THE ICHTHYOFAUNA OF THE WILDERNESS LAKES SYSTEM, WESTERN CAPE, WITH PARTICULAR EMPHASIS ON ALIEN FISH SPECIES AND THEIR ESTABLISHMENT SUCCESS

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GENERAL ABSTRACT

Freshwater fish species have been introduced into freshwater systems around the world, primarily for aquaculture, ornamental fish trade and sport fishing. Their introduction into estuarine systems is uncommon however, instances do occur and their establishment success and impacts on these estuarine systems is not well documented. The extent of invasion by four freshwater fishes in a RAMSAR listed estuarine system, the Wilderness Lakes, Western Cape was investigated. This thesis determined the relative abundance and distribution of alien fishes in relation the native fish biota, their establishment success in the system, what factors inhibited their establishment and whether the introduction of alien fishes negatively impacted the native fish biota. The distribution and abundance of fishes were assessed primarily using fyke nets, seine nets and gill nets in each of the lakes, interconnecting channels and the Touw Estuary. The fish fauna was made up of euryhaline marine species comprising 46%, native estuarine species comprising 18%, catadromous species comprising 7% and freshwater alien species comprising 29% of the total biomass sampled. A total of 26 species were sampled in the system, three of which were considered alien; *Oreochromis mossambicus*, *Gambusia affinis* and *Cyprinus carpio,* and *Micropterus salmoides* were not sampled but confirmed in the system. Establishment success was determined by evidence of: a sustainable breeding population, a wide distribution, abundant in the sampling area, and all size classes of fish sampled. *Gambusia affinis* and *Oreochromis mossambicus* have been recorded in the system for a minimum of 13 and 26 years respectively. They were widely distributed and highly abundant and are established in the system. *Micropterus salmoides* was first recorded in the system in 1985 but abundances have remained low with fish appearing to be limited to Island Lake and Langvlei. Reproduction appeared to be limited by higher salinity and these factors indicated that this was a casual species which relies on repeated introductions for population maintenance. *Cyprinus carpio* spawned successfully in 2010 and was widely distributed but abundances were still low with a total of 15 fish being sampled throughout the system, and was thus in the establishing phase. As these are considered freshwater alien species, the physico-chemical parameters in the estuarine environment inhibiting the establishment success of the alien fishes were investigated. *Gambusia affinis* and *O. mossambicus* were not limited by the physical environment, and while *O. mossambicus* cannot tolerate temperatures below 11^{o}C, temperatures only dropped below its tolerance for a total of two days between February 2010 and February 2011. *Cyprinus carpio* and *M. salmoides* were restricted by salinity in Rondevlei and Langvlei but could tolerate salinity in Island Lake and the Touw Estuary during closed mouth phases. While adults appeared to tolerate the salinity in the system, egg and larval development could be affected thus reducing the viability of the population. The abundance of alien fishes did not negatively impact the abundance of native fish species. The interactions between the native and alien fish biota tended towards biotic acceptance where, as alien abundance increased so did native fish abundance. The impacts of the alien fishes on the native fish biota were assessed by comparing the fish community from a study completed in 1985 to the findings of this study. From these two studies there were no apparent negative impacts on the native fish biota and the fish community composition would most likely be structured by estuarine mouth opening events.

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Contents

CHAPTER 1

GENERAL INTRODUCTION

South African estuaries

South Africa has approximately 250 estuaries along its coastline (James & Harrison 2010) which play an important role in promoting fish species richness in South Africa (Vorwerk 2000). They are the meeting place of rivers and seas and are dynamic environments characterised by large fluctuations in environmental conditions (Harrison & Whitfield 2006; James & Harrison 2010). Estuaries are characterised by influx of saltwater from the ocean and freshwater from rivers and tributaries. They are important nursery areas for several species of marine fishes, many of which are exploited in estuaries or coastal fisheries later in their life cycles (James & Harrison 2010).

The estuaries along the Southern African coast are divided into three biogeographic zones which were delineated in terms of water temperature, rainfall, river flow, marine biota and estuarine fishes (Day 1981; Potter *et al*. 1990; Whitfield 1994a; Harrison 2002). The cool-temperate zone extends from the Orange River to Cape Agulhas, the warm-temperate region extends from Cape Agulhas along the south, south-east and east coast to just south of Port St Johns and the subtropical region extends along the east coast from Port St Johns northwards to Kosi Bay (Harrison 2002). The species diversity is lowest in the cool-temperate region and increases up the coast towards the subtropical region with a gradual shift in species composition (Maree *et al*. 2000; Harrison 2002).

South African estuaries are highly variable habitats in which conditions such as salinity, temperature, turbidity and water currents can fluctuate rapidly, both temporally and spatially (Whitfield 1994b). The fish community structure in estuaries is highly dependent on both biotic and abiotic factors (Harrison & Whitfield 2006). Physico-chemical parameters such as salinity and temperature can strongly influence species composition, abundance and distribution in estuarine waters with salinity fluctuations appearing to be the major factor governing the diversity and abundance of fishes in estuaries (Whitfield 1994b; Harrison

2004; Harrison & Whitfield 2006). South African estuaries are under threat from habitat destruction, over exploitation, alteration of natural water courses and the introduction of non-native species (Chornesky & Randall 2003; Casal *et al*. 2007; Pejchar & Mooney 2009; James & Harrison 2010).

Fish communities in temporary open/closed estuaries

According to Whitfield (1994a) fish that utilise estuaries can be divided into two broad groups according to their spawning locations. The marine group which spawn at sea comprises large species that enter estuaries mainly as juveniles and usually return to sea prior to attaining sexual maturity. The estuarine group is made up predominantly of small sized species which have the ability to complete their life-cycle within estuaries (Whitfield 1994b). The native estuarine component is made up predominantly of smaller species (< 70 mm SL at sexual maturity) with Gobiidae, Syngnathidae, Clupidae, Clinidae, Atherinidae and Hemiramphidae all being common and abundant in temporary open/closed estuaries (Whitfield 1998; Vorwerk *et al*. 2001; James 2006; James & Harrison 2010). The marine spawning component dominates estuaries and the communities are made up mostly of larger marine species, most of which mature at > 200 mm SL (Whitfield 1990) where the juveniles are present in estuaries prior to maturity (Whitfield 1990; James & Harrison 2010). Smaller proportions of freshwater species and pure marine species occur in estuaries along the coast of southern Africa (Whitfield 1990; Whitfield 1998; Maree *et al*. 2000; Vorwerk *et al*. 2001; James 2006).

Invasive fish species in a global context

Invasive species are considered a threat to aquatic ecosystems worldwide (Pimentel *et al*. 2001). There are reports of dramatic changes in species diversity, abundance and energy flow following the establishment of a single new species (Casal *et al*. 2007). Fishes are among the most introduced group of aquatic animals in the world, with over 620 species being introduced to new systems worldwide (Gozlan *et al*. 2010). The purpose of introduction of alien species differs between species, however Gozlan (2008) determined the primary reasons for introduction were demands for fish products for food aquaculture (51%), ornamental fish (21%), sport fishing (12%) and fisheries (7%). Nearly 8% of all introductions are accidental with most of these attributed to escapes from aquaculture facilities (Gozlan *et al*. 2010). Non-native species have expanded their ranges by moving through aquatic connections created by humans and from legal introductions to enhance sport and commercial fisheries for ecological management (such as mosquito control), and through illegal releases of bait and aquarium fishes and escapes from fish farms (Gido & Brown 1999). The result of these introductions is that many ecosystems have been altered drastically by alien species (Gido & Brown 1999). There are currently eight fish species which rank in the world's 100 worst invasive species list (Lowe *et al*. 2000). These species are the brown trout (*Salmo trutta*), common carp (*Cyprinus carpio*), largemouth bass (*Micropterus salmoides*), Mozambique tilapia (*Oreochromis mossambicus*), Nile perch (*Lates niloticus*), rainbow trout (*Oncorhynchus mykiss*), walking catfish (*Clarias batrachus*) and the western mosquitofish (*Gambusia affinis*). Many of the world's aquatic fauna are in immediate danger of extinction due to biological invasion. Invasive species have the ability to displace native species, diminish diversity, disrupt ecosystems and cause significant economic impacts. About 80% of the world's endangered species are at risk due to non-native species introductions (Casal *et al*. 2007).

