process. A successful completion of the eradication of non-native fishes from the four rivers in this pilot project will establish protocols to follow in implementing the more comprehensive management strategy.

7.3.11 Stakeholder Buy-In

Securing stakeholder buy-in in the management of non-native fishes is essential to the success of any management plan. Education and defining the objectives of non-native fish management would go a long way towards establishing stakeholder buy-in. The key element to garnering stakeholder buy-in is trust. The stakeholders need to feel confident that the conservation authority is working towards a shared goal and that there are no hidden agendas. The conservation authority must be able to trust that the stakeholders will respect their conservation management and not deliberately introduce non-native species to new systems. The conservation authority must have integrity in dealing with the stakeholders and must maintain a high level of rapport without compromising conservation objectives. The stakeholder for their part must maintain integrity with the conservation authority.

The social survey showed that angling groups are supportive of native fish conservation and that most recognise the need for non-native fish management. However, the required trust for CapeNature is not there. CapeNature will need to carefully rebuild the trust with the angling groups in order to garner their support for conservation projects. Producing a clear conservation plan for the native fishes, with details of the required management of non-native fish populations, would constitute a major step towards rebuilding the trust.

7.3.12 Education and Public Involvement

The involvement of the general public and riparian land owners in the conservation of rivers and freshwater fishes, including the management of non-native fishes, is imperative to the success of any conservation strategy. In general, riparian land owners and the general public are likely to support conservation initiatives if they understand the objectives. Stakeholder support could be gained through communicating the objectives and the expected outcomes of conservation programmes, educating the stakeholders regarding the need for these programmes, and detailing the requirements from the stakeholders; including perceived sacrifices and beneficial outcomes. The participants in the conservation programme need to win the confidence of the stakeholders to ensure that the project is a success. An improvement in the communication of conservation projects is also necessary. Having an overarching conservation management plan, such as the Native Fish strategy of the Murray-Darling Basin, will help in communicating the objectives of conservation programmes and engender trust between the stakeholders and the conservation authority.

7.3.13 Conservation Lobby Group

In the USA, the Desert Fishes Council was formed by a group of concerned scientists and resource managers in 1969 to discuss what could be done to address the habitat destruction of the desert communities in the South-western United States (Pister 1990). Today, representation of more than 500 conservation agency and university scientists and resource managers, members of conservation organizations, and private citizens, all concerned with the preservation of aquatic ecosystem integrity, the Desert Fishes Council's function is to detect areas of weakness within the field of desert ecosystem conservation and utilize the full strength of its membership to compensate for bureaucratic inadequacies and to enhance government conservation programmes (Pister 1991). A freshwater fish lobby group similar to the Desert Fish Council could be established for the conservation of the freshwater fishes of the Cape Floristic Region. The broad function of the "Fynbos Fishes Council" could be to identify research needs of freshwater ecosystems in the Cape Floristic Region, detect areas of weakness in freshwater ecosystem conservation, and providing assistance in compensating for bureaucratic inadequacies, and enhancing government conservation programmes. This lobby group could also be responsible for raising the profile of native freshwater fish species and providing education material to the general public and land owners on freshwater fish and river conservation and restoration.

In addition to functioning as a lobby for freshwater fish conservation, this group could be responsible for generating educational materials and maintaining web sites informing the public and interested parties about freshwater fish and river conservation projects.

7.4 Summary and Recommendations

This thesis has explored the conservation of native freshwater fishes in the Cape Floristic Region, South Africa, through the management of non-native freshwater fishes. The conclusions from this work are summarised here with recommendations for further work.

7.4.1 Non-Native Fish Invasion in the Cape Floristic Region

A meta-analysis of the extent of freshwater fish invasion in the Cape Floristic Region showed that more than 90% of the river habitat in the three major catchments has been invaded by non-native fish and that catchments covering less than 1% of the area of the Cape Floristic Region have no recorded non-native fish introductions. This large-scale invasion has resulted in the fragmentation of the native fish populations. Furthermore, the majority of the native fishes of the Cape Floristic Region continue to be threatened by non-native fish. The meta-analysis was performed using data for entire catchments. While this was adequate for the purposes of this study to illustrate the magnitude of the freshwater fish invasion, repeating the study using quaternary catchment data is recommended. Data at quaternary catchment scale were not available, illustrating a need for the collection of data at this scale. The following studies are recommended:

- The taxonomy of the native freshwater fishes of the Cape Floristic Region should be completed and the new species already identified by genetic and morphological studies should be described.
- Data of the contact zones between native and non-native fishes should be collected for the five major catchments of the Cape Floristic Region: Olifants-Doring, Berg, Breede, Gouritz and Gamtoos Rivers.
- The extent of invasion should be re-evaluated when the fine-scale distribution data becomes available.
- Studies of the ecological impact of non-native species, particularly the South Africa sharptooth catfish and smallmouth yellowfish, should be conducted.
- The results of Jeremy Shelton's PhD evaluating the impacts of rainbow trout on native fish populations in the Breede River should be disseminated once the study has been completed.

7.4.2 Introductions in Mediterranean-climate Regions

The freshwater fish assemblages of the Mediterranean-climate regions have undergone profound changes, taxonomically and functionally. Changes in taxonomic similarity resulted in the regions and catchments moving towards a central point, i.e. they have become more similar. Changes in functional similarity resulted in the regions and catchments moving to a point not previously observed among Mediterranean-climate regions. A common set of species have been widely introduced, most originating from five taxonomic orders (Cypriniformes, Cyprinodontiformes, Perciformes, Salmoniformes and Siluriformes). Pathways for introductions are consistent across Mediterranean-climate regions and include: legal and illegal stocking, contamination of stocks, biological control of mosquitoes and aquatic plants, release of aquaculture species, and release of unwanted ornamental fish. Species successfully established in Mediterranean-climate regions and their functional traits could be included in risk assessment databases as species potentially invasive in the Cape Floristic Region. The following studies are recommended:

- The impact of the functional changes to freshwater assemblages on the structure, function and composition of aquatic ecosystems in Mediterranean-climate regions.
- Fine-scale studies of taxonomic and functional homogenization, including comparative studies within and across Mediterranean-climate regions.
- Evaluation of various sets of functional traits on the outcome functional homogenization studies.
- Comparative studies between taxonomic and functional homogenization between Mediterranean-climate regions and other climatic regions.

7.4.3 Non-native Species versus Habitat Degradation

A case study was used to explore whether the presence of non-native fish, Cape kurper, or habitat-related factors, could be identified as the primary factor preventing the recovery of a critically endangered, range-restricted Twee River redfin *Barbus erubescens* Skelton, 1974. The results of the study were inconclusive. The following studies are recommended to evaluate whether Cape kurper are preventing the re-colonization of redfin in the Suurvlei:

- Fine-scale habitat survey of the Suurvlei should be completed in conjunction with habitat utilization studies for redfin in the Hex River to determine whether suitable habitat for redfin is available in the Suurvlei. The Twee River redfin is too small to consider telemetry studies.
- Long-term monitoring of the water quality variables measured in the study (water chemistry, nutrients, aquatic macro-invertebrates, and some measure of pesticide exposure e.g. acetylcholine esterase inhibition) should be initiated to determine whether there are differences in water quality between the two tributaries.
- The structure of the aquatic invertebrate assemblages in the Suurvlei River should e determined to evaluate the potential impact of the proposed piscicide removal of Cape kurper to aquatic invertebrates.
- Observational laboratory and comparative growth experiments of the impact of Cape kurper on the behaviour and growth of endemic fish of the Twee River should be conducted *sensu* Marchetti (1999).
- *In situ* caged exclusion experiments in the Suurvlei River, or the removal of Cape kurper from the upper sections of the Suurvlei River should be conducted to determine whether redfin and galaxias can survive in the Suurvlei River.

- Stable isotope samples from the population of Cape kurper immediately below the barrier to invasion on the Middeldeur should be examined to determine whether there is any evidence of predation on redfin. A large population of redfin persists immediately above this barrier.
- The behavioural and ecological interactions between bluegill, Cape kurper and the endemic species of the Twee River.
- The interactions between Twee River redfin and Clanwilliam yellowfish in the lower Twee River should be evaluated. The current habitat utilization of redfin and stable isotopes could be used to determine whether yellowfish are competing with redfin or preying on them.

The bass and bluegill populations in the dams on the farm Tandfontein should be eradicated during the 2001/2 summer. These dams should be monitored regularly and managed such that they do not spill into the Middeldeur. Periodic eradication of non-native fish from these dams should be exercised. A survey of potential fish transfer routes into the dams should be conducted and the risk of re-introduction minimized through stewardship agreements.

7.4.4 Perceptions of Anglers

A social survey using the Conversion Model was conducted to determine the importance of non-native species to the three main angling groups: bank, artlure and fly anglers. The survey found that the majority of anglers were single-minded in their choice of target species, although about one third of the anglers pursued more than one target species. Among the native species, only the Clanwilliam yellowfish was of any interest to the anglers. Non-native species, particularly trout, bass and carp were important to fly, artlure and bank anglers respectively. Anglers considered that the conservation of native fishes to be extremely important, but less than half of the anglers believed CapeNature were doing a good job in conserving the native fishes of the region. Almost 80% of respondents were aware of piscicides as a means of eradicating non-native fishes, but less than 15% found the use of piscicides acceptable. The results of the social survey show that there scope for further such studies to understand the perceptions of anglers and other important stakeholders, such as riparian land-owners. The following studies are recommended:

- The perceptions of subsistence anglers should be evaluated.
- The perceptions of riparian land-owners towards freshwater fish conservation should be evaluated.

- The factors making the use of piscicides unacceptable to angling groups, and riparian land-owners, should be evaluated.
- Perceptions of how CapeNature can improve their native fish conservation efforts.
- Perceptions of the general public towards freshwater fish conservation.

The results of the above studies could be used to design appropriate education campaigns to raise the profile of native fish, and their conservation needs, among anglers, riparian land-owners and the general public.

7.4.5 Management of Non-native Species

Management of non-native fishes is recognised as the greatest challenge in freshwater fish conservation in the Cape Floristic Region. The majority of the native fishes of the region are threatened by the presence of non-native species and management of non-native fishes is fundamental to the long-term conservation of the native fishes. Many options are available to the conservation authority to improve their efforts to manage non-native fishes in the region. The following actions are recommended:

- Delineation of roles and responsibilities within the conservation agencies responsible for the conservation of native fish and the management of non-native fishes.
- Compilation of a comprehensive conservation plan for the native freshwater fishes of the Cape Floristic Region, including prioritization of the conservation actions, motivation for the proposed actions, impacts on non-native fish utilization, regular monitoring of priority native fish populations, budgetary requirements to complete the conservation plan, time-scales for implementation, responsibilities for implementation, and the expected outcomes. Annual progress reports should be compiled and the conservation plan revised every five years.
- Establishment of a native fish stock registry and a monitoring programme to annually visit the priority populations.
- Education programme to raise awareness of the perilous state of the native freshwater fishes of the region and to garner support for the conservation of these species.
- Publication of taxonomic revisions of the native freshwater fishes of the Cape Floristic Region.
- Completion of the pilot project to evaluate the use of piscicides for the eradication of non-native fishes.
- Establishment of long-term goals for the management of non-native fishes in the Cape Floristic Region.

- Establishment of Rapid-Response guidelines based on the approach of Britton *et al.* (2011a) to direct the response of the conservation authority following reports of a new introduction of non-native fish in the Cape Floristic Region. Adequate training in fish surveying, fish identification, and mitigation techniques, such as the use of piscicides, should be provided.
- Species-specific management plans should be compiled and published for sharptooth catfish, smallmouth yellowfish, carp, tench, smallmouth bass, spotted bass, largemouth bass, bluegill sunfish, mosquitofish, rainbow trout, banded tilapia and Mozambique mouthbrooder. The priorities are sharptooth catfish and smallmouth yellowfish.
- Establish small-scale non-native fish eradication projects aimed at using volunteers to remove non-native fish using mechanical removal techniques.
- Establish a long-term non-native fish management programme to systematically eradicate non-native fish from river reaches identified as priorities for the conservation of native species. This programme could be based on the barrier and chemical treatment strategy proposed by Clarkson *et al.* (2005).
- Establish incentives for land-owners to conserve native fishes and river habitat on their properties, or the formation of native fish sanctuaries on private land.
- Establishment of a lobby group to raise awareness for, and to champion the cause for freshwater fish conservation in the Cape Floristic Region.

Research projects could be established for many of these actions, such as evaluation of the effectiveness of mechanical removal techniques (including their ecological impact), comparative toxicity of piscicides to native and non-native species (including aquatic invertebrates), the behavioural and ecological interactions between native and non-native species, amongst others. There are many more actions that could be taken. Why have these not been initiated to date? The lack of coordination by the conservation authority is certainly a major factor. The lack of capacity within the conservation authority and the lack of accountability are further contributing factors. Conservation is prioritised by the perceived importance to the general public. In the Cape floristic Region, fynbos and birds are important in the public eye. Raising awareness of the need for freshwater fish conservation is required to ensure that it is raised in priority within the conservation authority.

7.5 Closure

The current conservation effort has neither reduced the rate of freshwater fish invasion in the region, nor improved the conservation status of the threatened native fishes. A new approach is necessary to change these outcomes. There are a number of options are available for conserving native fishes and managing non-native fishes in the Cape Floristic Region, most of which would take little effort to implement. The first step is to clarify the roles and responsibilities for the respective stakeholders in freshwater fish conservation. The compilation of a comprehensive conservation plan for the region is vital as such a plan will clarify the intentions of the conservation authority regarding the management of non-native fishes, demonstrate to potential funders that the conservation authority has clearly thought about its objectives improving the possibility of securing funds to complete the initial actions of the plan, and ensure accountability of the conservation authority to conserve the native fishes of the region. Once the conservation plan has been compiled, funding can be sought to address capacity issues within the conservation authority and for priority actions outlined in the conservation plan. Adequate performance in the completion of actions outlined in the conservation plan will increase funder confidence and further funding will be forthcoming to continue the implementation of the plan. Without a plan, the conservation of native fishes will continue to be ineffective and further populations will be lost to non-native fish invasion. With a conservation plan, funding should be forthcoming to implement priority conservation actions, stakeholders will understand the final outcome of the conservation initiatives, and the remaining native fish populations can be secured for future generations.

I must agree with Koehn and MacKenzie (2004) that it is not lack of knowledge that is impeding our progress in conserving native fish, and managing non-native fish, but a **lack of coordination**, or perhaps a **lack of will**, at all levels to acknowledge that **native fish are worthy of conservation** and **non-native fish are really a problem**.

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APPENDICES

The appendices contain supplementary material and the raw data used in analyses performed in this work. Some of the datasets are extensive and have been included electronically on the CD rather than as reams of unintelligible tables. Three Microsoft Excel spreadsheets containing the data have been included on the CD. The details of the files and data included on the CD are given in the respective appendices.

A.1 Geology of the Cape Floristic Region

To understand the biogeography of species, it is important to develop an understanding of the geological history the area under study. This subject is not the core of this thesis and a brief summary of the geological formation of southern Africa is presented. The information presented below was synthesised from McCarthy and Rubridge (2002), unless otherwise indicated.

Earth Scientists have proposed that the plates of the earth's crust have been swept together to form a number of super-continents that subsequently fracture and break up. The earliest recognised continents were Ur, Atlantica and Nena that formed between 3000 and 1500 Million years ago (Mya) and were swept together to form the first super-continent of Rodina[1] about 1100 Mya. Rodina broke-up to form Ur, Atlantica, Laurentia, Baltica and Siberia. During the break-up of Rodina about 700 Mya shallow seas formed over rifts in the Namaqua-Natal belt. Sediments accumulated in these seas forming primarily marine mudstones and sandstones of the Malmesbury Group with some marine limestones (e.g. Kango Group). Africa formed part of the super continent Gondwana, which was assembled from Atlantica and Ur between 700 and 500 Mya. About 500 Mya, what is currently southern Africa lay between East Antarctica, South America, and the now submerged Falklands Plateau. Gondwana subsequently collided with the remaining fragments of Rodina to form Pangaea (300 Mya). During the assembly of Pangaea, the rifts in the Namaqua-Natal belt closed and the sedimentary rocks were compressed, folded and metamorphosed. Granites (e.g. Cape Granites) intruded into these metamorphosed rocks formed part of the Pan-African network of metamorphic belts. Extension and rifting of the crust along what is now the southern Cape created a depression in which sedimentary rocks formed (The Cape Supergroup – The Table Mountain Group, The Bokkeveld Group and the Witteberg Group) between the Ordovician (495-433 Mya) and Carboniferous (354-290 Mya) periods.

Around 330 Mya (Carboniferous – Permian boundary) a subduction zone formed along the southern margin of Gondwana causing compression in the interior and the sedimentary rocks of the Cape Supergroup began to fold, forming a mountain range, the Cape Fold mountains. The Karoo Sea formed to the north of this mountain range where the Karoo Supergroup was laid down between the late Carboniferous and late Jurassic periods. This series ended with the formation of the Karoo Igneous Province (Drakensberg Group) which formed as Gondwana began to split up[2] at the end of

[1] There is a hypothesis by Rogers that an earlier super-continent, Columbia formed prior to Rodina (1800-1500 Mya).

[2] Pangaea broke up to form Laurasia and Gondwana just prior to the break up of Gondwana. Laurasia split up to form North America, Europe and Asia while Gondwana split up to form South America, Africa, Australia, India, Madagascar and Antarctica.

the Jurassic period with no major igneous events having taken place in the region since then (McCarthy and Rubidge 2005).

During the late Jurassic and Cretaceous, rivers eroded the Gondwana surface and by the beginning of the Cenozoic (65 Mya) following the split up of Gondwana, the southern African subcontinent became relative stabile and its form was essentially similar to that of the present day (Hendey 1983). The CFR is situated on a passive continental margin that was developed during the break-up of Gondwana and. The mountains of the CFR (the Cape Fold Mountains and outliers such as the Cape Peninsula and Piketberg) have been in existence throughout the Cenozoic (65 Mya) and their form has probably changed very little. The coastal lowlands have not been constant, however, being subjected to fluctuations in sea-level and their form being affected by deposition and erosion accompanying sea-level changes (Hendey 1983).

Globally, the sea-level has risen (i.e. marine transgression) and fallen (i.e. regression) as a result of plate tectonics and concentration of fresh waters in polar ice caps. Oceanic plate tectonics controlled global scale (eustatic) sea-level changes characteristic of the Cenczoic until the middle of the Miocene while glacio-eustatic sea-level changes controlled sea-levels thereafter as the volume of the polar ice-caps withdrew or added vast quantities of water to the oceans (Hendey 1983). Glacio-eustatic sea-level changes were generally of lower amplitude than tectonically controlled changes, but occurred in greater frequency with cycles during the Quaternary as short as 10 000 years. Sea-level changes influence climate (temperature and rainfall) with warm moist climates during periods of transgression and colder dry climates during periods of regression.

The magnitude of sea-level falls during the Tertiary are unknown or uncertain, although there is evidence that the Oligocene low stand, the most substantial fall in sea-level during the entire Cenozoic, was several hundred metres below the present sea-level (Figure A.1.1) and much, if not all, of the continental shelf was exposed at this time (Hendey 1983). During one of the recorded transgressions of the early Tertiary, the sea-level rose to in excess of 200m above the present sea-level, inundating the entire lowland areas of the CFR.

During the last glacial maximum, the sea-level to 120m below the present sea-level exposing wide areas of the continental shelf, particularly on the Agulhas Plain (Figure A.1.2) (Dingle and Rogers 1972). Comparable exposure of the continental shelf probably occurred during the earlier glacial maxima of the Pleistocene (Hendey 1983). During the interglacial periods of the Quaternary, including the present one, the sea-level stood at, or slightly above (up to 6m) the present sea-level.