Terminology from recent literature

Alien species- a species occurring outside its natural range and dispersal potential that might survive and subsequently reproduce, whose presence and dispersal is due to intentional or unintentional human action (Cambray 2003; Copp *et al*. 2005; Gozlan *et al*. 2010).

Invasive species– non-native species that has spread beyond the introduction site and become abundant and is an agent of change, and threatens native biological diversity (Cambray 2003; García-Berthou 2007).

Native species- an organism that has originated in a specific area without human involvement or that has arrived there without intentional or unintentional intervention of humans (Gozlan *et al*. 2010; Richardson *et al*. 2011).

Establishment- the process whereby an introduced species reproduces and forms a self-sustaining population (Gozlan *et al*. 2010). Weyl *et al*. (2009) classified alien fish as established when they were widely distributed in the sampling region, comprised a large proportion of the catch composition, all size classes were sampled and had formed a self sustaining population.

Alien fish fauna in estuaries

The introduction of non-native fish species is a large component of global change with a range of potential impacts on systems such as habitat degradation, extinction of native biota, changes in ecosystem functioning and the facilitation of subsequent invasions (Brown & Walker 2004; Smith & Walker 2004; Gurevitch & Padilla 2004; Copp *et al.* 2005; Canonico *et al*. 2005; Arim *et al*. 2006; Vitule *et al*. 2009). Freshwater invasive fish species have been recorded in estuaries and their associated rivers around South Africa including the Kowie River (Weyl & Lewis 2006; Wasserman & Strydom 2011), Great Fish River, (James *et al*. 2007) and Groot Brak Estuary (James & Harrison 2010). In each of these estuaries, the introductions of alien fish were not intentional. In many cases, by the time these species were identified in the system, they were established and in many cases had formed a successful breeding population (De Moor 1996).

Previous studies (Hall *et al*. 1987; Russell 1996, 1999b) on the Wilderness Lakes system reported three of the worlds 100 worst invasive freshwater fishes (Lowe *et al*. 2000) from the system, namely Mozambique tilapia *O. mossambicus*, western mosquitofish *G. affinis* and largemouth bass *M. salmoides*. Hall (1985) documented the presence of *O. mossambicus* and *M. salmoides*. *Gambusia affinis* and *M. salmoides* were reported in the Touw and Duiwe rivers by Russell (1999b). This study reported the presence of *O. mossambicus*, *M. salmoides* and *G. affinis* and was the first to report the presence of the common carp (*C. carpio*) in the Wilderness Lakes system. Each of species listed above is one of eight fish species currently on the world's 100 worst invasive species list (Lowe *et al*. 2000) with each originating from different continents and having been introduced for different reasons, they all pose possible threats to the system.

Oreochromis mossambicus

Oreochromis mossambicus (**Fig. 1.1**) has a native range from the lower Zambezi, inland Zimbabwe and the former Transvaal lowveld (now Mpumalanga) and south of the Tugela system along the narrowing coastal strip to the Eastern Cape (de Moor 1996). This species occurs naturally in estuaries on the east coast of South Africa down to the Bushmans River in the Eastern Cape 300 km north of the Wilderness Lakes (de Moor & Bruton 1988; Skelton 2001). It has been known to occur in the Wilderness Lakes system since the 1980s (Hall 1985), though its origins in the region are unknown (de Moor & Bruton 1988).

a) b)

Figure 1.1. a) Mature *Oreochromis mossambicus* (female above, male below) and b) juvenile *Oreochromis mossambicus* sampled in the Wilderness Lakes system, Western Cape between April 2010 and May 2011.

Oreochromis mossambicus is highly tolerant of a wide range of environmental variables, which makes it a popular aquaculture species (de Moor & Bruton 1988). It has been known to tolerate salinities of up to 120 ‰ with gradual acclimatization (Whitfield & Blaber 1979) and can tolerate temperatures between ten and 38^{o}C (Jobling 1981). As a result, *O. mossambicus* can adapt to many systems outside its natural range (Allanson *et al*. 1971; Chmilevskii 1998).

Maturity occurs at a young age and relatively small size. The size at maturity is very variable between systems and is dependent on environmental factors linked to food and temperature (Weyl & Hecht 1998). In Mnjoli Dam in Swaziland, the smallest sexually mature male and female *O. mossambicus* were 205 mm TL and 134 mm TL respectively (Khumalo 2006). In Lake Chicamba, Mozambique fish attained sexual maturity (Lm_{50}) at 251 mm TL (males) and 223 mm TL (females) (Weyl & Hecht 1998). In Lake Sibaya, the size at first maturity was 80 mm for females and 100 mm SL for males at an age of one and two years respectively (James & Bruton 1992). Numerous studies have determined the age-length relationship for *O. mossambicus* in African and Australian lakes (Arthington & Milton 1986; Weyl & Hecht 1998; Khumalo 2006). These studies showed maximum size of age zero fish being 190 mm, age one being between 80 and 240 mm, and age two fish ranging from 131 to 250 mm (Arthington & Milton 1986; Weyl & Hecht 1998; Khumalo 2006). *Oreochromis mossambicus* attain sexual maturity between the ages of one and three years (Arthington & Milton 1986; James & Bruton 1992; Weyl & Hecht 1998; Khumalo 2006).

The breeding season of *O. mossambicus* occurs between spring and summer, with adults migrating to the shallow inshore where the males are reported to construct nests into which females deposit their eggs (Bruton & Boltt 1975; Caskey *et al*. 2007). *Oreochromis mossambicus* are maternal mouth brooders, with females releasing the fry into very shallow water after 14 - 22 days of incubation where the juveniles form shoals (de Moor & Bruton 1988; Amorim *et al.* 2003; Caskey *et al*. 2007).

Oreochromis mossambicus is a broad spectrum omnivore (Costa-Pierce 2003). It is predominantly a detritivore but young *O. mossambicus* feed on zoobenthos and zooplankton (de Moor & Bruton 1988). Detritus forms a large component of the diet of adult fish in the Gascoyne River, Australia, and smaller fish in the Chapman River. The detritus is comprised of algae, vegetable matter, the diatom Bacillariophyceae and small quantities of inorganic matter including silt (Maddern *et al.* 2007). De Silva (1986) noted that *O. mossambicus* is omnivorous in Sri Lankan reservoirs; though a detritivorous diet is adequate for reproduction and normal growth. It is able to switch from one food source to another and can be a fierce predator on small fish fry (Costa-Pierce 2003). However, in most systems, the diet of *O. mossambicus* is made up predominantly of planktonic and epiphytic algae and macrophytes (Arthington & Milton 1986).

The introduction of *O. mossambicus* into Australian waterways has been primarily as an ornamental species. It has a rapid establishment rate in areas around Brisbane. *Oreochromis mossambicus* has been introduced to Tingalpa Reservoir since 1977 and in North Pine Dam since 1973. This species had established a breeding population by 1981 (Arthington & Milton 1986). In Gascoyne River, Australia, *O. mossambicus* was sampled in 1981 and spread rapidly throughout the main tributary, the Lyons River (Maddern *et al.* 2007).

Gambusia affinis

Gambusia affinis, a species native to southern America and northern Mexico, were deliberately introduced to South Africa in 1936 for mosquito control (Skelton & Weyl 2011). The earliest report of introduction from the Wilderness region is 1972 from the Karatara River in Sedgefield (de Moor & Bruton 1988). *Gambusia affinis* generally occur in shallow, slow flowing water that is densely vegetated. In deeper water *G. affinis* tend to be found along the shallow, well vegetated edges (Pyke 2008). They are able to tolerate a wide range of physical conditions, occurring in waters with temperatures of up to 44^{o}C (Jobling 1981; Nordlie 2006) and salinities of up to 41 ‰ (Chervinski 1983; Pyke 2005, 2008; Uliano *et al*.

2010). While *G. affinis* are able to tolerate a wide range of environmental variables, they are not often associated with estuaries.

Figure 1.2. Mature *Gambusia affinis* with male above and female below (source: Bishop Museum.org).

Gambusia affinis is a sexually dimorphic species where the females are larger than the males. Female fish give birth repeatedly in a spawning season. They are live-bearing fish with clutches ranging from five to over 100 juveniles per clutch (Pyke 2005). Females reach sexual maturity at approximately 14 mm in length. Males reach sexual maturity from eight mm and attain a size of approximately 30 mm, whereas females continue to grow to attain a length of 38.5 mm (Sloterdijk 2011). Females born early in the breeding season are smaller at sexual maturity, while females born late in the season will over-winter and spawn the following season, thus reaching a larger size at sexual maturity (Haynes & Cashner 1995; Pyke 2005). Females spawn repeatedly over one season and four to five generations occur during a breeding season, with newborn *G. affinis* being usually eight - nine mm long (Krumholz 1948; Blaylock 1969; Haynes & Cashner 1995). *Gambusia affinis* are short-lived species, with maximum longevity in wild populations seldom exceeding 12 to 15 months. Males die after one breeding season (Pyke 2005, 2008).

Gambusia affinis are opportunistic omnivores; in addition to preying on mosquito larvae, their diet includes algae, crustaceans, insects, and vertebrates, including amphibian larvae and small fishes (Leyse 2004). They are opportunistic feeders and will attack most potential prey encountered; prey preference may be based on a variety of factors including fish size and the size, abundance and activity of potential prey (Daniels & Felley 1992; Leyse 2004). *Gambusia affinis* are also cannibalistic: juvenile fish are eaten during periods of food scarcity (Dionne 1985; Meffe & Crump 1987). Introduced *G. affinis* can substantially reduce zooplankton populations and shift size frequencies of remaining populations through selective predation on larger individuals (Leyse 2004).

Gambusia affinis is one of the most widely distributed freshwater fish species and is listed as one of the world's 100 worst invasive species (Lowe *et al.* 2000; Alcaraz & Garcia-Berthou 2007). The ecological impacts of *G. affinis* are profound and cause cascade effects on ecosystems (Pyke 2005; Alcaraz & García-Berthou 2007).

Micropterus salmoides

The piscivorous *M. salmoides* originated in North America. It was first imported to South Africa in 1928 for sport fishing and has subsequently been introduced into most river catchments around the country (van Rensburg *et al*. 2011; Skelton & Weyl 2011). It is considered to be a freshwater species with limited tolerance for salinity (Meador & Kelso 1989; Norris 2007; Lowe *et al*. 2009). It prefers artificial impoundments and lakes to rivers and streams where water is moving (Stuber *et al*. 1982; de Moor & Bruton 1988). *Micropterus salmoides* prefer soft-bottomed lakes with some submerged vegetation and relatively clear water (Stuber *et al*. 1982). Vegetation cover plays a large role in structuring *M. salmoides* populations, to the extent that relative abundance increases with increased submerged vegetation cover (Paukert & Willis 2004).

Figure 1.3. Mature Micropterus salmoides sampled from the Wilderness Lakes system, Western Cape (source: Armand van der Harst).

Weyl & Hecht (1999) determined the age at maturity of *M. salmoides* in the subtropical Lake Chicamba, Mozambique. The smallest mature male and female fish were 305 mm FL and 290 mm FL respectively with a Lm_{50} at 0.9 years (Weyl & Hecht 1999). In Lake Manyame, Zimbabwe, *M. salmoides* were found to be reasonably long-lived and fast growing, with a lifespan exceeding nine years (Beamish *et al*. 2005). Growth was rapid with males and females attaining 50% of their maximum length within 1.1 and 1.7 years respectively (Beamish *et al*. 2005). A study by Marinelli *et al*. (2007) in Lake Bracciano, Italy found that the minimum size at which fish were mature was 162 mm and 174 mm for the females and the males respectively, both at age zero. The Lm_{50} was reached at 190 mm SL (females) and 200 mm SL (males) at age one. The life span of *M. salmoides* varies from system to system. Fish reach an age of thirteen years in Wriggleswade dam, Eastern Cape (Taylor & Weyl 2011; G. Taylor, Rhodes University, pers. comm.) whereas in Lake Naivasha, Kenya, fish reach a maximum age of four (Beamish *et al*. 2005). This is a guarding species with males constructing nests where females lay their eggs (de Moor & Bruton 1988). Spawning occurs once a season, with limited parental care (de Moor 1996).

The effects of *M. salmoides* as an invader have been well documented as it is an actively hunting piscivore (de Moor & Bruton 1988). A study by Weyl & Lewis (2006) illustrated some of the effects of *M. salmoides* as an invasive species, where remains of the estuarine fish *Monodactylus falciformis* and two Mugilid species, *Mugil cephalus* and *Myxus capensis* were found. The remains of these species were indicative of juvenile fish which migrate into estuaries from the sea. All three of these species are relatively common in the Wilderness Lakes system. While previous records of predation on freshwater species is well documented, predation on estuarine species has been limited to the small, shoaling species *Atherina breviceps* and *Gilchristella aestuaria* (de Moor & Bruton 1988). This is a first record for predation on these larger estuarine species.

Micropterus salmoides has been introduced into many countries worldwide, mainly for recreational fishing (Beamish *et al*. 2005; Rodriguez-sánchez *et al.* 2009). *Micropterus salmoides* were first introduced into the South African Cape coast in 1928, when it colonised rivers extremely rapidly. Literature reviews indicate that within ten years it had established a breeding population in at least five major catchment regions (de Moor 1996). In the Clanwilliam Olifant River, a single stocking of 30 fish was sufficient for the successful establishment of breeding populations in virtually the whole of the lower Olifants River (de Moor 1996).

Cyprinus carpio

Cyprinus carpio are native to eastern Europe and central Asia (de Moor & Bruton 1988; de Moor 1996; Khan *et al*. 2003; Koehn 2004) (**Fig. 1.4**). They were originally bred as ornamental and aquaculture species which resulted in wide-spread translocation around the world (Koehn 2004). Translocation of *C. carpio* by anglers is a major source of invasion into new catchments both in South Africa (de Moor & Bruton 1988) and elsewhere (Koehn 2004). *Cyprinus carpio* were introduced into South Africa in the 18th century as an ornamental fish but were subsequently introduced to water systems throughout South Africa and have become a popular angling species (de Moor & Bruton 1988). They have wide environmental tolerances which has allowed them to become successful freshwater invaders in parts of Europe, Asia, Africa, north, central and south America and Australia (Crivelli 1981; Koehn 2004; Zambrano *et al*. 2006; Skelton & Weyl 2011). In many invaded areas, risk assessment was not undertaken to determine the invasion impacts of *C. carpio* prior to their introduction (Koehn 2004). Their ability to adapt to new environments has been a major factor facilitating their success on a global scale.

Figure 1.4. Adult *Cyprinus carpio* sampled from the Wilderness Lakes System, Western Cape, South Africa.