The significance of the geological evolution of the CFR for freshwater fish biodiversity is that (1) land surface is ancient and have been stable for 3-400 My, (2) that biota is likely to have Gondwanan affinities (South America, Australia, India, South Africa); and (3) the sea level change and river capture have changed the drainage patterns, helping to explain some anomalous fish distribution patterns.

Figures

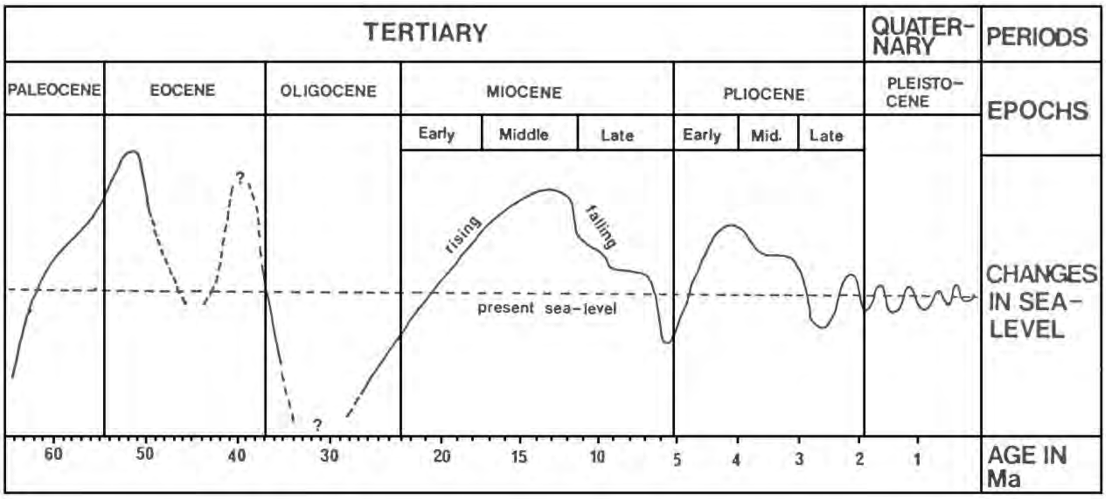

Figure A.1.1 – Diagrammatic representation of southern African seal-level changes during the Cenozoic Era (from Hendey (1983)). Ma = Mya.

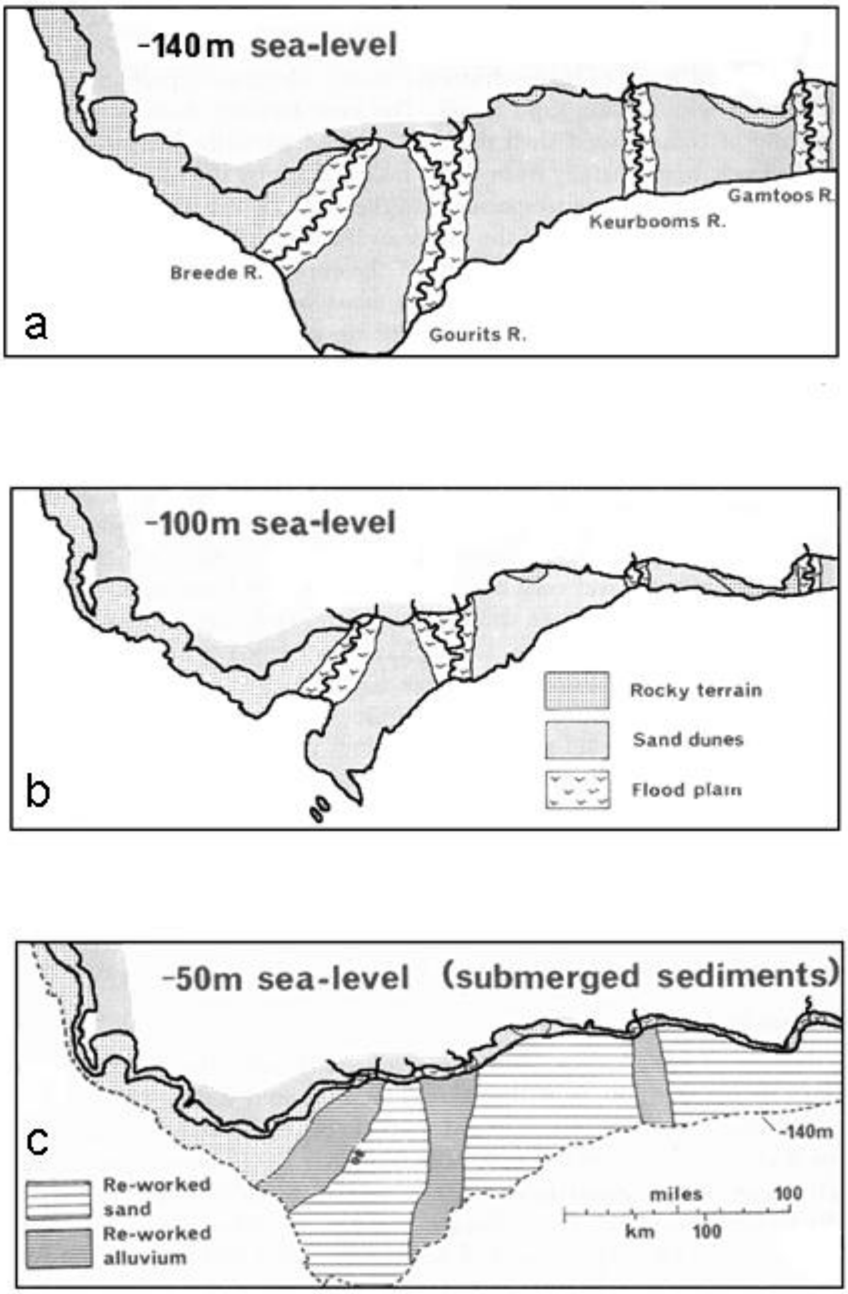

Figure A.1.2 – Palaeogeographies of the Agulhas Bank for selected sea-levels during the Pliestocene: a) -140m, b) -100m and c) -50m; showing superficial sediment types (Dingle and Rogers 1972)

A.2 River Systems of the Cape Floristic Region

The drainage history of South Africa has been described by Dingle and Hendey (1984), Partridge and Maud (1987), and Goudie (2005). Skelton (1994) provides a summary of how the drainage history has resulted in the current distribution of freshwater fish in the Zambezian and Southern (or Cape) Ichthyofaunal Province. As the southern African sub-continent was being defined following the break-up of western Gondwanaland beginning 144-138 Mya and completing in the early Cretaceous 113-97.5 Mya (Deacon 1983), two drainage patterns emerged – one drawing north and west from a cordon of high land on the elevated plateau and the other draining south and east from the outer slopes of the escarpment directly to the coast (Skelton 1994). During the late to middle Cretaceous two main rivers drained the interior of southern Africa. The southern river, the Karoo River (or Proto-Orange), had its source in the present Orange/Vaal basin, but its outlet at the present Olifants River mouth on the Atlantic coast (Figure A.2.1a). The northerly Kalahari or Molopo River drained southern Botswana and Namibia and entered the Atlantic via the lower Orange River (McCarthy and Rubidge 2005). By the early Cenozoic the lower Kalahari River had captured the upper part of the Karoo River due to uplift of the southern and eastern sub-continent margins at around 100–80 Mya (Goudie 2005), severing the connection between the Orange and Olifants systems (Figure A.1.1b). There is also evidence that the Berg River was connected to the Karoo-Olifants system during the Miocene and Pliocene epochs of the late Tertiary sub-era (Hendey 1983). The other two large rivers of the CFR, the Breede and Gouritz Rivers, were not linked during periods of low sea-level, as their course across the continental shelf have been kept separate (Hendey 1983) but smaller coastal rivers have been connected across the continental shelf during periods of lower sea levels (Swartz *et al.* 2007).

The relative stability of the subcontinent has resulted in the rivers of the CFR being largely unchanged since beginning of the Cenozoic (65 Mya) with river systems of the Hottentots-Holland mountains likely to have been confined to their present valleys through the Cenozoic (Hendey 1983). Climatic variations and sea-level changes over this period resulted in fluctuations in runoff and allowed certain neighbouring river systems to connect below the current sea-level and be separated at times of higher sea-level. The effect of sea-level fluctuation has been used recently to reconstruct the historical connections between the lineages of the Eastern Cape Redfin group (Swartz *et al.* 2007).

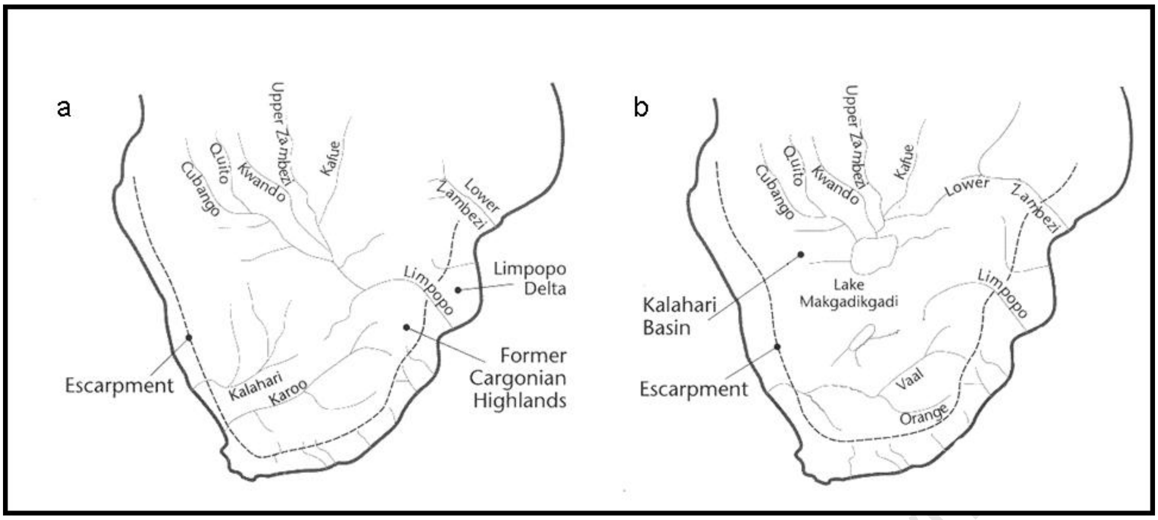

Figure A.2.1 Drainage evolution of the Orange-Vaal System showing a) the Karoo and Kalahari River systems shortly after the break-up of Gondwana and b) following the capture of the upper Orange River by the Kalahari River (adapted from McCarthy and Rubridge (2002))

A.3 The Cape Floristic Region and its Fishes

Presented in this Appendix are two manuscripts prepared for publication in the journal *African Journal of Aquatic Sciences*.

Marr, S.M. (in prep). Extent of the freshwater fish invasion in the Cape Floristic Region, South Africa: a preliminary estimate. *African Journal of Aquatic Science* **37**:in preparation.

Marr, S.M., N.D. Impson and D. Tweddle. (in press). Review of the proposal to eradicate non-native fish from priority river areas in the Cape Floristic Region of South Africa. *African Journal of Aquatic Science* **37**.

Extent of the freshwater fish invasion in the Cape Floristic Region, South Africa: a preliminary estimate

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Abstract
It has been estimated in popular literature that more than 90% of the freshwater habitat of the Cape Floristic Region has been invaded by introduced freshwater fish species. To date, this figure has not been evaluated. A meta-analysis of the extent of freshwater fish invasion in the Cape Floristic Region was performed to evaluate the extent the extent of freshwater fish invasion at a catchment level and at in three major river systems in the Cape Floristic Region. The results indicate that at least 90% the river habitat in the three major catchments is invaded and that catchments covering less than 1% of the area of the Cape Floristic Region are free from non-native fishes. These results highlight the importance of immediate conservation action to protect the remaining populations of freshwater fishes, most of which are listed as threatened in the most recent IUCN Red Data List.

Keywords: Cape Floristic Region, freshwater fish, conservation, invasion

Introduction

Although it is known among researchers that the freshwater ecosystems of the Cape Floristic Region have been extensively invaded by freshwater fishes, the magnitude of the invasion has not been previously documented. Estimates of the extent of invasion have been proposed, e.g. '90% of the rivers in the Cape Floristic Region have been invaded by non-native fish' (Hamman 2008), but none have the necessary data to support the figures presented. In order to conserve the remnant populations of native freshwater fishes, it is important that the scale of freshwater fish invasion in the region be communicated to the general public in an attempt to prevent further introductions. Therefore, a meta-analysis was performed to estimate the extent of freshwater fish invasion in the Cape Floristic Region based on the best available information. Due to the limited information available on the distribution of freshwater fishes in the Cape Floristic Regions, native and non-native, the data used in the estimate was based on published literature and expert opinion.

Methods

The estimate was compiled at two levels. Firstly, the extent of invasion was evaluated at a catchment level. Here we assumed that a species recorded as successfully established in the catchment had invaded the entire catchment. This is clearly not true as many species, such as the salmonids, have limited tolerance levels and will be restricted to regions of the catchments where their tolerance levels are not exceeded. However, most of the species introduced, such as cyprinids (Cyprinus carpio, Tinca tinca and Labeobarbus aeneus), centrarchids (Lepomis macrochirus and Micropterus sp.), silurids (Clarias gariepinus) and cichlids (Tilapia sparrmanii and Oreochromis sp.), have invaded extensive areas of the region. Secondly, the extent of freshwater fish invasion in three of the five major catchments of the Cape Floristic Region was compiled based on the length of river invaded. In addition, the length of river invaded was converted to an estimate of the available river habitat invaded using the Strahler stream order

Distribution Data

The non-native fish distributions for the Cape Floristic Region were compiled from a number of sources including Scott and Hamman (1984), de Moor and Bruton (1988), Bills (1999), Skelton (2001), Impson et al. (2002), Russell (2002), Russell and Impson (2006), Impson (2007) and Paxton and King (2009), supplemented with unpublished data from the South African Institute for Aquatic Biodiversity (E. Swartz, 2009, pers. comm.), CapeNature (D. Impson and A Turner, 2009, pers. comm.), Marine and Coastal Management (S Lamberth, 2009, pers. comm.), and the River Health Project. Only species not native to the Cape Floristic Region were included in the study with native translocated species being excluded from the analysis.

Catchments invaded

The Cape Floristic Region includes all catchments from the Olifants-Doring catchment in the west to the Swartkops catchment in the east (Figure 1). A list of introduced freshwater fish species for each catchment was compiled. Catchments without records, or where there was a high level of uncertainty in the records, were excluded from the study (Figure 1). Invasion maps were constructed for each introduced species using Arc View 3.3 GIS (Figure 2) and the maps depicting the number of species per catchment generated (Figure 3). The area of each catchment was calculated from the quaternary catchment shape files and the extent of invasion expressed as a percentage of the study area and the entire Cape Floristic Region.

River habitat invaded

To estimate the extent of invasion in a particular catchment, the length of river invaded was calculated for three of the five major catchments of the Cape Floristic Region: the Olifants-Doring, the Berg, and the Breede catchments. These three catchments represent 41.6% of the area of the catchments included in the above analysis and 35.6% of the Cape Floristic Region. The extents of known freshwater fish invasions were mapped on 1:250 000 maps for the selected catchments. Where it was not known whether the river reach contained non-native fishes, these reaches were designated as "Status unknown". The data were entered into Arc View 3.3 GIS package and the lengths of river invaded mapped on the 1:500 000 river cover shape files, available from the Department of Water Affairs (DWA). Only perennial rivers were considered in the calculation. The extent of invasion was calculated for river reaches where the invasion state was known. The extent of river habitat invaded was estimated by multiplying the length of each reach by the Strahler stream order raised to the power of n (where $1< n >2$) to account for the width and depth of the river. A value of 1.5 was chosen for comparison purposes. This correction was necessary to account for the difference in habitat available to fish between reaches with different stream orders.

Results

Reliable records were available for 42 of 58 catchments were identified for the Cape Floristic Region, accounting for 85.5% of the surface area of the region. A total of 16 species of freshwater fish have successfully established in the Cape Floristic Region (Table 1). Mozambique mouthbrooder Oreochromis mossambicus has been introduced into the largest number of catchments (30 catchments, 96.9% of the study area) but largemouth bass Micropterus salmoides has been introduced into catchments representing the largest area (97.1% of the study area, 26 catchments). Nine species have been introduced into catchments representing more than 90% of the study area: common carp; largemouth bass, smallmouth bass, spotted bass, bluegill sunfish, rainbow trout, Mozambique mouthbrooder and banded tilapia (Table 1 and Figure 2).

Catchments invaded

Ten or more non-native fish species have been introduced into five catchments: the Olifants-Doring; Berg; Eerste-Kuils; Breede and Gouritz catchments (Table 2). These catchments represent 67.7% of the catchment area included in the study. A further eight catchments have received between seven and nine non-native species: the Keisers, Lourens, Palmiet, Bot, Klein, Heuningness, Gamtoos and Bakens catchments (22.7% of the study area). The aforementioned 13 catchments represent more than 90% of the study area. No non-native fish have been recorded into only seven catchments, making up 0.8% of the study area. The number of non-native species per catchment is presented graphically in Figure 3.

River habitat invaded

The length of perennial and non-perennial components of the three rivers is presented in Table 3. The invasion status of 30.5% of the perennial river length in the three catchments was unknown (Figure 4). Of the length of river where the invasion status is known, an estimated 84.9% of the river length has been invaded by non-native fishes. At least 9.8% of the perennial river length of the three rivers has not been invaded. The Berg River has the highest level of invasion with an estimated 92.7% of its length invaded (Table 4). Conversely, the Olifants-Doring River has the lowest level of invasion with an estimated 80.4% of its river length invaded. When the length of river invaded is converted to the extent of river habitat invaded (using $n = 1.5$) the habitat invaded is estimated to be 92.6% and the unknown portion 24.2%. The habitat invaded in the Berg River is estimated to be 98.0%, 94.4% for the Breede River and 88.5% for the Olifants-Doring (Table 4).

Discussion

It is clear that the extent of freshwater fish invasion in the CFR is extensive. Only seven small coastal catchments have, to our knowledge, remained free of the introduction of non-native fishes. Considering the abundance of non-native fish populations in the region, it is clear that fish to stock these uninvaded catchments are free availability and it is likely that they will not remain free of non-native fishes for long.

The accuracy of the estimate is compromised by the large proportion of rivers placed in the "Status unknown" category. About one third of the perennial river length in the three catchments fell into this category. This highlights the poor knowledge of the distribution of the native and non-native fishes and the extent to which the river systems have been invaded by the non-native fishes. The results of this analysis suggest that the extent of invasion of the rivers in the Cape Floristic Region exceeds 85%, but further work is required to provide a better estimate. The inclusion of the Gouritz, Gamtoos and smaller system in the region may change this value, but initial calculations confirm that the level of invasion suggested by Hamman (2008) is a good estimate of the extent of freshwater fish invasion in the region, possibly even conservative. The exercise also highlights the lack of information available for the conservation and management of freshwater fishes in the Cape Floristic Region and the desperate need for researchers and conservation officials to start taking directed conservation action in order to conserve the remaining population of native freshwater fishes of this region.