Cyprinus carpio are essentially freshwater fish and cannot tolerate salinities in excess of 14 ‰ (Koehn 2004). *Cyprinus carpio* are believed to invade species-poor, abiotically harsh environments (de Moor 1996). Low fish species diversity in water systems such as the Vaal-Orange system seem to be prime examples of *C. carpio* invasive potential (de Moor 1996). Where they invade, populations can achieve biomasses as high as 3144 $kg.ha^{-1}$ and densities up to 1000 $individuals.ha^{-1}$.

Cyprinus carpio mature at an early age with fish reaching maturity in tropical climates between one and two years and in temperate climates between three and five years (or 355 -

430 mm SL), with males maturing earlier than females (Smith & Walker 2004; Koehn 2004; Oyugi *et al.* 2011 Winker *et al*. 2011). Optimal spawning temperature is between 18 and 24°C at freshwater inflow areas (Nathanael & Edirisinghe 2001). Females deposit the adhesive eggs on submerged aquatic vegetation where they hatch within three to six days (de Moor & Bruton 1988).

Cyprinus carpio are omnivorous fish that feed on the benthos, consuming chironomids, tubificids, larger zooplankton and zooperiphyton (García-Berthou 2001; Parkos III *et al*. 2003) and detritus. Michel & Oberdorff (1995) reported that chironomids and molluscs generally are the most important animal prey of *C. carpio*.

The invasive potential of *C. carpio* is well known (Smith & Walker 2004). In south-eastern Australia, *C. carpio* has spread rapidly since the introduction of the 'Boolara strain' in the 1960s (Stuart & Jones 2006). Feral *C. carpio* are estimated to comprise the largest fish biomass and be the most numerous large fish species in most coastal and inland rivers and across the Murray-Darling Basin, which is Australia's largest river catchment (Brown *et al*. 2005; Stuart & Jones 2006). *Cyprinus carpio* is now found in many of the foremost fresh waters of Victoria and New South Wales and each year there are colonisations of *C. carpio* into new areas and further incursion into large interior arid catchments. It continues to expand its range in Victoria and South Australia (Koehn 2004; Brown *et al*. 2005; Stuart & Jones 2006).

In Africa the spread of *C. carpio* is also a threat to native systems. In Lake Naivasha, Kenya, the first record of *C. carpio* was March 2002, and initially this species made up a minor component of the commercial fishery (Britton *et al*. 2007). Within two years there had been a major shift in catch composition from tilapia to *C. carpio*, with *C. carpio* comprising over 90% of the total catch (Britton *et al.* 2007).

Rationale for this study

Estuaries are important in promoting and conserving marine fish species and are host to a wide range of native and endemic fish species. The introduction of alien species into freshwater and estuarine systems is the second leading cause of biodiversity loss after habitat loss (García-Berthou *et al*. 2005). The impact of these species on freshwater systems is well documented, and includes competition, altering of invertebrate communities, predation, habitat alteration and cascade effects in aquatic communities (de Moor & Bruton 1988; Gratwicke & Marshall 2001; Khan *et al*. 2003; Gratwicke & Nhiwatiwa 2003; Ling 2004; Kumar & Hwang 2005; Canonico *et al*. 2005). The impact on estuarine fish communities is unknown. This is of particular concern because the presence of alien invasive fish has been identified as a major factor affecting the biodiversity of South African freshwater fishes (Tweddle *et al.* 2009).

To make predictions useful for the management and control of alien species, scientists need to understand the nature of invaders, the invasion process, and the impact of invasion on native species (Moyle & Marchetti 2006). Understanding introduction pathways, distribution, abundance and the establishment ability of these invasive fish species in the system is therefore an essential component to conservation planning and management of this RAMSAR site.

Aims and objectives

The overall aim of this project was to determine the fish fauna present in the Wilderness Lakes system and to contribute to the understanding of the dynamics of alien fishes in the system in order to contribute to an effective alien invasive management strategy for South African National Parks (SANParks).

Focusing on the Wilderness Lakes system, the objectives of the present study were to:

- Assess the spatial distribution and relative abundance of native and alien fishes.
- Determine the state of the alien fish invasions by assessing which species have established.
- Investigate biotic and abiotic factors that may have facilitated or inhibited establishment of alien fishes.
- Assess whether the relative abundance and diversity of native fish has changed since the fish fauna of the Wilderness Lakes system was last studied in 1983 – 1985 by Hall (1985).

Thesis structure

After the general introduction (Chapter 1) and a general methods chapter (Chapter 2) which outlines the sampling techniques used in subsequent chapters, there are four focal chapters. The first (Chapter 3) assesses the overall abundance and distribution of native and alien fish fauna in the Wilderness Lakes system. Chapter 4 focuses on the abundance, distribution and establishment success of the alien species currently present in the system. Following this is an assessment of what environmental factors have either facilitated or inhibited the establishment of alien species in the Wilderness Lakes system (Chapter 5). The last focal chapter (Chapter 6) assesses whether there are any indications of impact on native biota by comparing species distributions and abundance between two studies. The thesis ends with a general synthesis and potential management of the alien species and conclusion (Chapter 7).

CHAPTER 2

GENERAL METHODS

Study area

The Wilderness Lakes system is located between the towns of George (16 km) and Knysna (40 km) and adjoins the Indian Ocean on the southern Cape coast. It covers an area of approximately 13 km^2 (Randall 1990) and comprises the Touw Estuary, three coastal lakes and three interconnecting channels (**Fig. 2.1**). The Touw Estuary is connected to Island Lake via the Serpentine channel. Island Lake and Langvlei are connected via the Island Lake-Langvlei channel and Langvlei and Rondevlei are connected via the Rondevlei-Langvlei channel. The system is situated on the South coast of South Africa and falls within the warm temperate bioregion (Whitfield 1998; James & Harrison 2010). Touw estuary is classified as a temporary open/closed estuary (Whitfield 1998; James & Harrison 2008) where the mouth is open to the ocean for varying periods of time.

The catchment area of the Wilderness Lakes contains three rivers, the Touw River (96.2 km^2), the Duiwe River (42.1 km^2) and the Langvlei Spruit (8.2 km^2) (Hall 1985). The hydrology of the lakes is dominated by the flooding of the Touw and Duiwe Rivers when the mouth of the estuary is closed and by tidal influence when the mouth is open. The upper lakes are filled by reverse flow, where water from the Touw and to a lesser extent the Duiwe River flows back via the lower lakes and channels. This is especially the case when the mouth of the estuary is closed, and the process serves to reduce the effects of floods in the Touw River estuary. The height of sand bar at the mouth of the estuary has to be artificially maintained to prevent flooding in the catchment (Russell *et al*. 2010).

Rondevlei is the smallest lake with a surface area of 1.43 km^2 and a maximum depth of 6m. Langvlei is the shallowest lake with a maximum depth of 4 m, but it has the largest surface area of 2.16 km^2. Island Lake has a maximum depth of 6.5 m and is the deepest in the system with a surface area of 1.5 km^2. The Touw Estuary varies in depth from between 1.5 m and 3 m deep (Hall *et al*. 1987; Russell 2003).

Conservation

The national conservation importance of the Wilderness Lakes system has been assessed in several studies, with the Wilderness Lakes systems ranking 9th of South Africa's estuaries in terms of water bird conservation (Turpie 1995), and 24th in terms of overall conservation importance, which includes criteria such as size, diversity of habitat, zonal rarity and biodiversity (Turpie *et al*. 2002). The international conservation importance of the Touw system was highlighted when on 28 June 1991 the majority of the wetland system (Serpentine channel, Island Lake, Langvlei, Rondevlei and interconnecting channels) was declared a RAMSAR site. In a prioritisation of South African estuaries based on their potential importance to estuarine associated fish species, the Wilderness Lakes system rated 11 out of 248 estuaries (Maree *et al*. 2003).