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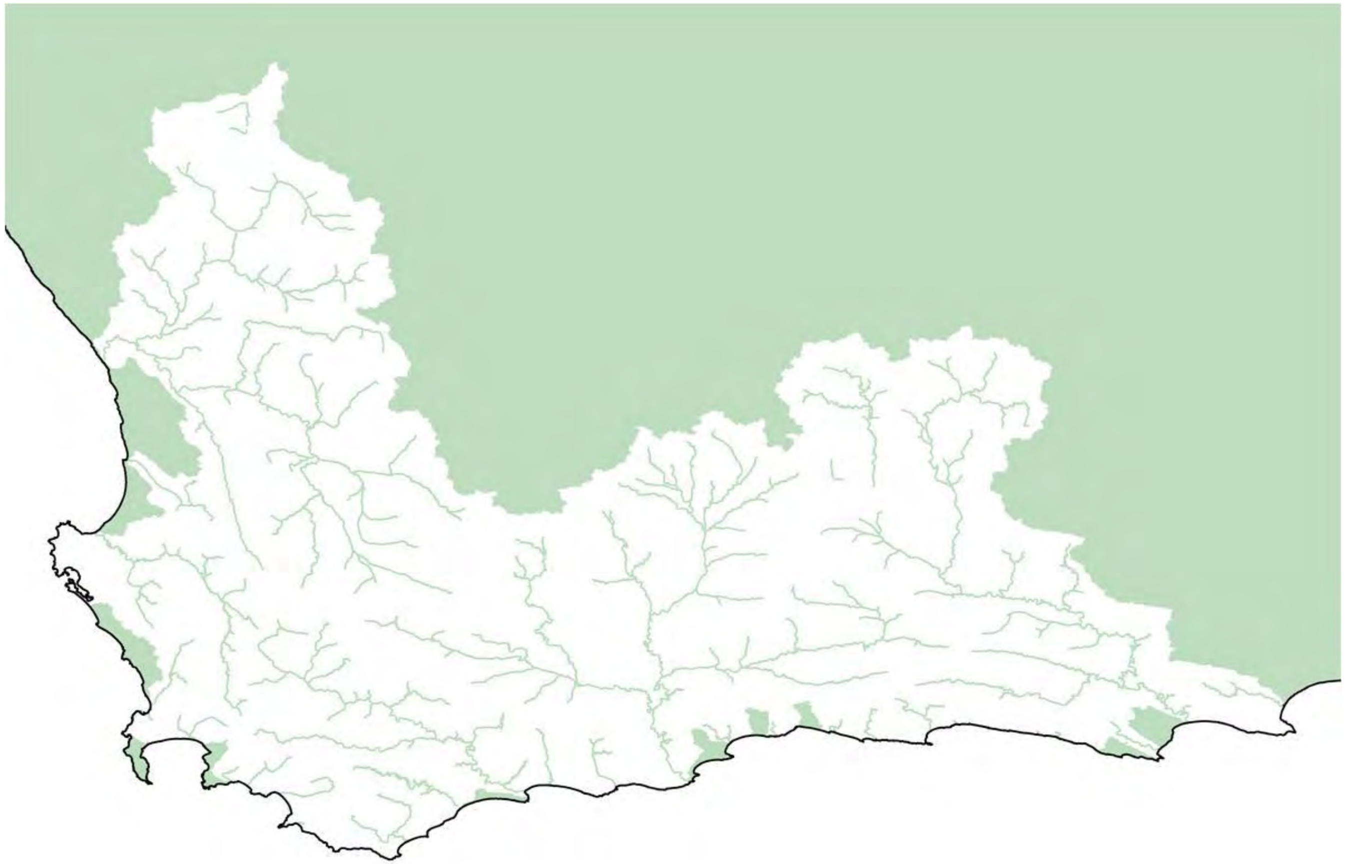

Figure 1: Map of the Cape Floristic Region showing the catchment area included in the study. Shaded areas were not included in the study.

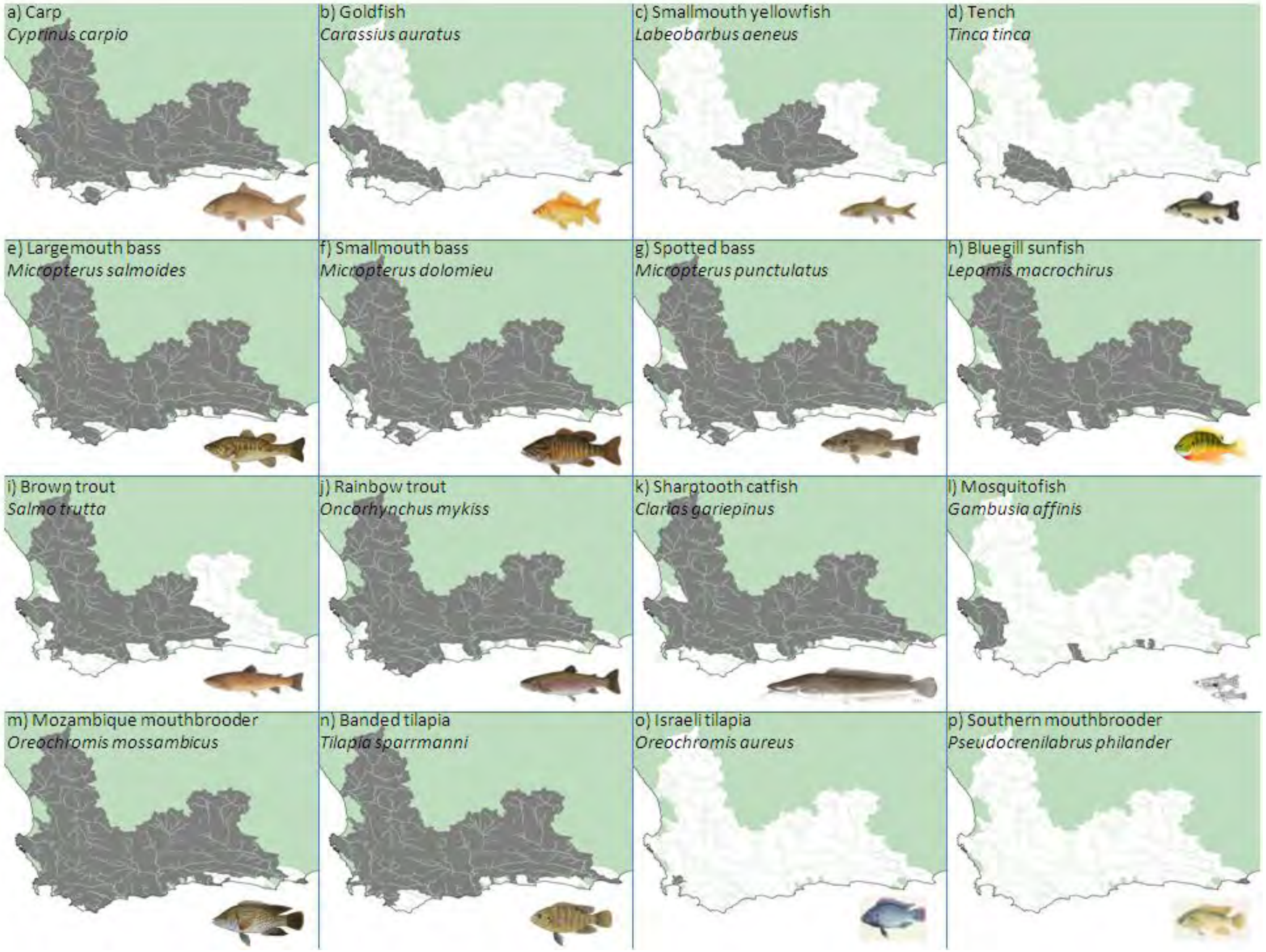

Figure 2: Maps reflecting the catchment area invaded by non-native fish in the Cape Floristic Region by species: a) carp; b) goldfish; c) smallmouth yellowfish; d) tench; e) largemouth bass; f) smallmouth bass; g) spotted bass; h) bluegill sunfish; i) brown trout; j) rainbow trout; k) sharptooth catfish; l) western mosquitofish; m) Mozambique mouthbrooder; n) banded tilapia; o) Israeli tilapia; and p) southern mouthbrooder.

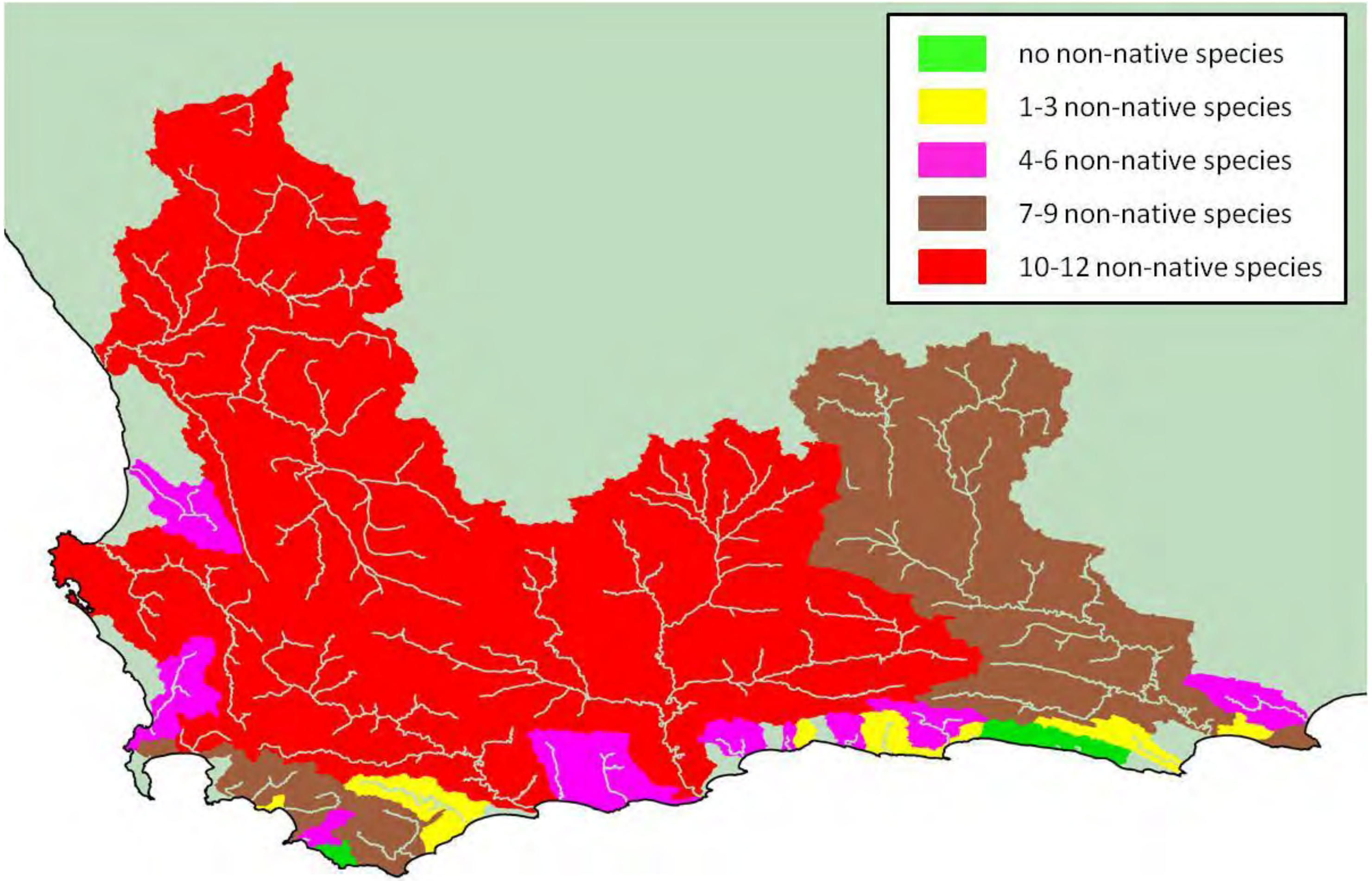

Figure 3: Map reflecting the number of non-native fishes per catchment of the Cape Floristic Region

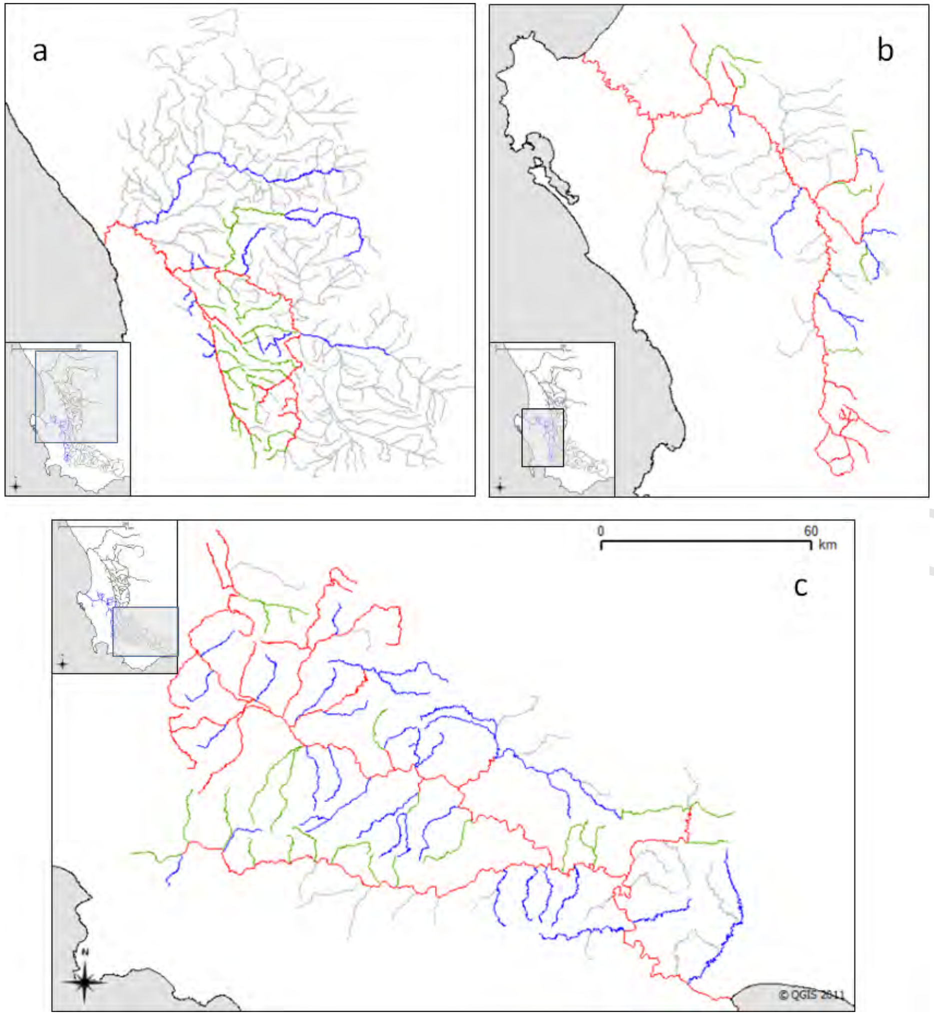

Figure 4: Map reflecting the extent of non-native fish invasion in three major rivers of the Cape Floristic Region: a) Olifants-Doring; b) Brerg and c) Breede. Red indicate invaded reaches, grey non-perennial reaches, blue reaches where invasion state is not known and green reaches that are partially invaded

Tables

Table 1: Extent of invasion by introduced freshwater fishes in three major catchments in the Cape Floristic Region

Common Name	Species	Authority	Number of catchments	% Study Area	% of CFR Area
Common carp	*Cyprinus carpio*	Linnaeus, 1758	17	92.3	78.9
Goldfish	*Carassius auratus*	(Linnaeus, 1758)	7	15.0	12.8
Smallmouth Yellowfish	*Labeobarbus aeneus*	(Burchell, 1822)	2	34.7	29.7
Tench	*Tinca tinca*	Linnaeus, 1758	1	9.0	7.7
Largemouth bass	*Micropterus salmoides*	(Lacepède, 1802)	26	97.1	83.1
Smallmouth bass	*Micropterus dolomieu*	(Lacepède, 1802)	17	93.0	79.6
Spotted bass	*Micropterus punctulatus*	(Rafinesque, 1819)	9	90.1	77.0
Bluegill	*Lepomis macrochirus*	Rafinesque, 1819	20	93.7	80.1
Rainbow trout	*Oncorhynchus mykiss*	(Walbaum, 1792)	16	93.2	79.7
Brown trout	*Salmo trutta*	Linnaeus, 1758	8	68.7	58.7
Sharptooth catfish	*Clarias gariepinis*	(Burchell, 1831)	13	89.9	76.9
Mosquitofish	*Gambusia affinis*	(Baird & Girard, 1853)	10	8.1	6.9
Mozambique mouthbrooder	*Oreochromis mossambicus*	(Peters, 1852)	30	96.9	82.9
Banded tilapia	*Tilapia sparrmanii*	A. Smith, 1840	22	94.4	80.7
Israeli tilapia	*Oreochromis aureus*	(Steindachner, 1864)	1	0.4	0.3
Southern mouthbrooder	*Pseudocrenilabrus philander*	(Weber, 1897)	1	0.2	0.2

Table 2: Extent of invasion by introduced freshwater fishes in three major catchments in the Cape Floristic Region (CFR)

Number of non-native species	Number of catchments	% Study Area	% of CFR
not included	16		14.5
0	7	0.8	0.7
1-3	9	2.6	2.2
4-6	13	6.3	5.4
7-9	8	22.7	19.4
10-12	5	67.7	57.9
Total CFR	58		

Table 3: Extent of invasion by introduced freshwater fishes in three major catchments in the Cape Floristic Region

	Olifants-Doring	Berg	Breede
Total length of river in catchment (km)	7453	1409	2470
Perennial length of river in catchment (km)	1992	894	2089
Perennial length (% of total length)	26.7	63.5	84.6
Invasion status unknown (% of perennial length)	32.8	13.7	35.6

Table 4: Estimated extent of river length and river habitat invaded by non-native freshwater fishes in three major catchments in the Cape Floristic Region. The length of stream is converted to habitat by multiplying the respective reach lengths by the Strahler stream order raised to the power of n. The result is presented as the estimated percentage of habitat invaded.

n	Three Rivers	Olifants	Berg	Breede
	%	%	%	%
0	84.9	80.4	92.7	84.9
1	90.4	85.8	96.7	91.8
1.5	92.6	88.5	98.0	94.4
2	94.3	90.7	98.8	96.4

Review of the proposal to eradicate non-native fish from priority rivers in the Cape Floristic Region, South Africa.

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ABSTRACT:

Non-native fish are considered the most important threat to the survival of a unique assemblage of indigenous freshwater fishes of the Cape Floristic Region (CFR) of South Africa. A pilot project to evaluate the use of the piscicide rotenone to eradicate non-native fish from selected river reaches has been proposed by CapeNature. Four rivers are included in the pilot project, each having unique characteristics and challenges to achieving non-native fish eradication and the restoration of the indigenous fish fauna. In this paper, we discuss the management methods available to reduce the impact of non-native fish on indigenous species and describe the proposal by CapeNature to eradicate non-native fish from four rivers in the CFR. We justify the need for the project and the site selection, and discuss the findings of the environmental impact assessment (EIA) undertaken for the project. The EIA concluded that the project was justified and necessary, the choice of rivers sound, and that, in most of the cases, the use of piscicides is recommended. The successful completion of the pilot project will help establish protocols for implementing a wider conservation plan for the indigenous freshwater fishes of the CFR.

KEY WORDS:

River rehabilitation; conservation; piscicides; non-native fish; eradication

Introduction

The Cape Floristic Region (CFR), a 88,000 km^2 Mediterranean-climate region on the south-western tip of Africa, is a recognised biodiversity hotspot with high levels of endemicity (Myers *et al.* 2000, Cowling and Pressey 2003, Darwall *et al.* 2009). Internationally acclaimed for its plant biodiversity, the CFR is also a centre of endemism for freshwater fauna, including aquatic invertebrates, amphibians and fishes (Skelton *et al.* 1995, Wishart and Day 2002, Younge and Fowkes 2003, Darwall *et al.* 2009). The proportion of endemic species for freshwater fauna is higher than that of plants, though species numbers are lower (Wishart and Day 2002). The freshwater fishes of the CFR exhibit typical characteristics of old, well established mountain faunas, including a high degree of endemicity (89%), geographically-restricted ranges, relative inflexibility in life history styles, and a low resilience to disturbance, especially introduced piscivorous fish (Skelton 1987). Most species are relatively small, < 150 mm SL when adult. None are entirely piscivorous, although the Cape kurper *Sandelia capensis* (Cuvier, 1831) and some of the larger cyprinids, e.g. Clanwilliam yellowfish, *Labeobarbus capensis* (A. Smith, 1841), are partly piscivorous as adults (Skelton 2002).

Twenty seven taxa of primary freshwater fish are currently recognised for the CFR (Impson 2007, Tweddle *et al.* 2009), but this number will increase following taxonomic revisions of the genera *Pseudobarbus* Smith, 1841, *Galaxias* Cuvier, 1816 and *Sandelia* Castelnau, 1861. Of the taxa currently recognised, six are listed as Critically Endangered (22%), ten as Endangered (37%), three as Vulnerable (11%), three as Near Threatened (11%), two as Least Concern (7%), and three as Data Deficient (11%) in the 2007 IUCN assessment (Impson 2007, Tweddle *et al.* 2009). Diadromous taxa are present in the CFR but are not addressed in this paper because all are currently listed as Least Concern (Tweddle *et al.* 2009). With the imminent recognition of additional endemic taxa, most with extremely limited distributions, the number of threatened taxa is considered conservative. The 2007 IUCN assessment lists the major threats to the CFR's 24 endemic primary freshwater fish taxa as: non-native fish (23 taxa), habitat destruction (18 taxa), pollution including pesticides (2 taxa), utilization (3 taxa) and genetic integrity (4 taxa) (Tweddle *et al.* 2009). Researchers and conservation officials in the CFR agree that non-native fish are the most significant threat to the long-term survival of their indigenous freshwater fish assemblages (Barnard 1943, Coke 1988, Skelton *et al.* 1995, Impson and Hamman 2000, Skelton 2000, Impson *et al.* 2002a, b, Skelton 2002, Cambray 2003a, b, Impson 2007, Tweddle *et al.* 2009).

This paper reviews the impact of non-native fish on the indigenous freshwater fishes of the CFR and the management methods available to reduce these impacts. A proposal by CapeNature, the conservation agency for the Western Cape, to eradicate non-native fish from four rivers in the CFR is described, justifying the need for the project, site selection and reporting the findings of the environmental impact assessment undertaken to review the proposed project.

Non-native fish introductions

The CFR has a long history of freshwater fish introductions (de Moor and Bruton 1988, Skelton 1990, Skelton *et al.* 1995, Skelton 2000, 2002), dating back to the late 1700s following introductions by Dutch and English settlers (de Moor and Bruton 1988, Picker and Griffiths 2011). To date, 24 species of freshwater fish have been introduced to the inland waters of the CFR for recreational angling, aquaculture, or biocontrol of aquatic weeds and mosquitoes (Table 1). Of these, seven have failed to establish self-sustaining populations and one, Eurasian perch *Perca fluviatilis* Linnaeus, 1758, is believed to have been extirpated. In addition, triploid grass carp *Ctenopharyngodon idella* (Valenciennes, 1844) have been introduced to farm dams for aquatic weed control. Although these fish are sterile (a requirement of permits), they are long lived, grow to be very large (over 1 m), and escapees into natural watercourses could have an impact on both aquatic vegetation and invertebrates (Pípalová 2006).

Three endemic taxa have been translocated between catchments within the CFR. Cape kurper was illegally introduced into a dam in the Twee River Catchment (Olifants-Doring System) in the 1950s (Hamman *et al.* 1984) and subsequently invaded this catchment following the rupture of the dam during a rainfall event (Impson *et al.* 2007, Marr *et al.* 2009). In the 1980s, the conservation authority in the Western Cape (the then Cape Department of Nature and Environmental Conservation) intentionally introduced the Clanwilliam yellowfish above waterfall barriers in the Twee (Impson *et al.* 2007), Ratels (Impson 2010), and Boontjies Rivers (Impson and Tharme 1998) of the Olifants-Doring catchment as a conservation measure. Breede River redfin *Pseudobarbus* sp. "burchelli Breede" were unwisely cultured in outdoor ponds at the Jonkershoek hatchery by the conservation authority from which they escaped and invaded the Eerste River near Stellenbosch in the 1980s.

In the 1980s, the conservation authority declared a moratorium on the stocking of fish into the rivers of the CFR (Coke 1988, Skelton 2000). Since 1988, no new species have been recorded but the secondary spread of species previously introduced continues unabated (Impson *et al.* 2002a, b, Skelton 2002, Impson 2006, 2007). Of particular concern is sharptooth catfish *Clarias gariepinus* (Burchell, 1822), which has been introduced into several river systems by anglers, farm labourers, rural land-owners, and inter-basin transfer schemes (Cambray 2003c, Impson 2006, 2007). It is possible that additional taxa have been introduced into the region, but these introductions have not yet been confirmed by researchers or conservation officials.

It has been estimated that more than 90% of mainstem river habitat in the CFR has been invaded by non-native fishes (Marr in press). Only seven small coastal catchments, together representing 0.8% of the CFR by area, have no recorded non-native fish introductions (Marr in press). The majority of catchments contain four or more non-native species, with the major river systems of the Olifants-Doring, Berg, Breede and Gouritz containing 10 or more non-native species (Figure 1). Indigenous primary freshwater fish are absent, or rare, in the majority of the reaches invaded by non-native species, and are largely restricted to reaches above barriers that have prevented invasion by non-native species (Skelton 1990, Skelton *et al.* 1995, Skelton 2000, Impson *et al.* 2002b, Skelton 2002, Impson 2007, Tweddle *et al.* 2009).

Non-native fish impacts on indigenous fishes in the CFR

Freshwater communities are particularly vulnerable to non-native species introductions (Saunders *et al.* 2002, Skelton 2002, Cambray 2003b). The introduction of non-native fish results in impacts at genetic (gene transcription, hybridisation); individual (behaviour, morphology, vital rates); population (transmission of parasites/diseases, demographic effects, distributional effects); community (species extirpations, compositional changes, alterations in food webs); and ecosystem (biochemical cycles, energy fluxes between ecosystems, ecological engineering) levels (Cucherousset and Olden 2011).

The negative ecological impact of non-native fish on indigenous fish was first recognised in the late 1930s, especially in the biodiversity hotspot of the Cederberg, Western Cape (Barnard 1943). Predation by non-native fish on indigenous fish assemblages was raised as a significant concern in the CFR when the predatory effects of rainbow trout *Oncorhynchus mykiss* (Walbaum, 1792) and smallmouth bass *Micropterus dolomieu* Lacepède, 1802 on *Pseudobarbus* redfin were noticed (Hey 1926, Barnard 1943, Harrison 1950, 1951a, b, 1952a, b, Skelton 1983, Coke 1988, Skelton 1990, Skelton *et al.* 1995, Skelton 2000, 2002). Despite this, non-native fish continued to stocked throughout the CFR until the 1980s (Coke 1988, Impson and Hamman 2000, Skelton 2000, Impson *et al.* 2002a, Richardson *et al.* 2004, Tweddle *et al.* 2009). However, the initial impact of non-native fishes on indigenous fish assemblages in the CFR was poorly recorded and not studied in detail, because most impacts occurred before any great interest in conserving the indigenous taxa emerged.

There is a strong correlation between the presence of predatory non-native fish and the absence of indigenous taxa, a situation analogous with that in the Colorado River in the USA (Marsh and Pacey 2005). This immiscibility between indigenous and non-native fishes is particularly evident where centrarchids (basses and sunfish) and salmonids are present. Non-native fish affect the behaviour and composition of the indigenous fish assemblages in the CFR (Cambray 2003a, b, Woodford and Impson 2004, Woodford *et al.* 2005, Shelton *et al.* 2008) as well as lower trophic levels of the food web, including aquatic invertebrates and algae (Lowe *et al.* 2008). Because of this, the remaining indigenous fish populations can only be conserved, and their ranges increased, through the elimination of non-native species from rivers identified as conservation priorities (Impson 2007). At national level, critical biodiversity areas for indigenous freshwater fauna have been mapped through the National Freshwater Ecosystem Priority Areas (NFEPA) project (Hill 2009). These identify priority areas for freshwater biodiversity conservation that require, among other intervention measures, non-native fish control.

Management of non-native fish populations

Although the presence of predatory non-native species in the CFR poses the greatest threat to indigenous fish assemblages (Impson and Hamman 2000, Impson *et al.* 2002b, Skelton 2002, Impson 2007, Tweddle *et al.* 2009), it is acknowledged that non-native freshwater fishes are the mainstay of South Africa's recreational angling and freshwater aquaculture industries (van Rensburg *et al.* 2011). A balance must therefore be maintained between the conservation of indigenous taxa and the management of non-native fish (Chadderton 2003, van Rensburg *et al.* 2011). This balance is an important aim of regulations being developed as part of the National Environmental Management: Biodiversity Act (NEM:BA) of 2004. NEM:BA designates non-native species into three categories: **prohibited species** which may not be imported into the country, **exempted species** which may be freely traded within the country without permit, and **invasive non-native species** which must be controlled by means of a management plan. The latter category includes the species most commonly targeted by anglers, such as brown trout *Salmo trutta* Linnaeus, 1758, rainbow trout, largemouth bass *Micropterus salmoides* (Lacepède, 1802), smallmouth bass, and carp *Cyprinus carpio* Linnaeus, 1758. For all other non-native species, a risk assessment is required prior to the issue of permits (van Rensburg *et al.* 2011). It is envisaged that an area approach (i.e. zoning) will be part of the management plans developed for economically and recreationally important non-native species. The zoning of areas for use of various non-native species allows for trade-offs between conservation priorities (e.g. NFEPA) and recreational-economic interests (van Rensburg *et al.* 2011), and has been completed in an open participatory basis involving experts on aquatic conservation issues and representatives from key angling and aquaculture groups.

Prevention of future introductions is the most effective way of addressing invasion by non-native species (Saunders *et al.* 2002, Britton *et al.* 2011) and can be enhanced by instituting adequate decision support tools and risk assessment metrics (Britton *et al.* 2011). Capacity at relevant levels (enforcement, taxonomic specialists, and communication) is critical for prevention programmes to be effective. Where non-native species have already become established, active management needs to focus on reducing their impacts and preventing further spread (Saunders *et al.* 2002, Britton *et al.* 2011). Eradication is more cost-effective than long-term mechanical control (Bomford and O'Brien 1995, Bomford and Tilzey 1997, IUCN 2000), particularly for recent introductions, or where the species has been spatially constrained (Britton *et al.* 2011). Eradication of non-native fishes is achieved by chemical treatments, e.g. piscicides, or by draining water-bodies (Finlayson *et al.* 2000, Collares-Pereira and Cowx 2004, Britton *et al.* 2011). Sustained mechanical removal has only been successful at small scales (Britton *et al.* 2011). Eradication should, however, only be attempted where it is ecologically feasible and has financial and political support (Huntley 1999, IUCN 2000). Where eradication is not feasible, control is the next best alternative. Control programmes using mechanical removal techniques (e.g. electrofishing or netting) are generally effective in suppressing population abundance and reducing their recruitment (Britton *et al.* 2011). Eradication and control of non-native fishes remain constrained by their lack of selectivity, and the challenges of treating large spatial scales effectively (Britton *et al.* 2011). Non-native species invasions are, however, generally irreversible (Cucherousset and Olden 2011) and the current technologies available for the eradication of established non-native species are likely to result in the loss of indigenous taxa from treated reaches during the eradication (Myers *et al.* 2000), most of which will re-colonise the treated reaches from adjacent areas (Vinson *et al.* 2010).

While large-scale eradication of non-native fish is difficult, and rarely implemented, small-scale projects to eradicate non-native fish from priority reaches of small rivers to re-establish threatened fish species to parts of their original distribution range, and/or increase their distribution range, can be successfully completed with the current available technologies. For example, a piscicide based conservation management strategy, such as the one proposed for control of non-native fish in the lower Colorado River (Clarkson *et al.* 2005), could be implemented as follows:

a) Conservation authorities and other stakeholders identify rivers to be devoted exclusively to the conservation of indigenous fishes, and those for non-native fish utilisation, e.g. aquaculture and angling.
b) In rivers identified as conservation priorities where non-native species are present, beginning from headwaters, temporary barriers are installed and any indigenous fish above the barrier are caught and retained in off-stream holding facilities.
c) Non-native species above the barrier are chemically removed, after which the indigenous fish are returned, or introduced from an appropriate stock.
d) The process is then successively repeated downstream, removing the temporary upstream barrier after each step and eradication of the non-native species confirmed.

The risk of illegal release of non-native fish removed back into the treatment area must be minimized and new introductions of other non-native fish species prevented. Education and publicity initiatives with local land-owners and angling bodies should be established, in conjunction with simple and inexpensive monitoring protocols to detect non-native species in restored reaches, to ensure long-term success of conservation programmes (Clarkson *et al.* 2005). Establishing privately owned protected areas as part of South Africa's Protected Areas Act, Act 57 of 2003, through stewardship initiatives is necessary to ensure the active conservation of the indigenous freshwater fish of the CFR.

The NFEPA project and the zoning of catchments satisfied the first step in the above management strategy. The next step was to begin active non-native fish eradication projects in priority conservation areas. In order to achieve this, the CAPE Alien Fish Eradication Project has been proposed, a pilot project to evaluate the use of piscicides as a tool to eradicate non-native fish from selected rivers in the CFR. This has been done in an attempt to comply with section 28 of South Africa's National Environmental Management Act (NEMA) which obliges conservation authorities to remedy degradation of environments harmed by non-native and/or invasive species.

The CAPE Alien Fish Eradication Project

The Cape Action Plan for People and the Environment (CAPE) Project, a comprehensive conservation plan for the CFR (Younge and Fowkes 2003), recognised the need for intervention to conserve the freshwater fishes of the CFR and established the CAPE Alien Fish Eradication Project to identify priority rivers for non-native fish eradication (Impson 2007). Funded by the World Bank and managed by CapeNature (formerly the Cape Department of Nature and Environmental Conservation), this project was separated into two phases (Impson 2007):

- Phase 1 – identification of priority rivers for the eradication of non-native fish and completion of an Environmental Impact Assessment (EIA) for the most suitable rivers, and
- Phase 2 – execution of the intervention if deemed to be effective and appropriate.

The pilot project aims to evaluate the eradication of non-native fish from four streams in the CFR using the piscicide rotenone and to monitor the subsequent recovery of the treated reaches, specifically the recovery threatened indigenous fishes. If successful, the project will provide protocols for the implementation of active indigenous fish conservation projects in the CFR, and elsewhere in South Africa.

Selection of Rivers

Criteria for the selection of rivers appropriate for the pilot project were determined at a specialist workshop held at the South African Institute for Aquatic Biodiversity, Grahamstown (Impson, 2007). These criteria included biological, land-use, social, financial and logistical considerations (see Impson, 2007 for discussion). Field surveys of all potential rivers identified by the workshop were conducted in March and April 2004 to evaluate the selected rivers against these criteria. A subsequent workshop selected six rivers for the pilot project: the Rondegat, Krom and Twee Rivers in the Cederberg; the Dorps and Paradys Rivers in the Gouritz catchment; and the Krom River in the Eastern Cape (Figure 1). Further field surveys, using a combination of fyke netting, seine netting, electric fishing and snorkel surveys, delineated the distribution ranges of indigenous and non-native fish and identified potential barriers that could be used as the upper or lower barriers for the eradication of non-native fish using piscicides. The capture data from these field trips are summarised in the supplemental material.

At the third specialist meeting, the list was reduced to four rivers believed to have the highest likelihood of success, the Krom, Rondegat and Twee rivers in the Cederberg, and the Krom River (Eastern Cape). Each river had different characteristics and conservation targets. The Dorps and Paradys Rivers in the Gouritz catchment were considered unsuitable for the pilot project due to the lack of suitable barriers limiting the upper range of the non-native fish invasion, accessibility to the river to implement the eradication, and potential social issues regarding the use of a piscicide in the water supply of a town in a water-limited area.

Rondegat River, Cederberg

The Rondegat River, a tributary of the Olifants River, drains a moderately transformed catchment, containing pristine natural areas, cattle pastures, and fruit orchards, before discharging into Clanwilliam Dam. Above a small waterfall barrier 5 km above Clanwilliam Dam (Figure 2), the river holds healthy populations of five indigenous fish taxa Clanwilliam redfin *Barbus calidus* Barnard, 1938, fiery redfin *Pseudobarbus phlegethon* (Barnard, 1938), Clanwilliam rock catlet *Austroglanis gilli* (Barnard, 1934), Cape galaxias *Galaxias zebratus* Castelnau, 1861 (possibly an undescribed taxon), and Clanwilliam yellowfish (Woodford 2005, Woodford *et al.* 2005). Below the barrier, only large Clanwilliam yellowfish and invasive smallmouth bass are found. A water abstraction weir, situated 4 km downstream of the barrier, marks the lower boundary of the proposed intervention area. Below this weir, the river is dry for most of the summer months, but requires reinforcement to prevent re-invasion by bass or other species. The river flows in a single channel through the proposed intervention area, although several furrows irrigate pastures alongside the river. If successful, the project will extend the distribution of the five indigenous taxa in the Rondegat River by more than 20%, providing more varied habitat with larger, deeper pools, and establishing a buffer zone between the non-native species and the indigenous taxa. It could also allow the reintroduction of Clanwilliam sawfin *Barbus serra* Peters, 1864 and Clanwilliam sandfish *Labeo seeberi* (Gilchrist & Thompson, 1911), reported Van Rensburg (1966) from the lower reaches of the Rondegat but extirpated following bass invasion, an exciting development if these two endangered taxa re-establish.

Krom River, Cederberg

The Krom River is a tributary of the Matjies River in the Doring Catchment. A large waterfall above Disa Pool (Figure 3) marks the upper limit of fish distribution. Below Disa Pool, the river flows through a gorge of bedrock steps, pools, and chutes. The valley opens up below the upper gorge and the low gradient river with sandy runs and pools flows through a near-pristine valley before entering the highly transformed campsite at Krom River farm. Below the farm, the river flows through the near-pristine Matjies River Nature Reserve to the confluence with the Matjies River. One indigenous species, Clanwilliam rock catlet occur from the lower reaches of the upper gorge to just above the campsite (Figure 3). Redfin, possibly Clanwilliam redfin and/or Doring fiery redfin *Pseudobarbus* sp. "phlegethon Doring", were reportedly common in pools near the campsite before the introduction of trout, but appear to have been extirpated. Three non-native species have been introduced: rainbow trout in 1957 (Weaver 2008), bluegill sunfish *Lepomis macrochirus* Rafinesque, 1819 and largemouth bass in the 1980s. Rainbow trout occur from Disa Pool to just above the Matjies River confluence, while bluegill sunfish and largemouth bass occur in farm dams and in the river from the campsite downstream. A derelict weir just above the confluence with the Matjies River is the potential lower boundary for the non-native fish eradication, but requires refurbishment.

Non-native fish eradication from the Krom River provides an opportunity to establish a sanctuary for the Critically Endangered Doring fiery redfin, an undescribed taxon restricted to two unsustainable populations in the Doring system, both severely threatened by bass. With the upper reaches in a private conservancy, farming activities at Krom River farm being scaled back in favour of ecotourism, and the lower reaches in the Matjies Nature Reserve, the Krom River has the potential to be a "flagship" project for fish conservation and environmental awareness in the CFR, and South Africa as a whole.