Current and historical use of the Wilderness Lakes system

Rondevlei and Langvlei are closed to any utilisation while Island Lake and Touw Estuary are open to the public for recreational fishing and boating. Fishing with hook and line is permitted in Island Lake and portions of the Touw Estuary and Serpentine channel. Fishing is undertaken primarily by local residents and no charge is levied. The quantity of fish removed is unknown though is unlikely to be substantial. Department of Agriculture, Forestry and Fisheries regulations with respect to size and bag limits are enforced.

Most forms of boating are permitted on Island Lake, with canoeing permitted on the Serpentine channel and the Touw Estuary. Boating is undertaken by both local residents and seasonal visitors, with higher utilization possibly by the latter group.

Swimming and picnicking is permitted in demarcated areas at Island Lake and along the Touw Estuary. Hiking/walking trails have been laid out in the Touw River/Serpentine area on the floodplain, in the forest above Island Lake and in the fynbos area east of Rondevlei. Bird hides at Rondevlei and Langvlei are highly utilised by the public, and there are numerous points on the water between the Touw River estuary and Langvlei accessible for birding.

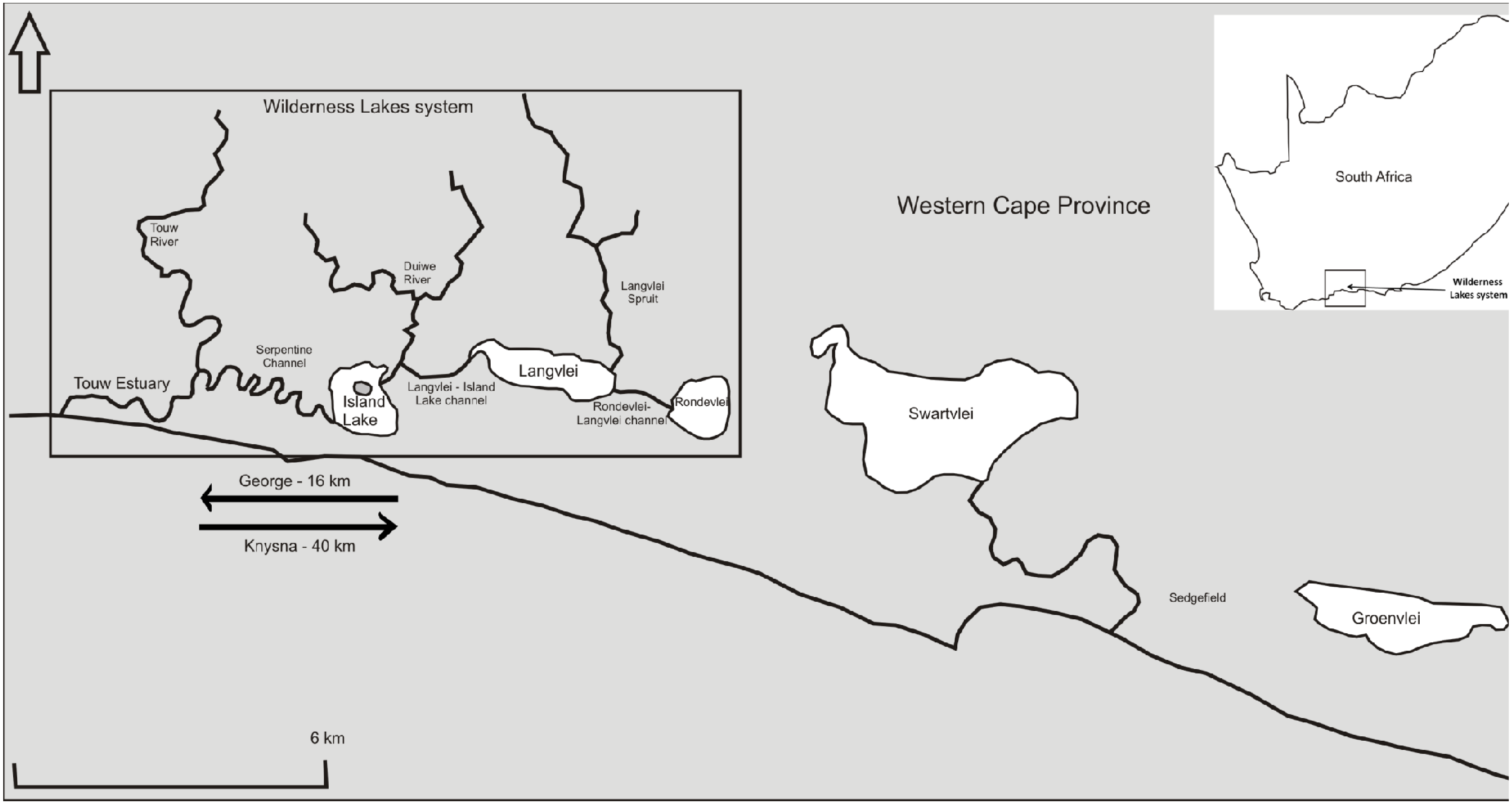


Figure 2.1. Wilderness Lakes system situated in the Garden Route National Park (33°50' - 34°30' S; 22°33' - 22°50' E) between George and Knysna in the Western Cape, South Africa.

Enforcement patrols are conducted SANParks on a regular basis and illegal gill nets are collected frequently in the Wilderness Lakes system. The nets are made of packaging plastic with polystyrene floats (**Fig. 2.2**).

Figure 2.2. Example of illegal gill net made from packaging plastic with polystyrene floats found in the Wilderness Lakes system, Western Cape.

CLIMATIC CONDITIONS

Rainfall and temperature

The Wilderness Lakes system occurs in the relatively small perennial rainfall zone of South Africa (Russell *et al*. 2010). Annual rainfall is between 600 and 700 mm with little seasonal variation, but slight peaks do occur from January to March, and again from August to November. Mean rainfall in the upper catchments is 900 - 1000 mm y^{-1}, and mean monthly rainfall varies between 11 and 244 mm (Russell *et al*. 2010).

This study was initiated during a drought period experienced throughout the Western Cape. Between the months of March 2010 and May 2010 (during the drought period) the water levels in the lakes, channels and estuary were very low (**Fig. 2.3**). Rondevlei, Rondevlei-Langvlei channel and Langvlei were considered to be separate water bodies with no movement of water or fish biota between them. During this time there was a substantial die back of submerged aquatic vegetation assumed to be due to the higher salinity levels. This was seen especially in Langvlei (I. Russell, SANParks aquatic scientist, pers. comm.).

Figure 2.3. Rondevlei, Wilderness Lakes system in April 2010 during the drought period when this study was initiated.

After May 2010 higher rainfall patterns resumed which increased the water levels by approximately 50 cm (**Fig. 2.4**). This increase in water levels caused Rondevlei and Langvlei to join again via the Rondevlei-Langvlei channel and the remainder of the study was conducted under increasing water levels and vegetative growth.

Figure 2.4. Rondevlei, Wilderness Lakes system in October 2010 when water levels had risen after a period of high rainfall.

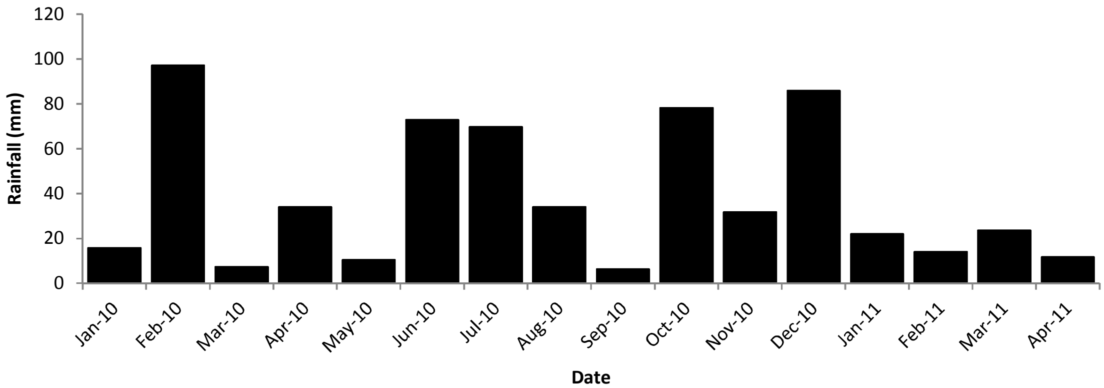


Figure 2. 5. Total monthly rainfall for the Wilderness Lakes system for the period January 2010 to April 2011 (SANParks unpublished data).