Suurvlei River, Cederberg

The Suurvlei River, a tributary of the Twee River in the Doring River catchment, drains an area of intense deciduous fruit farming. Two fish, Twee River redfin, *Barbus erubescens* Skelton, 1974 and an undescribed taxon of *Galaxias* are the only indigenous fish of the Twee River, both endemic to this small catchment. Of these, only the redfin has been recorded from the Suurvlei tributary in recent surveys (Impson *et al.* 2007, Marr *et al.* 2009). Three non-native fish are present in the upper Twee catchment, translocated CFR endemics Cape kurper and Clanwilliam yellowfish, and North American bluegill sunfish (Impson *et al.* 2007, Marr *et al.* 2009). Only Cape kurper has been recorded in the proposed intervention area (Figure 4). The upper range of Cape kurper is a culvert on the Buffelshoek tributary and a bedrock step on the Suurvlei River. A small population of redfin persists above the Cape kurper distribution in the Suurvlei River (Figure 4). The lower limit of the proposed intervention is a bedrock step at the Suikerbossie Bridge (Figure 4), however, this barrier is considered insufficient to prevent re-invasion by Cape kurper and the construction of a weir is recommended for the site. Eradication of the Cape kurper from the Suurvlei will extend available habitat for the Twee River redfin, and allow the reintroduction of *Galaxias*, but should be regarded as only the first step in a greater conservation programme for the Twee River fish.

Krom River, Joubertina (Eastern Cape)

The Krom River originates in the Formosa State Forest and drains eastwards along the Lang Kloof on the western edge of the Eastern Cape. Two undescribed taxa occur in the headwaters of the Krom River, Krom River redfin *Pseudobarbus* sp. "afer Krom" and a *Galaxias* taxon. *Galaxias* were recorded above a large waterfall while redfin occurred below the waterfall downstream to a set of weirs built by the Working for Wetlands programme (Figure 5). Redfin were also found in a tributary downstream of the weirs (Figure 5), currently protected from bass invasion by the water abstraction off-take. Largemouth bass are present in farm dams above the redfin distribution and could invade the upper reaches through irrigation furrows. Large floods may flush the bass from the flashy upper reaches into the slower flowing middle reaches where they are common. The eradication project proposes to remove bass from between the waterfall and the Working for Wetlands weirs and from the farm dams upstream to prevent re-invasion. This would establish a sanctuary for the undescribed Krom River taxa.

Environmental Impact Assessment

The eradication of non-native species is not a listed activity in terms of South Africa's environmental legislation. However, under the National Environmental Management Act 107 of 1998, Section 28, a duty of care is required when engaging in activities that may have a detrimental effect on the environment. In addition, the National Water Act requires that the introduction of a chemical into a water body be formally assessed. Further, the sponsor of the project, The World Bank, insisted that an environmental impact assessment be completed to determine whether the proposed use of piscicides is viable and environmentally safe in the CFR. In 2008, CapeNature appointed Enviro-Fish Africa (Pty) Ltd to carry out the EIA on its behalf. The four rivers identified for the pilot project were selected on the basis that (a) indigenous taxa were critically endangered and non-native fish were a major threat to their survival, (b) the eradication of non-native fish was feasible, (c) the rivers would provide healthy habitat for the indigenous fishes when the non-native species were removed, and (d) the rivers were not of major importance to anglers.

The EIA concluded that the treatment will have some negative initial impacts on the aquatic invertebrate fauna but that the majority of organisms could be expected to survive the treatments (Enviro-Fish Africa 2009). In addition, rapid recovery of the stream faunas was predicted following colonisation from reaches up- and down-stream of the treatment reaches. Further, the project is unlikely to have any significant impacts, either positive or negative, on the regional conservation status of non-fish vertebrate fauna, because mammals, reptiles amphibians and birds would not be affected at the concentrations of rotenone required to kill fish (Enviro-Fish Africa 2009). The project was endorsed as vital for the survival of endangered fishes (Enviro-Fish Africa 2009). Systematic monitoring was recommended for each of the four rivers before, during and after the treatments. The legal assessment of the project concluded that Section 28 of NEMA places an obligation on CapeNature to remedy degradation of environments harmed by non-native and/or invasive species and that the proposed project is in accordance with international best practices for managing invasive non-native species (Enviro-Fish Africa 2009).

The EIA concluded that the justification for the project and the choice of rivers was sound. In most of the rivers proposed for the project, the use of piscicide, specifically those containing rotenone, was recommended. The Rondegat River was recommended as the first river treated. In the upper part of the Krom River (Cederberg), trial of physical eradication methods were recommended to minimise impacts on the indigenous Clanwilliam rock catlets and macro-invertebrates. Should the physical methods prove ineffective, rotenone treatment should proceed with rescue populations of aquatic fauna being kept in holding facilities for the duration of the treatment.

During the EIA, it became apparent that sectors of the flyfishing community (notably trout enthusiasts) took exception to a river containing rainbow trout (Krom River, Cederberg) being chosen for the project. Concern was also expressed regarding the use of piscicides containing rotenone and their impacts on non-target fauna, especially aquatic invertebrates. The media publicized the project and were initially very critical of it, with negative articles appearing in newspapers, e.g Cape Argus Bamford (2008), magazines, e.g. Flyfishing Thorpe (2008) and Farmers Weekly Steyn (2010), and several fishing web-sites. In retrospect, CapeNature was at fault by not embarking on a pro-active campaign to diffuse the potential media storm. The EIA, however, played a critical role in changing attitudes towards the project. This was achieved in three ways: 1) the project was being assessed by highly competent independent specialists, 2) the EIA held two rounds of public participation allowing stakeholders adequate opportunity to express their concerns, and 3) the outcome of the EIA recommended that the project proceed.

Current status of the project

CapeNature intends to treat the lower Rondegat River with the approved piscicide CFT Legumine in 2012. Two treatments will be undertaken in late summer, with the possibility of a third treatment a year later if some smallmouth bass survive the initial treatments. The use of three treatments is standard practice in the USA (BJ Finlayson, California Department of Fish and Game, 2010, pers. comm.). CapeNature has prepared documents to guide the final planning of the project, including Public Involvement, Communication, Fish Rescue, Monitoring, and Treatment Plans, in accordance with guidelines adopted by the American Fisheries Society (Finlayson *et al.* 2000). Comprehensive independent pre- and post-monitoring is being undertaken.

Concluding remarks

South African fish conservation experts agree that non-native fish pose the greatest threat to the continued survival of their unique and highly threatened freshwater fishes. Non-native fishes not only cause localized extirpations of indigenous fishes, their presence have major effects on trophic food webs and other aspects of ecosystem functioning. Through the recently completed NFEPA conservation planning process, priority fish conservation and river rehabilitation areas requiring non-native fish eradication have been identified. The four rivers chosen for the CAPE Alien Fish Eradication Project are amongst these. The use of approved piscicide containing rotenone is the preferred method for the eradication of non-native fish and has a proven track record of success in the USA, where its use is carefully managed (Finlayson *et al.* 2010). CapeNature's project has undergone a comprehensive EIA and is now in the implementation phase. It is expected that the project will achieve its objectives and provide a proven methodology to help conserve the unique aquatic biota of South Africa through the eradication of non-native fish from critical biodiversity areas identified through the NFEPA project, ultimately resulting in the down-listing of many threatened fish taxa.

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Figures

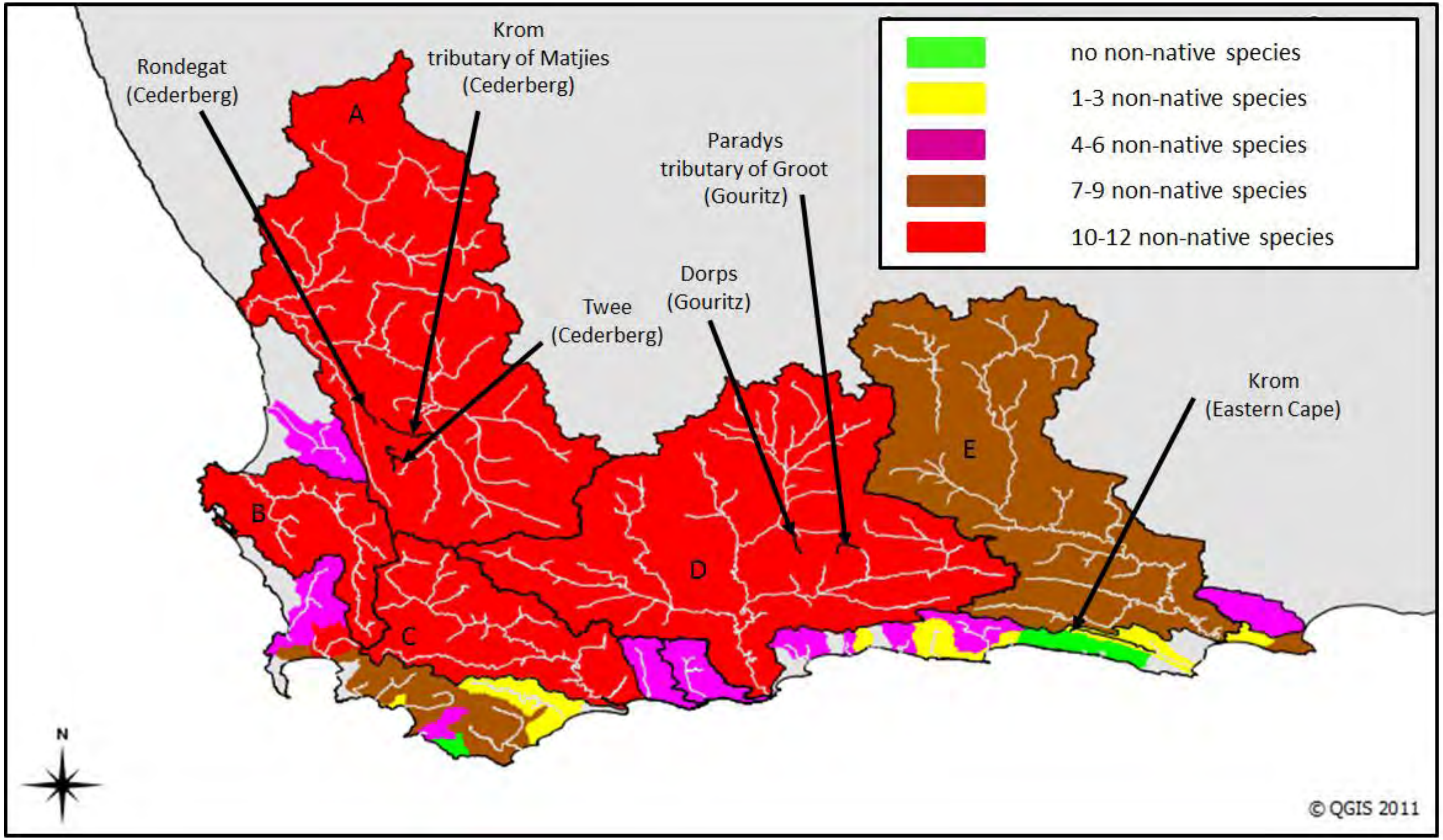

Figure 1: Map reflecting the number of non-native fish by catchment in the Cape Floristic Region

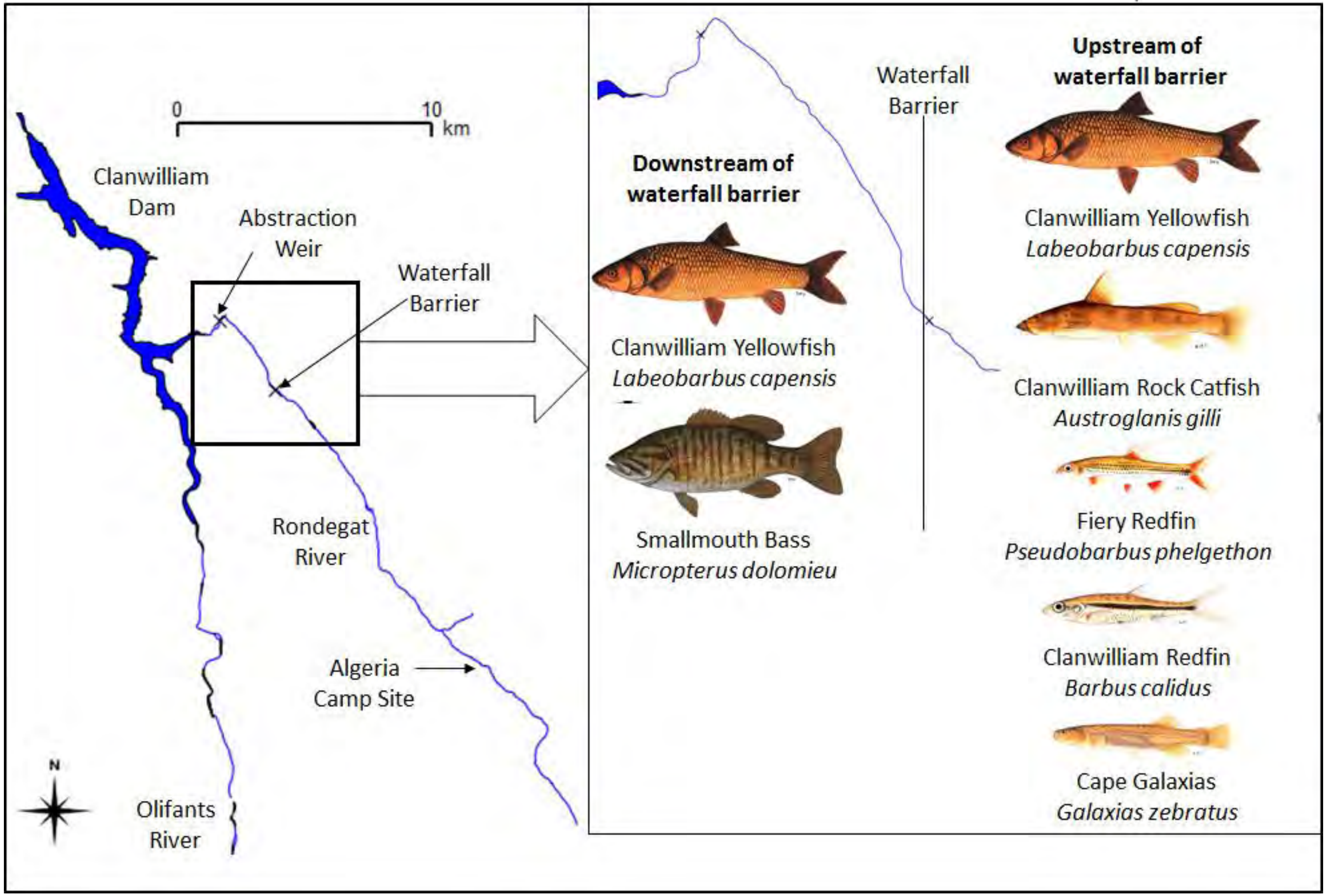

Figure 2 Rondegat River showing land marks, barriers and fish distributions.

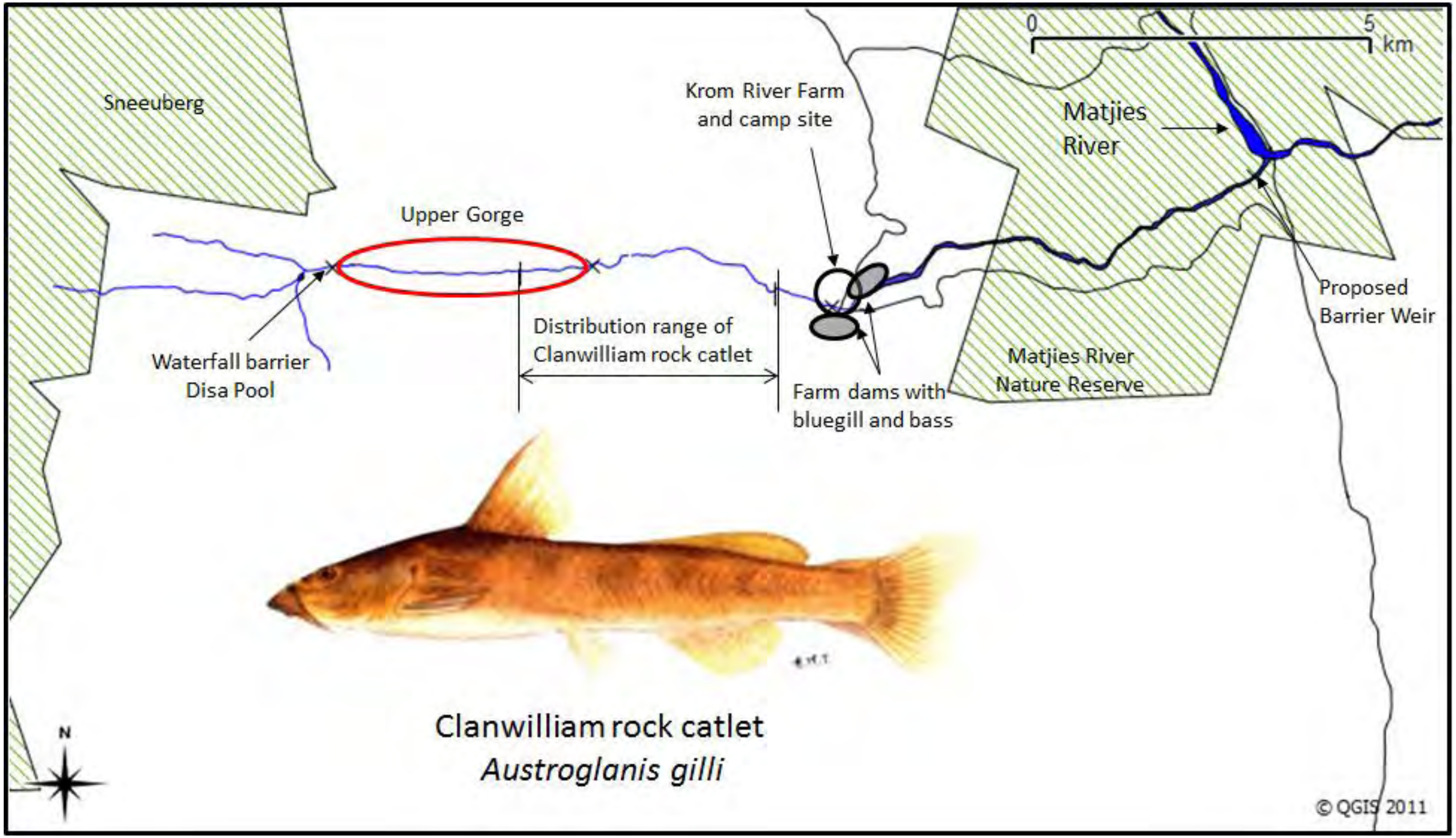

Figure 3 Krom River (Cederberg) showing land marks, barriers and fish distributions.

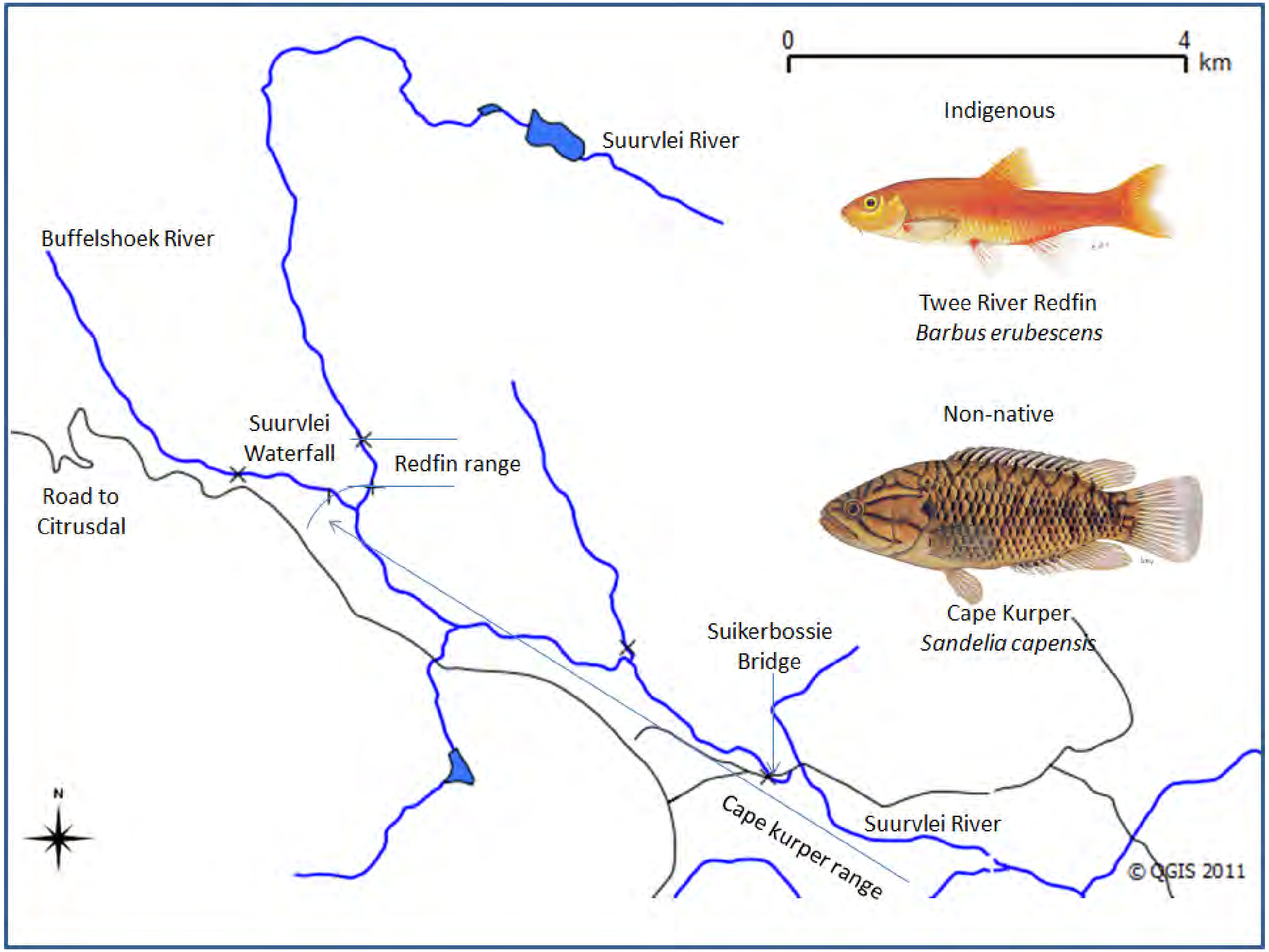

Figure 4 Suurvlei River showing land marks, barriers and fish distributions.

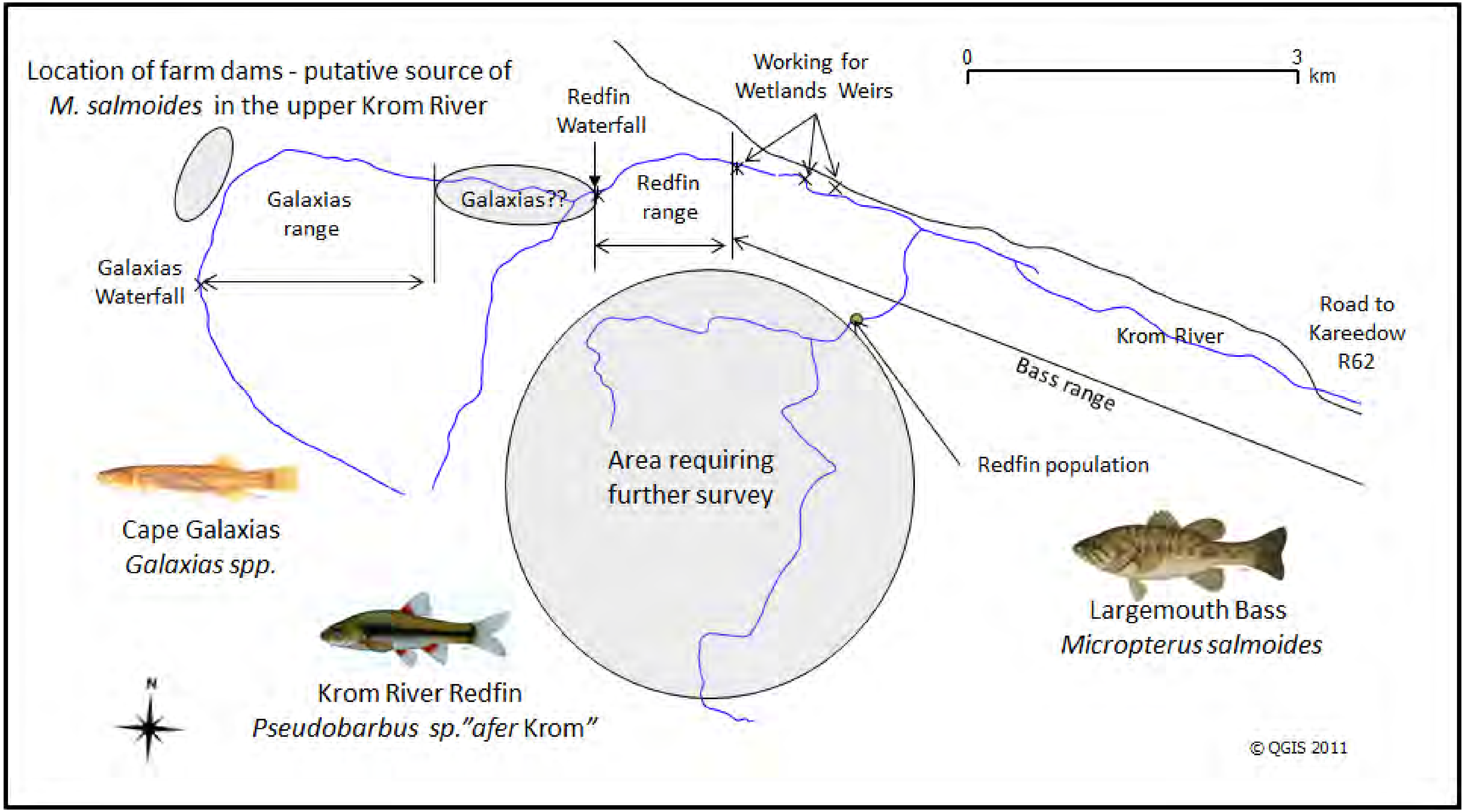

Figure 1 Krom River (Eastern Cape) showing land marks, barriers and fish distributions.

Tables

Table 1: Summary of freshwater fish introductions to the Cape Floristic Region (de Moor and Bruton 1988, Skelton 2001)

Year	Aquaculture	Reason for introduction	Outcome of introduction
1700s	Carp (*Cyprinus carpio*)	Ornamental, aquaculture, angling	Successful
	Goldfish (*Carassius auratus*)	Ornamental	Successful
1890s	Brown trout (*Salmo trutta*)	Angling	Successful
	Rainbow trout (*Oncorhynchus mykiss*)	Angling, aquaculture	Successful
	Atlantic salmon (*Salmo salar*)	Angling	Failed
1910	Tench (*Tinca tinca*)	Angling	Successful
1912	Guppy (*Poecilia reticulata*)	Biological control (mosquitoes)	Failed
1915	Israeli tilapia (*Oreochromis aureus*)	Aquaculture	Successful
	Eurasian Perch (*Perca fluvialitis*)	Angling	Successful +++
1928	Largemouth bass (*Micropterus salmoides*)	Angling	Successful
1936	Mozambique mouthbrooder (*Oreochromis mossambicus*)	Aquaculture	Successful
	Mosquitofish (*Gambusia affinis*)	Biological control (mosquitoes)	Successful
1937	Smallmouth bass (*Micropterus dolomieu*)	Angling	Successful
1938	Bluegill (*Lepomis macrochirus*)	Fodder for angling species	Successful
1939	Spotted bass (*Micropterus punctulatis*)	Angling	Successful
1941	Banded tilapia (*Tilapia sparrmanii*)	Fodder for angling species	Successful
1950	Brook char (*Salvelinus fontinalis*)	Angling	Failed
1953	Smallmouth yellowfish (*Labeobarbus aeneus*)	Angling	Successful
1959	Nile tilapia (*Oreochromis niloticus*) ++	Aquaculture	Failed
	Mango tilapia (*Sarotherodon galilaeus*)	Aquaculture	Failed
	Red-bellied tilapia (*Tilapia zilli*)	Biological control (aquatic plants)	Failed
1980s	Sharptooth catfish (*Clarias gariepinus*)	Aquaculture, angling	Successful
	Grass carp (*Ctenopharyngodon idella*) ++++	Biological control (aquatic plants)	Sterile triploids
	Southern Mouthbrooder (*Pseudocrenilabrus philander*)	Fodder for angling species	Successful

Note -

++ Nile tilapia was recorded as being established on the Cape Flats near Cape Town, but no populations have been recently recorded for this species and it is believed that this species did not established in the Cape Floristic Region, even though climate models predict that this species should be able establish in the region (Zambrano *et al.* 2006). The aquaculture industry is currently pushing for permission for hybrid *O. mossambicus* x *O. niliticus* to be introduced into the Cape Floristic Region.

+++ Eurasian perch successfully established in two systems in the Cape Floristic Region but has since been extirpated from both systems. One system was treated with rotenone and drained.

++++ Grass carp has been introduced into water bodies in the Cape Floristic Region to control aquatic vegetation. Only sterile triploid fish from certified hatcheries are permitted to be introduced. The species has established in the Orange-Vaal and Pongola systems even though they were expected not to establish in these systems. The species is long lived and could survive in natural water courses if they escape from the farm dams.

A.4 Freshwater Fish Introductions in Mediterranean-Climate Regions

Appendix A.4.1: Catchments included in the study from each ecoregion.

Appendix A.4.2: Freshwater fishes introduced into the Mediterranean-climate regions included in this study.

Appendix A.4.3: Catchmewnt and regional level distribution data for the freshwater fishes of Mediterranean-climate regions (included on the CD included – File: Med climate.xlsx, tab Catch Historical for the historical data and tab Catch Current for the current distribution data. The regional data is included in the same workbook: tab Reg Historical for the historical data and tab Reg Present for the current distribution data)

Appendix A.4.4: Functional traits of the freshwater fishes of Mediterranean-climate regions included in this study. The categories are expolained in Table 4.2 in the body of the thesis. (included on the CD included – File: Med climate.xlsx, tab functional traits)

Appendix A.4.5: The number of freshwater fish species in oder and family containing freshwater species.

Appendix A.4.1: Catchments included in the study from each ecoregion.

Ecoregion	Catchments
Western Iberia	Tambre, Northern coastal, Uila, Minho, Lima, Cávado, Ave, Douro, Vouga, Mondego, Extremadura coastal, Tagus, Sorraia, St. Estevão, Alentejo coastal, Sado, Mira, Navea, Galicia and Asturias, Algarve coastal
Southern Iberia	Guadiana, Odiel, Tinto, Guadiamar, Guadalquivir, Guadalete, Southern València, Segura
Eastern Iberia	Júcar, Túria, Mijares, Ebro, Cantabria and Basque, Nalón, Catalunya
Cantabrian Coast Languedoc	Massane, Tech, Tet, Agly, Berre, Aude, Orb, Hérault, Lez, Salaison Vidourle Rhône Touloubre Arc Huveaune Gapeau Argens Siagne Loup Var
Dalmatian Coast	Soča, Mirna, Raša, Lika (Jadova), Zrmanja, Krka, Cetina, Neretva
South East Adriatic	Skadar Lake system, Drin, Ohrid Lake system, Mati, Erzeni, Skumbini, Semeni, Aoos/Vjose, Prespa
Ionian Drainages	Kalamas, Zaravina, Pamvotis, Paramythia, Kalodiki, Acheron, Ziros, Louros, Arachthos, Vouvos, Vlychos, Voulkaria, Astakos, Acheloos, Evinos, Glafko,s Piros, Tsivlos, Prokopos, Kotychi, Pinios Pel, Alfios, Neda, Yiannousagas, Peristeras, SW Messinia, Pamissos, Kandila, Feneos, Stymphalia, Taka, Evrotas, Vassilopotamos, Smynous, Adreli, Lerni, Kato Almyri
Aegean Drainages	Mornos, Assopos Pel, Krios, Krathis, Vouraikos, Keronitis, Selinous, Meganitis, Phoenix, Volinaios, Erassinos Arg, Vouliagmeni, Erassinos Vra, Rafina, Kato Souli, Marathon, Kifissos Att, Assopos Beo, Kifissos Beo, Yliki, Sperchios, Cholorema, Kireas, Manikiotiko, Rigia
Vardar	Axios, Loudias, Vegoritis, Kastoria, Aliakmon, Mavroneri, Pinios The
Thrace	Evros, Avas, Filiouri, Kompsatos, Vistonis, Kossinthos, Laspias, Nestos, Marmaras, Nevrokopi, Strymon, Ladopotamos, Mavrolakas, Asprolakas, Rihios, Volvi, Doirani, Anthemountas, Gallikos, Marmara Sea region, Thrace region, Meric River, Lake Gala, Lake Hamam
Western Anatolia	Aegean coastal basin, Lake Koycegiz, Esen River, Bakırçay River, Gediz River, Cine River, Lake Isıklı, Lake Marmara , Lake Bafa, Menderes River, Dalaman River, Sarıcay River
Southern Anatolia	North-eastern Med, Eastern Anatolia, Asi River, Lake Amik, Ceyhan River, Dicle River, Manavgat River, Seyhan River, Aksu River
Central Anatolia	Lake Aksehir, Lake Eber, Lake Beysehir, Lake Egirdir, Lake Mogan, Lake Salda, Lake Golhisar, Lake Akgol, Insuyu river, Eregli Marsh, Lake Burdur, Lake Karamık, Lake Golcuk , Lake Abant, Lake Manyas, Lake Iznik, Lake Uluabat, Lake Sapanca
Northern Anatolia	Black Sea costal, Western Black Sea, Coruh River, Aras River, Sakarya River, Kızılırmak River, Yesilırmak River, Lake Cıldır, Yuksekova
California	Tomales Bay, Russian River, Gualala River, Garcia River, Navarro River, Big River, Noyo River, Matolle River, Bear River, Eel River, Mad River. Little River, Redwood Creek, Smith River, San Diego, San Luis Rey, Santa Margarita, Los Angeles, Santa Clara, Santa Ynez, Santa Maria, San Luis Obispo, Morro, Big Sur, Carmel River, Goose Lake, Pit River, McCloud River, Central Valley, Clear Lake, Monterey Bay, Kern River
Chile	Maipo, Rapel, Maule, Itata, Bio-Bio, Imperial, Tolten
SW Australia	Arrowsmith, Hill, Moore, Swan, Canning, Serpentine, Murray, Harvey, Collie, Preston, Brunswick, Capel, Abba-Ludlow, Carbanup, Vasse, Margaret, Blackwood, Donnelly, Warren, Gardner, Shannon, Deep, Walpole, Frankland, Kent, Denmark, Hay, King, Kalgan, Goodga, Angove. Waychinnicup, Pallinup
SW Cape	Olifants-Doring, Verlorenvlei, Berg, Diep, Liesbeek-Black-Salt, Keisers, Hout Bay, Eerste-Kuils, Lourens, Palmiet, Bot, Onrus, Klein (Hermanus), Uilkraal, Haelkraal, Ratels, Heuningnes, De Hoop, Breede, Duiwenhoks, Goukou, Gouritz, Klein Brak, Groot Brak, Gwaing, Swart, Kaaimans, Diep(E), Homtini, Karatara, Hoekraal, Knysna, Piesang, Keurbooms, Groot, Bloukrans, Elandbos, Kleinbos, Storms, Elands, Groot(E), Klasies, Krom, Gamtoos, Van Stadens, Maitland, Bakens, Swartkops

Appendix A.4.2: Species introduced into the Mediterranean-climate regions

Species	Authority	Family	Order	Native range
Acipenser transmontanus	Richardson, 1836	Acipenseridae	Acipenseriformes	P N Am
Odontesthes bonariensis	(Valenciennes, 1835)	Atherinopsidae	Atheriniformes	S Am
Cheirodon interruptus	(Jenyns, 1842)	Characidae	Characiformes	S Am
Alosa sapidissima	(Wilson, 1811)	Clupeidae	Clupeiformes	E N Am
Dorosoma petenense	(Gunther, 1867)	Clupeidae	Clupeiformes	E N Am
Catostomus fumeiventris	Miller, 1973	Catostomidae	Cypriniformes	W N Am
Catostomus platyrhynchus	(Cope, 1874)	Catostomidae	Cypriniformes	W N Am
Abramis bjoerkna	(Linnaeus, 1758)	Cyprinidae	Cypriniformes	N Eur
Alburnus alburnus	(Linnaeus, 1758)	Cyprinidae	Cypriniformes	N Eur
Carassius auratus	(Linnaeus, 1758)	Cyprinidae	Cypriniformes	E Asia
Ctenopharyngodon idella	(Valenciennes, 1844)	Cyprinidae	Cypriniformes	E Asia
Cyprinella lutrensis	(Baird & Girard, 1853)	Cyprinidae	Cypriniformes	E N Am
Cyprinis carpio	Linnaeus, 1758	Cyprinidae	Cypriniformes	C Eurasia
Labeobarbus aeneus	(Burchell, 1822)	Cyprinidae	Cypriniformes	Afr
Notemigonus crysoleucas	(Mitchill, 1814)	Cyprinidae	Cypriniformes	E N Am
Pimephales promelas	Rafinesque, 1820	Cyprinidae	Cypriniformes	E N Am
Puntius conchonius	(Hamilton, 1822)	Cyprinidae	Cypriniformes	E Asia
Richardsonius egregious	(Girard, 1858)	Cyprinidae	Cypriniformes	W N Am
Rutilus rutilus	(Boulenger, 1890)	Cyprinidae	Cypriniformes	C Eurasia
Scardinius erythrophthalmus	(Linnaeus, 1758)	Cyprinidae	Cypriniformes	N Eur
Tinca tinca	(Linnaeus, 1758)	Cyprinidae	Cypriniformes	N Eur
Menidia beryllina	(Cope, 1867)	Atherinidae	Cyprinodontiformes	E N Am
Aphanius fasciatus	(Valenciennes, 1821)	Cyprinodontidae	Cyprinodontiformes	Mediterranean
Lucania parva	(Baird & Girard, 1853)	Cyrpinodontidae	Cyprinodontiformes	E N Am
Fundulus heteroclitus	(Linnaeus, 1758)	Fundulidae	Cyprinodontiformes	E N Am
Cnesterodon decemmaculatus	(Jenyns, 1842)	Poeciliidae	Cyprinodontiformes	S Am
Gambusia affinis	(Baird & Girard, 1853)	Poeciliidae	Cyprinodontiformes	E N Am
Gambusia holbrooki	Girard, 1859	Poeciliidae	Cyprinodontiformes	E N Am
Phalloceros caudimaculatus	(Hensel, 1868)	Poeciliidae	Cyprinodontiformes	S Am
Poecilia latipinna	(Lesueur, 1821)	Poeciliidae	Cyprinodontiformes	E N Am
Poecilia reticulate	Peters, 1859	Poeciliidae	Cyprinodontiformes	S Am
Esox Lucius	Linnaeus, 1758	Esocidae	Esociformes	Circiumpolar
Culaea inconstans	(Kirkland, 1840)	Gasterosteidae	Gasterosteiformes	E N Am
Lepomis cyanellus	Rafinesque, 1819	Centrarchidae	Perciformes	E N Am
Lepomis gibbosus	(Linnaeus, 1758)	Centrarchidae	Perciformes	E N Am
Lepomis gulosus	(Cuvier, 1829)	Centrarchidae	Perciformes	E N Am
Lepomis macrochirus	Rafinesque, 1819	Centrarchidae	Perciformes	E N Am
Lepomis microlophus	(Gunther, 1859)	Centrarchidae	Perciformes	E N Am
Micropterus coosae	Hubbs & Bailey, 1940	Centrarchidae	Perciformes	E N Am
Micropterus dolomieu	Lacepede, 1802	Centrarchidae	Perciformes	E N Am
Micropterus punctulatus	(Rafinesque, 1819)	Centrarchidae	Perciformes	E N Am
Micropterus salmoides	(Lacepede, 1802)	Centrarchidae	Perciformes	E N Am
Pomoxis annularis	Rafinesque, 1818	Centrarchidae	Perciformes	E N Am
Pomoxis nigromaculatus	(Lesueur, 1829)	Centrarchidae	Perciformes	E N Am
Australoheros facetus	(Jenyns, 1842)	Cichlidae	Perciformes	S Am
Geophagus brasiliensis	(Quoy & Gaimard, 1824)	Cichlidae	Perciformes	S Am
Oreochromis aureus	Steindachner, 1864	Cichlidae	Perciformes	Afr
Oreochromis mossambicus	(Peters, 1852)	Cichlidae	Perciformes	Afr
Oreochromis niloticus	(Linnaeus, 1758)	Cichlidae	Perciformes	Afr