The Wilderness Lakes region experiences relatively mild summer and winter temperatures. The minimum and maximum temperatures in the Wilderness Lakes were -0.7(August 2010) and 37.6^{o}C (March 2011) respectively (**Fig. 2.6**). South-west winds prevail throughout the year though warm north and north-east winds are fairly common (Russell *et al*. 2010).

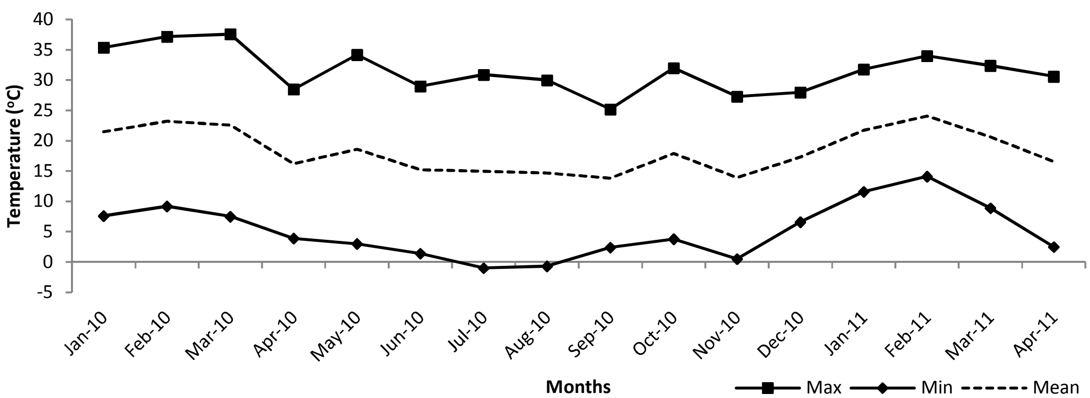


Figure 2.6. The maximum, mean and minimum air temperatures (^{o}C) for the Wilderness Lakes system between the months of January 2010 and April 2011 (SANParks, unpublished data).

THE ESTUARINE ENVIRONMENT

As part of long term monitoring by the South African National Parks (SANParks), physico-chemical parameters are measured quarterly. Between 1991 and 1997, water temperature (^{o}C) and salinity (g/kg) were measured using a YSI Model S-C-T meter while dissolved oxygen (mg/l) was measured using a SG 867 digital O_2 meter and pH with a Knick 751 pH meter. In recent years, newer instruments (YSI 550A Dissolved Oxygen; YSI Model 60 Handheld pH and Temperature; YSI Model 30 Handheld Salinity, Conductivity and Temperature instrument) have been used. Measurements are taken 30 cm below the water surface at 5 established sites in Rondevlei, Langvlei and Island Lakes and nine sites in the Touw Estuary. These data are kept in a database by SANParks.

Mouth state

The Touw Estuary mouth state is recorded daily and is partially managed by SANParks. When water levels in the estuary reach between 2.1 m – 2.4 m above mean sea level, the mouth is opened manually by means of heavy earth moving equipment as recommended by the Council for Scientific and Industrial Research (CSIR) (1981, 1982). This is done to reduce flooding in the catchment. In 2008 and 2009 the mouth state was open for both years whereas in 2010 the mouth state was only open for 26 days.

Table 2.1. State of the Touw Estuary mouth openings in days (percentage) recorded between 2000 and 2010

Year	Days Closed (%)	Days Open (%)
2000	216 (59.0)	150 (41.0)
2001	156 (42.7)	209 (57.3)
2002	282 (77.3)	83 (22.7)
2003	216 (59.2)	149 (40.8)
2004	109 (29.8)	257 (70.2)
2005	200 (54.8)	165 (45.2)
2006	138 (37.8)	227 (62.2)
2007	144 (39.5)	221 (60.5)
2008	0 (0.0)	366 (100.0)
2009	0 (0.0)	365 (100.0)
2010	339 (92.9)	26 (7.1)

Salinity

The Wilderness Lakes system often exhibits a reverse salinity gradient, with the salinity increasing higher up the system towards Rondevlei. Allanson & Whitfield (1983) measured the salinity in the system as: Wilderness Lagoon: 14 – 25 ‰, Island Lake: 6 – 10 ‰, Langvlei: 10 - 13 ‰ and Rondevlei: 12 – 16 ‰. Hall (1985) also recorded a reverse salinity gradient however the salinities were lower, with 4 ‰ in Island Lake, 5 ‰ in Langvlei and 9 ‰ in

Rondevlei. The salinity in the Touw Estuary is dependent on the state of tide and mouth state. Generally the salinity in the lakes varies between 2 and 16 ‰, with the highest values in Rondevlei, whereas in the estuary it varies between 0 and 25 ‰ (Russell 1999b). The salinity profile for each of the lakes and Touw Estuary measured quarterly since 1991 is shown in **Figure 2.7**.

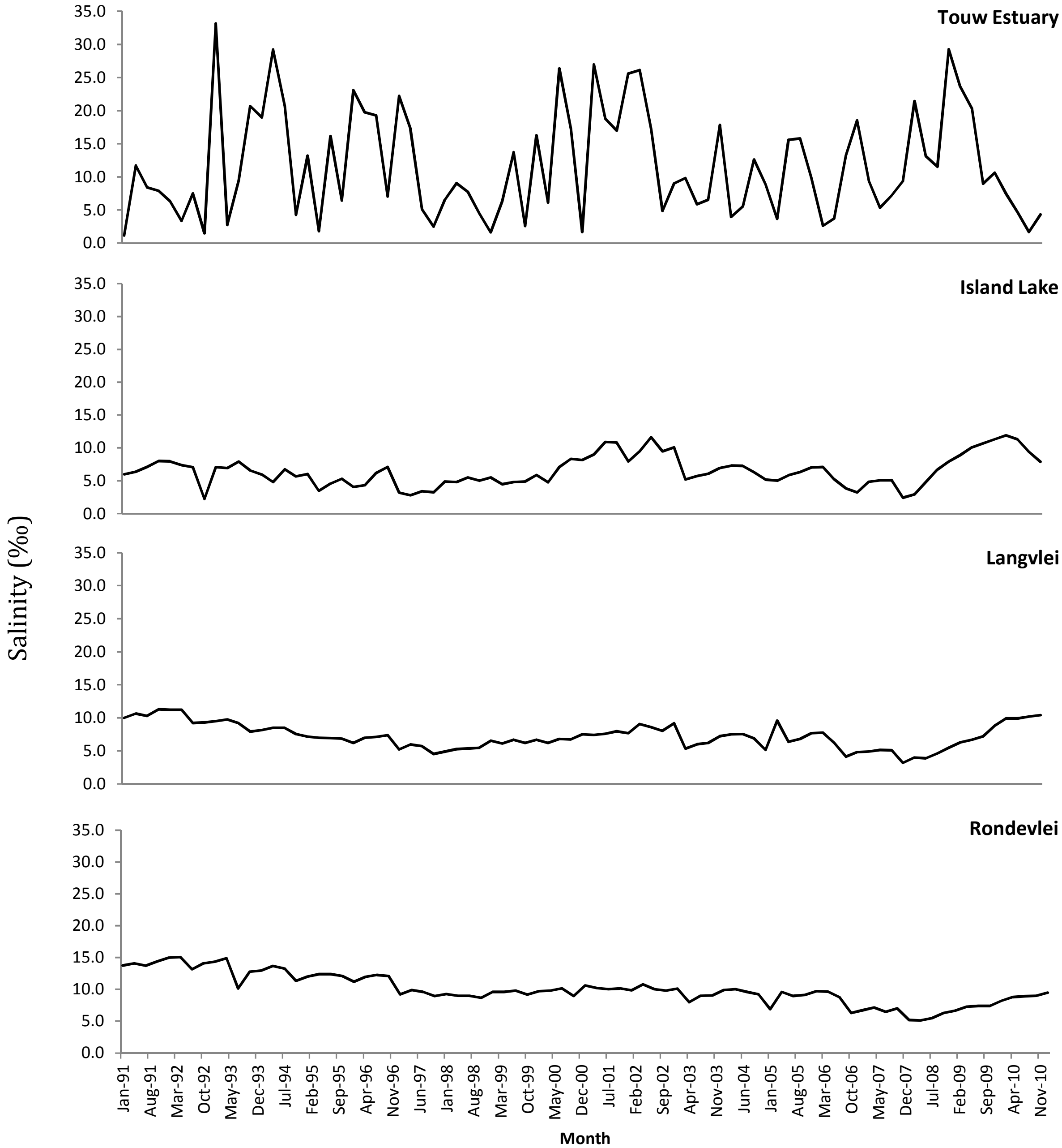


Figure 2.7. Salinity profile of each of the sampling areas in the Wilderness Lakes system for the period January 1991 to November 2010.

Temperature

The water temperature in the Wilderness Lakes system fluctuates seasonally, with temperature reaching an average maximum of 27.0°C in summer and an average minimum of 10.7°C in winter. The recorded temperature of each of the lakes and the Touw Estuary measured quarterly since January 1991 is shown in **Figure 2.8**.

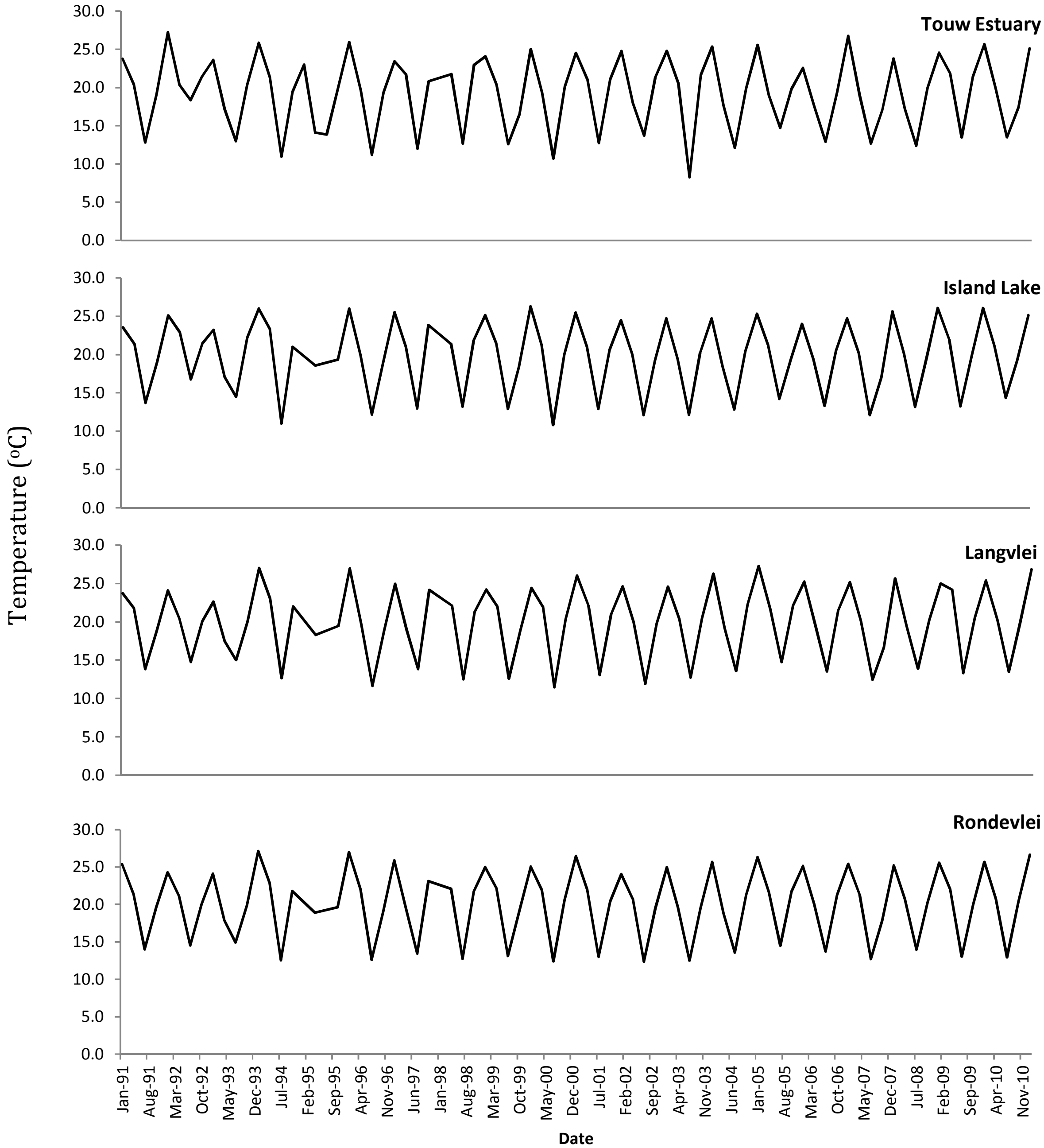


Figure 2.8. Temperature (°C) profile for each of the water bodies of the Wilderness Lakes system for the period January 1991 to November 2010.

Vegetation

The Wilderness Lakes are bordered by a margin of emergent aquatic plants made up predominantly of *Phragmites australis* (common reed) and to a lesser extent the *Juncus kraussii* (dune slack rush), *Schoenoplectus scirpoides* (Biesiegoed) and *Typha capensis* (bulrush) (Russell 2003). Submerged aquatic macrophytes occur in each of the lakes and are made up mostly of stands of *Potamogeton pectinatus* (fennel pondweed), Charophyta and filamentous algae (*Cladophora* spp.). *Zostera capensis* (eelgrass) occurs sporadically in the Touw Estuary (Howard-Williams & Liptrot 1980; Weisser & Howard-Williams 1982; Allanson & Whitfield 1983). Areas deeper than 3 m are relatively sparse vegetatively as submerged macrophytes do not generally occur in water deeper than 3 m (Russell 2003).

FIELD SAMPLING AND ANALYSIS

The Lakes system was sampled using a seasonally and geographically stratified sampling design for a period of 13 months (April 2010 – May 2011). The year was divided into four seasonal sampling periods which coincided with the seasons used during a previous study by Hall (1985): June – August (winter), September – November (spring) December – February (summer) and March – May (autumn). Four different net types were used to sample as great a size range of fish over as wide of a range of habitats as possible.

Fyke nets

Each season 12 double-ended fyke nets (8 m guiding net, first-ring diameter of 55 cm, 10 mm mesh size at the cod end) were set in each of the lakes. These nets were separated into four sites with three nets set in approximately the same location each season. The Rondevlei-Langvlei channel and the Serpentine had three nets set each and the Touw estuary had two sites of three nets set each season (**Fig. 2.9**).