Species	Authority	Family	Order	Native range
Tilapia sparrmanii	A. Smith, 1840	Cichlidae	Perciformes	Afr
Tilapia zilli	(Gervais, 1848)	Cichlidae	Perciformes	Afr
Acanthogobius flavimanus	(Temminck & Schlegel, 1845)	Gobidae	Perciformes	E Asia
Tridentiger bifasciatus	Steindachner, 1881	Gobidae	Perciformes	E Asia
Morone chrysops	(Rafinesque, 1820)	Percichthyidae	Perciformes	E N Am
Perca flavescens	(Mitchill, 1814)	Percidae	Perciformes	E N Am
Perca fluviatilis	Linnaeus, 1758	Percidae	Perciformes	N Eur
Percina macrolepida	Stevenson, 1971	Percidae	Perciformes	E N Am
Sander lucioperca	(Linnaeus, 1758)	Percidae	Perciformes	N Eur
Bidyanus bidyanus	(Mitchell, 1838)	Tetrapontidae	Perciformes	Australia
Hypomesus nipponensis	McAllister, 1963	Osmeridae	Salmoniformes	E Asia
Coregonus clupeaformis	(Mitchill, 1818)	Salmonidae	Salmoniformes	P N Am
Hucho hucho	(Linnaeus, 1758)	Salmonidae	Salmoniformes	C Eurasia
Oncorhynchus kisutch	(Walbaum, 1792)	Salmonidae	Salmoniformes	N Pacific
Oncorhynchus mykiss	(Walbaum, 1792)	Salmonidae	Salmoniformes	N Pacific
Oncorhynchus nerka	(Walbaum, 1792)	Salmonidae	Salmoniformes	N Pacific
Oncorhynchus tshawytscha	(Walbaum, 1792)	Salmonidae	Salmoniformes	N Pacific
Salmo trutta	Linnaeus, 1758	Salmonidae	Salmoniformes	N Eur
Salvelinus fontinalis	(Mitchill, 1814)	Salmonidae	Salmoniformes	P N Am
Salvelinus namaycush	(Walbaum, 1792)	Salmonidae	Salmoniformes	P N Am
Clarias gariepinus	(Burchell, 1822)	Clariidae	Siluriformes	Afr
Ameiurus catus	(Linnaeus, 1758)	Ictaluridae	Siluriformes	E N Am
Ameiurus melas	(Rafinesque, 1820)	Ictaluridae	Siluriformes	E N Am
Ameiurus natalis	(Lesueur, 1819)	Ictaluridae	Siluriformes	E N Am
Ameiurus nebulosus	(Lesueur, 1819)	Ictaluridae	Siluriformes	E N Am
Ictalurus furcatus	(Valenciennes, 1840)	Ictaluridae	Siluriformes	E N Am
Ictalurus punctatus	(Rafinesque, 1818)	Ictaluridae	Siluriformes	E N Am
Silurus glanis	Linnaeus, 1758	Siluridae	Siluriformes	C Eurasia

Appendix A.4.5a: The global number of freshwater species in each order compiled from Nelson (2006), Berra (2007), and Eschmeyer and Fricke (2001)

Order	No. of species
Ceratodontiformes	6
Polypteriformes	16
Acipenseriformes	14
Lepisosteiformes	6
Amiiformes	1
Hiodontiformes	2
Osteoglossiformes	218
Anguilliformes	15
Clupeiformes	79
Gonorynchiformes	31
Cypriniformes	3268
Characiformes	1674
Siluriformes	2740
Gymnotiformes	134
Esociformes	10
Osmeriformes	82
Salmoniformes	45
Percopsiformes	9
Ophidiiformes	5
Gadiformes	1
Batrachoidiformes	6
Mugiliformes	1
Atheriniformes	210
Beloniformes	98
Cyprinodontiformes	996
Gasterosteiformes	21
Symbranchiformes	96
Scorpaeniformes	60
Perciformes	2040
Pleuronectiformes	10
Tetraodontiformes	14

Appendix A.4.5b: The global number of freshwater species in each family compiled from Nelson (2006), Berra (2007), and Eschmeyer and Fricke (2001)

Order	**Family**	# of species
Ceratodontiformes	Ceratodontidae	1
Lepidosireniformes	Lepidosirenidae	1
Lepidosireniformes	Protopteridae	4
Polypteriformes	Polypteridae	12
Acipenseriformes	Acipenseridae	26
Acipenseriformes	Polyodontidae	2
Lepisosteiformes	Lepisosteidae	7
Amiiformes	Amiidae	1
Osteoglossiformes	Osteoglossidae	8
Osteoglossiformes	Arapaimidae	2
Osteoglossiformes	Pantodontidae	1
Osteoglossiformes	Hiodontidae	2
Osteoglossiformes	Notopteridae	10
Osteoglossiformes	Mormyridae	195
Osteoglossiformes	Gymnarchidae	1
Anguilliformes	Anguillidae	20
Clupeiformes	Denticipitidae	1
Clupeiformes	Clupeidae	92
Clupeiformes	Engraulidae	17
Clupeiformes	Pristigasteridae	5
Clupeiformes	Sundasalangidae	7
Gonorynchiformes	Kneriidae	30
Gonorynchiformes	Phractolaemidae	1
Cypriniformes	Cyprinidae	2051
Cypriniformes	Psilorhynchidae	18
Cypriniformes	Cobitidae	234
Cypriniformes	Balitoridae	785
Cypriniformes	Vaillantellidae	3
Cypriniformes	Gyrinocheilidae	3
Cypriniformes	Catostomidae	76
Characiformes	Citharinidae	8
Characiformes	Distichodontidae	96
Characiformes	Alestidae	120
Characiformes	Hepsetidae	1
Characiformes	Hemiodontidae	29
Characiformes	Parodontidae	29
Characiformes	Curimatidae	101
Characiformes	Prochilodontidae	21
Characiformes	Anostomidae	143
Characiformes	Chilodontidae	8
Characiformes	Erythrinidae	16
Characiformes	Lebiasinidae	66

Order	**Family**	# of species
Characiformes	Gasteropelecidae	10
Characiformes	Ctenoluciidae	7
Characiformes	Acestrorhynchidae	14
Characiformes	Cynodontidae	14
Characiformes	Serrasalmidae	83
Characiformes	Characidae	1074
Characiformes	Crenuchidae	82
Siluriformes	Diplomystidae	6
Siluriformes	Lacantuniidae	1
Siluriformes	Ictaluridae	51
Siluriformes	Bagridae	208
Siluriformes	Cranoglanididae	5
Siluriformes	Austroglanididae	3
Siluriformes	Siluridae	97
Siluriformes	Schilbeidae	65
Siluriformes	Pangasiidae	28
Siluriformes	Amblycipitidae	33
Siluriformes	Amphiliidae	80
Siluriformes	Akysidae	57
Siluriformes	Sisoridae	185
Siluriformes	Erethistidae	33
Siluriformes	Clariidae	111
Siluriformes	Heteropneustidae	4
Siluriformes	Claroteidae	85
Siluriformes	Chacidae	3
Siluriformes	Olyridae	4
Siluriformes	Malapteruridae	19
Siluriformes	Ariidae	75
Siluriformes	Anchariidae	6
Siluriformes	Plotosidae	31
Siluriformes	Mochokidae	204
Siluriformes	Doradidae	88
Siluriformes	Auchenipteridae	105
Siluriformes	Pimelodidae	103
Siluriformes	Pseudopimelodidae	34
Siluriformes	Heptapteridae	199
Siluriformes	Cetopsidae	42
Siluriformes	Hypophthalmidae	4
Siluriformes	Aspredinidae	39
Siluriformes	Nematogenyidae	1
Siluriformes	Trichomycteridae	240
Siluriformes	Callichthyidae	199
Siluriformes	Loricariidae	813
Siluriformes	Scoloplacidae	5
Siluriformes	Astroblepidae	54

Order	**Family**	# of species
Gymnotiformes	Sternopygidae	29
Gymnotiformes	Apteronotidae	84
Gymnotiformes	Rhamphichthyidae	14
Gymnotiformes	Hypopomidae	20
Gymnotiformes	Gymnotidae	36
Esociformes	Esocidae	5
Esociformes	Umbridae	7
Osmeriformes	Lepidogalaxiidae	1
Osmeriformes	Osmeridae	10
Osmeriformes	Salangidae	20
Osmeriformes	Retropinnidae	6
Osmeriformes	Galaxiidae	50
Salmoniformes	Salmonidae	207
Percopsiformes	Percopsidae	2
Percopsiformes	Aphredoderidae	1
Percopsiformes	Amblyopsidae	6
Gadiformes	Lotidae	1
Ophidiiformes	Bythitidae	5
Batrachoidiformes	Batrachoididae	6
Mugiliformes	Mugilidae	7
Atheriniformes	Atherinidae	27
Atheriniformes	Bedotiidae	16
Atheriniformes	Melanotaeniidae	75
Atheriniformes	Pseudomugilidae	18
Atheriniformes	Atherinopsidae	79
Atheriniformes	Telmatherinidae	18
Atheriniformes	Phallostethidae	21
Cyprinodontiformes	Aplocheilidae	14
Cyprinodontiformes	Nothobranchiidae	240
Cyprinodontiformes	Rivulidae	339
Cyprinodontiformes	Profundulidae	7
Cyprinodontiformes	Fundulidae	44
Cyprinodontiformes	Valenciidae	2
Cyprinodontiformes	Goodeidae	49
Cyprinodontiformes	Poeciliidae	342
Cyprinodontiformes	Cyprinodontidae	121
Cyprinodontiformes	Anablepidae	17
Beloniformes	Belonidae	12
Beloniformes	Hemiramphidae	60
Beloniformes	Adrianichthyidae	31
Elassomatiformes	Elassomatidae	7
Gasterosteiformes	Gasterosteidae	18
Gasterosteiformes	Indostomidae	3
Syngnathiformes	Syngnathidae	**17**
Synbranchiformes	Synbranchidae	22

Order	Family	# of species
Synbranchiformes	Mastacembelidae	83
Synbranchiformes	Chaudhuriidae	10
Scorpaeniformes	Cottidae	42
Scorpaeniformes	Cottocomephoridae	9
Scorpaeniformes	Comephoridae	2
Scorpaeniformes	Abyssocottidae	24
Perciformes	Centropomidae	4
Perciformes	Ambassidae	35
Perciformes	Percichthyidae	41
Perciformes	Perciliidae	2
Perciformes	Latidae	9
Perciformes	Moronidae	6
Perciformes	Terapontidae	33
Perciformes	Kuhliidae	2
Perciformes	Centrarchidae	33
Perciformes	Percidae	224
Perciformes	Apogonidae	11
Perciformes	Datnioididae	5
Perciformes	Sciaenidae	30
Perciformes	Polynemidae	33
Perciformes	Toxotidae	7
Perciformes	Monodactylidae	6
Perciformes	Polycentridae	4
Perciformes	Nandidae	9
Perciformes	Cichlidae	1577
Perciformes	Embiotocidae	1
Perciformes	Bovichtidae	1
Perciformes	Cheimarrichthyidae	1
Perciformes	Rhyacichthyidae	3
Perciformes	Odontobutidae	22
Perciformes	Eleotridae	120
Perciformes	Gobiidae	200
Perciformes	Kurtidae	2
Perciformes	Scatophagidae	1
Perciformes	Anabantidae	32
Perciformes	Osphronemidae	125
Perciformes	Helostomatidae	1
Perciformes	Channidae	31
Pleuronectiformes	Achiridae	0
Pleuronectiformes	Soleidae	1
Pleuronectiformes	Cynoglossidae	6
Tetraodontiformes	Tetraodontidae	20

A.5 Twee River Case Study

Appendix A.5.1: Fish distribution survey results – paper published in the journal *African Journal of Aquatic Sciences*. Marr, S.M., L.M.E. Sutcliffe, J.A. Day, C.L. Griffiths and P.H. Skelton. (2009). Conserving the fishes of the Twee River, Cederberg: revisiting the issues. *African Journal of Aquatic Science* **34**:77-85.

Appendix A.5.2: Water chemistry and nutrient data for the water quality sites in the Suurvlei and Middeldeur Rivers.

Appendix A.5.3: Invertebrate assemblages identified at the water quality sites in the Suurvlei and Middeldeur Rivers (included on the CD included – File: Twee.xlsx, tab inverts)

Appendix A.5.4: Instream and riparian zone transect data for the Suurvlei and Middeldeur Rivers (included on the CD included – File: Twee.xlsx, tab habitat).

Appendix A.5.5: Invertebrate assemblages identified at the water quality sites in the Suurvlei and Middeldeur Rivers (included on the CD included – File: Twee.xlsx, tab inverts)

Appendix A.5.6: Instream and riparian zone transect data for the Suurvlei and Middeldeur Rivers (included on the CD included – File: Twee.xlsx, tab habitat).

Appendix A.5.7: Results of the acetylcholine esterase inhibition analyses of brain tissue of Cape kurper in the Twee River catchment (μmol/min/mg tissue)

Appendix A.5.8: Stable isotope results for the freshwater fishes of the Twee River (included on the CD included – File: Twee.xlsx, tab isotope).

Appendix A.5.9: Length-weight and gape data for Cape kurper from the Twee River catchment (included on the CD included – File: Twee.xlsx, tab Cape kurper length weight and Cape kurper gape)

Appendix A.5.10: Morphometric measurements for Cape kurper in the Twee River catchment (included on the CD included – File: Twee.xlsx, tab Cape kurper morphometrics).

Appendix A.5.2: Water chemistry and nutrient data for the water quality sites in the Suurvlei and Middeldeur Rivers.

Site	pH	Cond (µS/cm)	DO (%)	DO (mg/L)	Temp	Turbidity (NTU)	Ammonium (uM/L)	Nitrite (uM)	Nitrate (uM/L)	SRP (uM/L)
SUUR01	6.3	18.84	102.4	9.83	19.5	0.86	0.00	0.05	8.30	0.06
SUUR01	5.64	18.84	102	8.82	19.4	0.69	0.00	0.05	8.30	0.06
SUUR01	5.35	18.85	101.6	8.76	19.3	0.68	0.00	0.05	8.30	0.06
SUUR02	6.85	20.9	112.3	10.2	17.5	1.29	0.00	0.09	16.51	0.06
SUUR02	6.48	20.9	106.8	9.65	17.4	0.63	0.00	0.09	16.51	0.06
SUUR02	6.3	20.8	110	9.9	17.5	0.71	0.00	0.09	16.51	0.06
SUUR03	6.91	21.2	108.3	9.28	20.2	1.02	0.00	0.14	16.85	0.06
SUUR03	6.05	21.2	110.6	9.51	19.6	0.8	0.00	0.14	16.85	0.06
SUUR03	5.91	21.1	110.8	9.48	19.9	0.67	0.00	0.14	16.85	0.06
SUUR04	6.51	23.9	80.6	6.27	24.5	1.78	0.67	0.13	22.28	0.06
SUUR04	6.4	23.9	101.5	8.35	21.6	0.95	0.67	0.13	22.28	0.06
SUUR04	5.56	23.9	113.5	9.56	21.3	0.82	0.67	0.13	22.28	0.06
MID01	4.44	24.1	90.4	7.67	19.7	2.35	0.34	0.28	18.20	0.10
MID01	4.54	24.8	92.4	8.01	19	0.8	0.34	0.28	18.20	0.10
MID01	4.87	24.1	101.6	8.74	19.1	0.93	0.34	0.28	18.20	0.10
MID02	5.4	33	91.7	8.1	18.7	0.58	0.73	0.25	20.28	0.03
MID02	5.49	33	81.4	7.12	18.2	1.32	0.73	0.25	20.28	0.03
MID02	5.13	33	95.9	8.5	18.2	1.56	0.73	0.25	20.28	0.03
MID03	4.9	30.9	58.2	5.47	15.3	1.42	0.10	0.28	13.97	0.06
MID03	4.78	30.4	57.1	5.38	15.2	1.97	0.10	0.28	13.97	0.06
MID03	4.73	30.2	57.2	5.44	15.1	2.26	0.10	0.28	13.97	0.06
MID04	6.37	32.5	77.8	6.9	17.8	1.96	0.00	0.28	13.28	0.01
MID04	5.64	32.6	76	6.97	16.4	1.6	0.00	0.28	13.28	0.01
MID04	5.58	32.7	75	6.99	15.6	1.65	0.00	0.28	13.28	0.01

Appendix A.5.7: Results of the acetylcholine esterase inhibition analyses of brain tissue of Cape kurper in the Twee river catchment (μmol/min/mg tissue)

	Site			
Sample	S1	M1	M2	M3
1	1.777	5.209	2.804	2.780
1	0.947	5.228	2.214	2.276
1	1.365	5.218	4.994	4.742
2	2.392	7.460	1.845	1.248
2	2.405	7.220	2.903	1.937
2	2.380	6.058	2.497	2.577
3	0.793		1.581	1.704
3	2.503		1.986	4.194
3	1.734		0.953	4.403
4	2.048		0.732	1.611
4	1.845		2.804	2.571
4	3.339		1.753	4.047
5	1.402		0.910	1.722
5	1.882		1.052	2.140
5	3.280		0.824	1.544
6	1.042		0.744	1.015
6	0.996		2.571	1.445
6	1.089		2.472	1.347
7	1.070			1.636
7	3.290			1.230
7	3.081			2.189
8	3.592			1.950
8	2.159			3.426
8	1.919			2.368
9	3.333			3.622
9	3.112			3.961
9	2.749			4.244
10	2.995			2.860
10	2.891			2.319
10	3.389			1.212
11	1.913			1.611
11	3.542			1.285
11	2.614			0.664
12	0.781			0.953
12	1.341			1.415
12	2.103			1.070
13	2.116			2.571
13	1.667			3.598
13	1.568			3.942

A.6 Perceptions of Anglers

Appendix A.6.1: Angling and Conservation Questionnaires with pictures of the fish used in the survey

Appendix A.6.2: Results of the Angling Questionnaire (included on the CD included – File: Survey.xlsx, tab Angling)

Appendix A.6.3: Results of Question 8 in the Angling Questionnaire (included on the CD included – File: Survey.xlsx, tab Angling (Q8))

Appendix A.6.4: Results of the Conservation Questionnaire (included on the CD included – File: Survey.xlsx, tab Conservation)

Appendix A.6.5: Results of Question 8 in the Conservation Questionnaire (included on the CD included – File: Survey.xlsx, tab Conservation (Q8))

Appendix A.6.6: Demographics of the respondents of the survey (included on the CD included – File: Survey.xlsx, tab Demographics)

Appendix A.6.7: Output from the Conversion Model

Appendix A.6.1 – Angling and Conservation Questionnaires and picture of the fish used in the survey

ANGLING QUESTIONNAIRE

1. Please tell me which of the following FISH have ever heard of. ("X" ALL THAT APPLY)

2. Now please tell me which of these FISH you have ever FISHED for. ("X" ALL THAT APPLY)

3. Could you tell me all the FISH you FISH for regularly? By 'regularly', I mean those you've FISHED for within the past 6 months ("X" ALL THAT APPLY)

4. And now, please tell me which **ONE** species of FISH you FISH for most often.

5. For each FISH you say you use regularly, please tell me approximately how many times you FISHED for the species in the past year. (RECORD THE NUMBER OF UNITS)

	Ever heard of	Ever fished for	Fished for regularly	Fished for most often (Favourite)	# Fish caught in past year
Rainbow trout					
Brown trout					
Smallmouth bass					
Largemouth bass					
Bluegill sunfish					
Carp					
Sharptooth catfish (barbel)					
Sawfin					
Yellowfish					
Whitefish					
Sandfish					
Moggel					
Cape kurper					
Cape galaxias					
Redfin					
Rock catlets					
Mozambique tilapia					
Banded tilapia					

*6. Now, I'd like you to think about everything that you look for in a FISH for angling, and then rate each of the FISH you know, using the following scale: if you think a particular FISH is "perfect", then you would give it a score of "10". On the other hand, if you think it is "terrible", you would give it a "1". In between there are other ratings which you can use to indicate how you feel, for example, if you think a particular FISH is "very good", but not "perfect", you would rate it anywhere between "7" and "9", depending how you feel. Or if you think it is merely "okay", you would give it a "5" or a "6", depending on how you feel.

It doesn't matter whether you have FISHED for the particular FISH or not, we are interested in your opinion of all the FISH you have heard of. Let us begin with…… Using this scale, how would you rate it?

	Terrible									Perfect
Rainbow trout	1	2	3	4	5	6	7	8	9	10
Brown trout	1	2	3	4	5	6	7	8	9	10
Smallmouth bass	1	2	3	4	5	6	7	8	9	10
Largemouth bass	1	2	3	4	5	6	7	8	9	10
Bluegill sunfish	1	2	3	4	5	6	7	8	9	10
Carp	1	2	3	4	5	6	7	8	9	10
Sharptooth catfish (barbel)	1	2	3	4	5	6	7	8	9	10
Sawfin	1	2	3	4	5	6	7	8	9	10
Yellowfish	1	2	3	4	5	6	7	8	9	10
Whitefish	1	2	3	4	5	6	7	8	9	10
Sandfish	1	2	3	4	5	6	7	8	9	10
Moggel	1	2	3	4	5	6	7	8	9	10
Cape kurper	1	2	3	4	5	6	7	8	9	10
Cape galaxias	1	2	3	4	5	6	7	8	9	10
Redfin	1	2	3	4	5	6	7	8	9	10
Rock catlets	1	2	3	4	5	6	7	8	9	10
Mozambique tilapia	1	2	3	4	5	6	7	8	9	10
Banded tilapia	1	2	3	4	5	6	7	8	9	10

*7. Some decisions are extremely important, for example, for many people the decision about whom to marry or whether to get married at all is extremely important. On the other hand, there are many things which people consider to be less important, for example, what brand of paper plates to take on a picnic. Thinking now about the types of FISH you can FISH for, how important to you is the choice of which species to FISH for? ("X" ONLY ONE BOX)

- Extremely important ☐ 1
- Very important ☐ 2
- Moderately important ☐ 3
- Slightly important ☐ 4
- Not at all important ☐ 5

8. I am going to read you some statements about FISH. For each statement, please tell me which brand or brands you think are described by each statement. Let's start with…

	Most challenging to fish for	Most satisfying to fish for	Only for beginners	Only for the serious fisher	Biggest rush	Waste of time	Catching them proves you are a man	The best fighting	The most tasty
Rainbow trout									
Brown trout									
Smallmouth bass									
Largemouth bass									
Bluegill sunfish									
Carp									
Sharptooth catfish (barbel)									
Sawfin									
Yellowfish									
Whitefish									
Sandfish									
Moggel									
Cape kurper									
Cape galaxias									
Redfin									
Rock catlets									
Mozambique tilapia									
Banded tilapia									

*9. Think about each of the FISH you FISH for regularly. Which one statement best describes your feelings about ....? (READ OUT FOR EACH STATEMENT)

	I can think of many good reasons to continue fishing for ..., and no good reasons for me to change.	I can think of many good reasons to continue fishing for ..., but there are also good reasons for me to change.	I can think of few good reasons to continue fishing for..., and there are many good reasons for me to change.
Rainbow trout	1	2	3
Brown trout	1	2	3
Smallmouth bass	1	2	3
Largemouth bass	1	2	3
Bluegill sunfish	1	2	3
Carp	1	2	3
Sharptooth catfish (barbel)	1	2	3
Sawfin	1	2	3
Yellowfish	1	2	3
Whitefish	1	2	3
Sandfish	1	2	3
Moggel	1	2	3
Cape kurper	1	2	3
Cape galaxias	1	2	3
Redfin	1	2	3
Rock catlets	1	2	3
Mozambique tilapia	1	2	3
Banded tilapia	1	2	3

CONSERVATION QUESTIONNAIRE

1. Please tell me which of the following FISH ARE INDIGENOUS TO THE WESTERN CAPE. ("X" ALL THAT APPLY)

2. Now please tell me which of these FISH you consider to be endangered. ("X" ALL THAT APPLY)

3. Could you tell me all the FISH you consider need conservation action ("X" ALL THAT APPLY. IF 'NONE OF THESE' – THANK AND TERMINATE)

4. And now, please tell me which one species of FISH you would support conservation for. (SINGLE MENTION. CHECK THAT THIS BRAND HAS ALSO BEEN MENTIONED IN Q3)

5. For each FISH species you say you support conservation for, please tell me approximately how many times you contributed to the conservation of the species in the past year. (RECORD THE NUMBER OF UNITS)

	Are indigenous to Western Cape	You considered to be endangered	Require conservation action	You would support conservation for	Have contributed towards conservation action for in the last year
Rainbow trout					
Brown trout					
Smallmouth bass					
Largemouth bass					
Bluegill sunfish					
Carp					
Sharptooth catfish (barber)					
Sawfin					
Yellowfish					
Whitefish					
Sandfish					
Moggel					
Cape kurper					
Cape galaxias					
Redfin					
Rock catlets					
Mozambique tilapia					
Banded tilapia					

*6. Now, I’d like you to think about everything that you look for in a FISH for conservation, and then rate each of the FISH you know, using the following scale: if you think a particular FISH has a “high conservation value”, then you would give it a score of “10”. On the other hand, if you think it has “little or no conservation value”, you would give it a “1”.

It doesn’t matter whether you have FISHED for the particular FISH or not, we are interested in your opinion of all the FISH you have heard of. Let us begin with……(ASK FOR ALL FISH). Using this scale, how would you rate it?

	Low Value									High Value
Rainbow trout	1	2	3	4	5	6	7	8	9	10
Brown trout	1	2	3	4	5	6	7	8	9	10
Smallmouth bass	1	2	3	4	5	6	7	8	9	10
Largemouth bass	1	2	3	4	5	6	7	8	9	10
Bluegill sunfish	1	2	3	4	5	6	7	8	9	10
Carp	1	2	3	4	5	6	7	8	9	10
Sharptooth catfish (barber)	1	2	3	4	5	6	7	8	9	10
Sawfin	1	2	3	4	5	6	7	8	9	10
Yellowfish	1	2	3	4	5	6	7	8	9	10
Whitefish	1	2	3	4	5	6	7	8	9	10
Sandfish	1	2	3	4	5	6	7	8	9	10
Moggel	1	2	3	4	5	6	7	8	9	10
Cape kurper	1	2	3	4	5	6	7	8	9	10
Cape galaxias	1	2	3	4	5	6	7	8	9	10
Redfin	1	2	3	4	5	6	7	8	9	10
Rock catlets	1	2	3	4	5	6	7	8	9	10
Mozambique tilapia	1	2	3	4	5	6	7	8	9	10
Banded tilapia	1	2	3	4	5	6	7	8	9	10

*7. Thinking now about the FRESHWATER FISH of the WESTERN CAPE, how important to you is the conservation of the INDIGENOUS SPECIES of FISH? (“X” ONLY ONE BOX)

Extremely important	☐	1
Very important	☐	2
Moderately important	☐	3
Slightly important	☐	4
Not at all important	☐	5

8. I am going to read you some of the THREATS to FRESHWATER FISHES. For each THREAT, please tell me which FRESHWATER FISH species are most threatened by this threat.

	Habitat degradation	Water pollution	Water abstraction	Pesticides	Alien plants	Alien Fish
Rainbow trout						
Brown trout						
Smallmouth bass						
Largemouth bass						
Bluegill sunfish						
Carp						
Sharptooth catfish (barber)						
Sawfin						
Yellowfish						
Whitefish						
Sandfish						
Moggel						
Cape kurper						
Cape galaxias						
Redfin						
Rock catlets						
Mozambique tilapia						
Banded tilapia						

*9. Think about Conservation of the Fish of the Western Cape, which one statement best describes your feelings about ....? (READ OUT FOR EACH STATEMENT)

	I can think of many good reasons to conserve ..., and no good reasons to let a population be extirpated.	I can think of many good reasons to conserve ..., but there are also good reasons to let a population be extirpated.	I can think of few good reasons to conserve..., and there are many good reasons to let a population be extirpated.
Rainbow trout	1	2	3
Brown trout	1	2	3
Smallmouth bass	1	2	3
Largemouth bass	1	2	3
Bluegill sunfish	1	2	3
Carp	1	2	3
Sharptooth catfish (barber)	1	2	3
Sawfin	1	2	3
Yellowfish	1	2	3
Whitefish	1	2	3
Sandfish	1	2	3
Moggel	1	2	3
Cape kurper	1	2	3
Cape galaxias	1	2	3
Redfin	1	2	3
Rock catlets	1	2	3
Mozambique tilapia	1	2	3
Banded tilapia	1	2	3

Angler Demographics

Age Group

Less than 18	18 to 25	25 to 45	46 to 60	More mature

Freshwater Angling Activity

How would you describe your involvement in freshwater angling?

I have	Occasional	Frequent	Very Frequent	Manic

Organizational affiliation

Do you belong to any angling club or organization?

Yes	No	Used to

If YES, please specify which clubs or organizations

Do you take part in any competitive angling?

Yes	No	Used to

If YES, please specify which the competition circuits that you have participated in.

Angling Techniques

Which angling techniques do you use?

Fly	Art Lure	Bait	Other

Which angling techniques do you use most frequently?

Fly	Art Lure	Bait	Other

Do you think that CapeNature are doing a good job of conserving the indigenous freshwater fishes of the Western Cape?

Yes	No

Eradication techniques

In order to remove problem species threatening the indigenous fishes of the Western Cape, eradication techniques are required for the total removal of the problem species from selected water bodies.

	Ever heard of	Believe is effective	Find acceptable	Would support the use of
Netting				
Fishing				
Electrofishing				
Chemical treatment				
Biological control - genetic				


Rainbow trout


Brown trout


Smallmouth bass


Largemouth bass


Bluegill


Carp


Sharptooth catfish - Barbel


Sawfin


Yellowfish


Whitefish


Sandfish


Moggel


Cape kurper


Cape galaxias


Redfin


Rock catlet


Mozambique tilapia


Banded tilapia

Appendix A.6.7: Output of the Conversion Model

No	Ref	Strength of commitment	States of Mind	CM Segments																	
				Rainbow trout	Brown trout	Smallmouth bass	Largemouth bass	Bluegill sunfish	Carp	Sharptooth catfish (barbel)	Mozambique mouthbrooder	Banded tilapia	Sawfin	Yellowfish	Whitefish	Sandfish	Moggel	Cape kurper	Cape galaxias	Redfin	Rock catlets
1	B1	3	3	6	6	3	3	5	3	3	6	6	6	5	4	6	6	6	6	6	6
2	B2	4	5	5	8	6	6	8	4	7	8	8	8	8	8	8	8	6	8	8	8
3	B3	4	3	4	4	4	4	4	4	4	6	6	5	3	3	5	8	8	8	7	8
4	B4																				
5	B5	3	3	7	7	6	4	7	3	3	4	7	5	5	4	7	8	8	8	8	8
6	B6	2	1	8	8	8	2	8	3	8	8	8	8	8	7	8	8	8	8	8	8
7	B7	1	1	8	8	8	8	8	1	3	8	8	8	8	8	8	8	8	8	8	8
8	B8	2	1	8	8	8	8	8	2	8	8	8	8	8	8	8	8	8	8	8	8
9	B9	2	1	8	8	8	8	8	2	7	8	8	8	8	8	8	8	8	8	8	8
10	B10	2	1	8	8	8	8	8	2	7	8	8	8	8	8	8	8	8	8	8	8
11	B11	2	1	8	8	8	8	8	2	4	8	8	8	4	8	8	8	4	8	8	8
12	B12	2	1	4	7	7	2	4	4	7	6	7	7	8	7	7	7	7	7	7	7
13	B13	2	1	4	4	4	4	4	2	4	3	7	6	3	6	6	6	8	8	8	8
14	B14	2	3	6	6		4	4	2	2	6	6	6	6	7	6	6	6	8	8	6
15	B15	3	5	4	4	4	4	7	3	4	5	7	5	4	4	5	4	8	8	8	8
16	A1	3	4	6	8	3	4	8	8	8	7	8	8	5	8	8	8	8	8	8	8
17	A2	2	3	7	7	2	2	7	7	8	8	8	8	8	8	8	8	7	8	7	8
18	A3	1	1	8	8	8	8	8	8	8	1	8	8	8	8	8	8	8	8	8	8
19	A4	2	3	6	6	2	2	4	4		8	8	8	3	8	8	8	8	8	8	8
20	A5	2	1	4	8	2	6	8	6	8	8	8	8	8	7	8	8	8	8	8	8
21	A6	3	4	4	4	3	4	7	7	6	6	8	6	5	6	8	8	8	8	8	8
22	A7	1	1	6	6	1	3	8	7	8	4	8	8	7	8	8	8	8	8	8	8
23	A8	2	2	6	6	6	2	8	4	7	6	7	8	3	8	8	8	8	8	8	8
24	A9	2	1	7	7	3	2	8	7	8	8	8	8	8	8	8	8	8	8	8	8
25	A10	2	1	4	7	3	2	4	8	4	4	4	7	7	7	7	7	4	7	7	7
26	A11	1	1	1	8	4	4	8	7	8	8	8	8	6	8	8	8	8	8	8	8
27	A12	1	1	4	3	1	3	7	8	7	6	7	6	6	6	7	7	8	8	8	8
28	A13	2	3	4	4	2	2	8	4	8	8	8	8	8	8	8	8	8	8	8	8
29	A14	2	3	6	6	2	2	6	4	4	7	7	8	6	6	8	7	7	8	8	8
30	A15	1	1	4	6	1	3	6	8	8	7	7	7	7	7	7	7	7	7	7	7
31	A16	2	3	6	6	2	2	6	4	4	7	7	8	6	7	8	6	8	8	8	8
32	A17	1	1	6	6	4	4	8	3	8	6	7	7	1	7	7	7	8	8	8	8
33	A18	2	3	8	8	2	2	8	8	8	8	8	8	8	8	8	8	8	8	8	8
34	A19	2	3	8	8	2	2	8	8	8	8	8	8	8	8	8	8	8	8	8	8
35	F1	2	3	2	2	8	8	8	8	8	7	7	8	7	7	8	8	8	8	7	8
36	F2	3	3	3	6	4	4	8	8	8	7	8	7	5	7	7	7	8	8	8	8
37	F3	2	2	4	4	6	6	8	7	7	8	8	6	2	6	8	8	8	8	8	8
38	F4	2	1	2	4	6	6	8	8	8	8	8	8	6	6	8	8	8	8	8	8
39	F5	2	2	2	3	8	8	8	8	8	8	8	7	6	7	8	7	8	8	8	8
40	F6	2	1	3	7	3	3	8	6	7	7	7	4	2	6	8	7	8	8	8	8
41	F7	2	1	2	3	7	7	8	4	7	8	8	6	3	6	8	8	8	8	8	8
42	F8	2	3	2	2	2	2	8	8	8	8	8	7	2	8	8	8	8	8	8	8
43	F9	3	1	3	2	8	8	7	7	8	7	7	5	5	5	6	8	5	8	6	8
44	F10	2	1	2	6	4	7	8	6	7	6	8	6	3	6	7	7	7	8	8	8
45	F11	2	3	2	2	8	8	8	8	8	8	8	8	8	8	8	8	8	8	8	8
46	F12	2	3	7	6	4	7	8	8	8	8	8	2	2	6	6	6	7	8	6	8

Table A.6.6: Output of the Conversion Model (cont)

No	Ref	Strength of commitment	States of Mind	CM Segments																	
				Rainbow trout	Brown trout	Smallmouth bass	Largemouth bass	Bluegill sunfish	Carp	Sharptooth catfish (barbel)	Mozambique mouthbrooder	Banded tilapia	Sawfin	Yellowfish	Whitefish	Sandfish	Moggel	Cape kurper	Cape galaxias	Redfin	Rock catlets
47	F13	2	3	2	2	8	8	8	8	8	7	8	3	2	6	7	7	8	8	8	8
48	F14	2	1	2	4	4	4	8	8	7	6	8	8	8	8	8	8	6	8	8	8
49	F15	1	1	3	1	8	8	8	8	8	8	8	7	6	7	8	7	8	8	8	8
50	F16	2	1	4	6	4	4	7	4	7	4	8	4	2	6	8	8	8	8	8	8
51	F17	2	1	2	6	8	8	8	4	8	8	8	8	7	7	8	8	8	8	8	8
52	F18	2	3	2	2	8	8	8	8	7	8	8	7	8	8	8	8	8	8	8	8
53	F19	3	2	3	4	4	4	8	8	8	8	8	4	2	6	6	6	8	8	8	8
54	F20	2	1	2	3	3	3	7	4	6	6	6	7	6	6	6	6	6	6	6	6
55	F21	2	3	2	2	6	4	8	8	7	7	7	7	7	7	7	7	7	7	7	7
56	F22	2	1	2	3	8	6	8	8	8	7	8	8	3	8	8	8	8	8	8	8
57	F23	2	2	2	6	3	3	8	8	8	6	8	8	3	6	8	8	8	8	8	8
58	F24	2	1	2	4	6	3	8	7	8	7	7	7	3	6	7	7	7	7	7	7
59	F25	2	1	3	2	8	8	8	8	8	8	8	8	8	8	8	8	8	8	8	8
60	F26	2	3	2	2	3	4	4	8	4	8	8	4	2	6	8	8	8	8	8	8
61	F27	3	5	3	3	4	4	7	4	6	6	7	5	5	5	5	5	7	7	7	7
62	F28	2	3	3	2	6	6	8	4	8	4	7	2	2	2	6	6	6	8	6	8
63	F29	2	3	2	2	4	3	8	8	8	8	8	6	6	6	8	8	8	8	8	8
64	F30	2	3	2	2	2	7	7	7	8	7	7	7	7	7	8	8	7	7	7	7
65	F31	2	3	2	2	3	3	4	6	8	7	7	6	6	7	8	8	8	8	8	8
66	F32	2	3	2	2	3	3	8	8	8	8	8	8	8	8	8	8	8	8	8	8
67	F33	2	3	2	2	8	8	8	8	8	8	8	8	8	8	8	8	8	8	8	8
68	F34	1	1	1	3	8	8	8	8	8	8	8	8	3	6	8	8	8	8	8	8
69	F35	2	3	2	2	7	7	8	8	8	8	8	8	7	8	8	8	8	8	8	8
70	F36	2	1	2	3	4	4	7	4	4	6	6	3	3	3	6	6	6	6	6	6