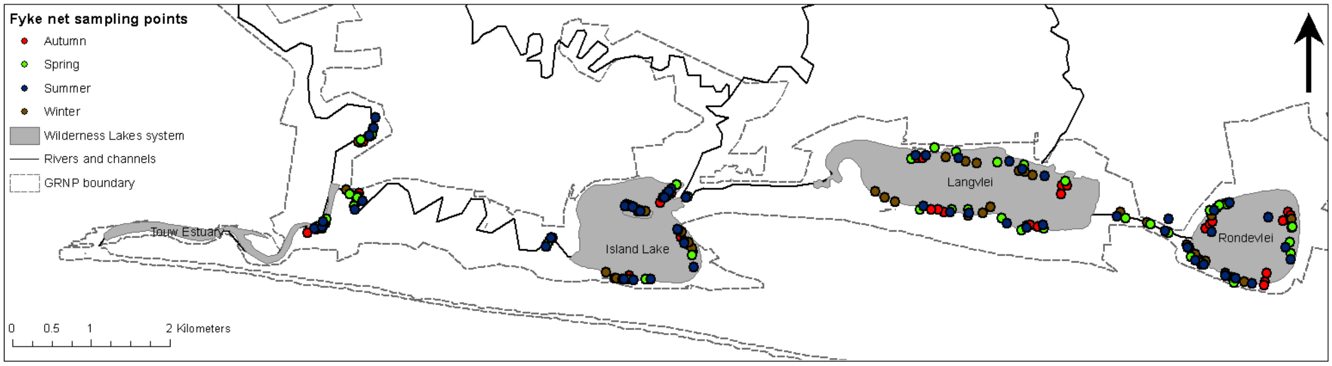


Figure 2.9. Seasonal fyke net sampling points in the Touw Estuary, Serpentine channel, Island Lake, Langvlei, Rondevlei-Langvlei channel and Rondevlei. Colours represent different seasons.

Fyke nets are considered a passive gear type and were set in water approximately 1 - 1.5 m deep. Many of these areas were on the verge between sandy substrates and beds of *Potomogeton pectinatus*. All fyke nets were fitted with an "otter guard" comprising plastic mesh with openings no larger than 10 cm x 10 cm (**Fig. 2.10**) to prevent Cape clawless otters, *Aonyx capensis*, from entering the nets. Although the use of these otter guards influenced the maximum size of fish that could enter the nets, their use was considered critical to avoid bycatch. All fyke nets were set in the evening (between 16:00 and 18:00) and lifted the next morning (between 06:00 and 08:00) with an average soak time of 16 hours. All the fyke nets were set and collected in the same sequence as to minimize variance in soak time.

Figure 2.10. Example of the otter guards which were made of plastic mesh with openings no larger than 10 cm x 10 cm fitted to the openings of all the fyke nets.

Gill nets

A set of five multi-meshed gill nets each measuring 35 m x 2.75 m with stretch meshes of 35, 45, 57, 73, 93, 118 and 150 mm (5 m per mesh size) were set for two consecutive nights per season in different positions in each of the lakes (**Fig. 2.11**). In the upper and lower reaches of the Touw estuary (two nets), the Duiwe River (one net) and the lower serpentine (one net) nets were only set once per season (**Fig. 2.11**). Gill nets were deployed at sunset and had an average soak time of two hours; the nets were manned at all times to reduce mortality and incidental bycatch. Several mesh sizes were used to eliminate size and species selectivity which could cause potential problems when investigating population and size frequency analyses (Prchalova *et al*. 2009).

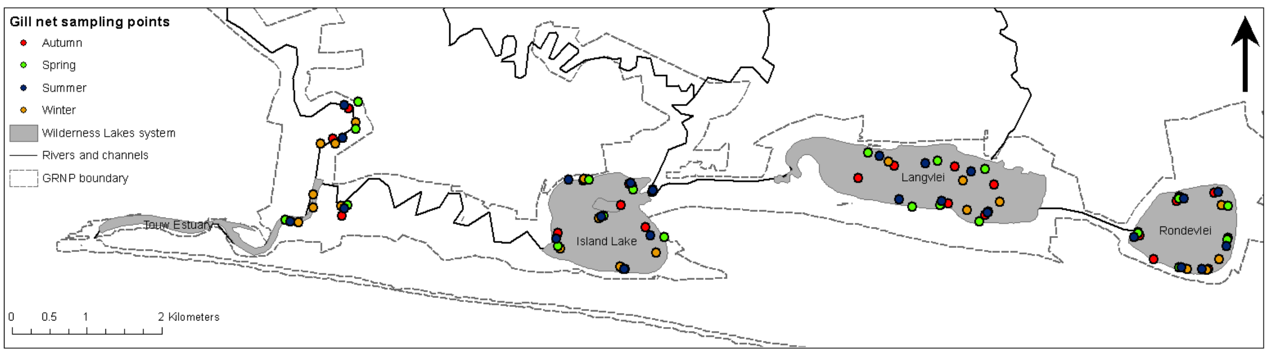


Figure 2.11. Seasonal gill net sampling points in the Touw Estuary, Serpentine channel, Island Lake, Duiwe River, Langvlei and Rondevlei. Colours represent different seasons.

Seine nets

Two beach seine nets were used seasonally. A large seine net (30 m long × 2 m deep with a bag, 12 mm stretched mesh) was laid from a boat approximately 30 m offshore (**Fig. 2.12**). The large seine net was used to target fish both from the littoral and from the pelagic zones. Due to progressively increasing water levels in the lakes throughout the study, hauling up became progressively challenging. Initially three pulls at three sites were completed in each of the lakes and the Touw Estuary (**Fig 2.13**). During spring and summer a filamentous epiphytic algae (*Enteromorpha* spp.) increased in abundance in the littoral regions of the lakes which clogged the large seine net making it exceptionally difficult to haul in. As water levels increased and the rate of algal growth increased, the number of suitable sites decreased which resulted in lower numbers of pulls that could be successfully completed. In the final, summer season it was only possible to complete the Touw Estuary seine netting.

Figure 2.12. Examples of the 30 m seine net sampling with the net being deployed and hauled in.

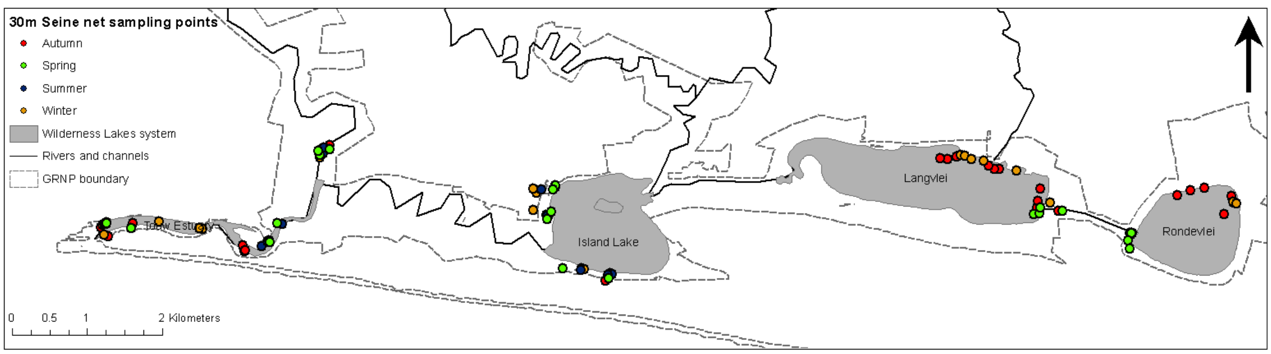


Figure 2.13. Seasonal 30 m seine net sampling points in the Touw Estuary, Island Lake, Langvlei, Rondevlei-Langvlei channel and Rondevlei. Colours represent different seasons.

A small seine net (10 m × 2 m with 8 mm mesh) was laid by hand and pulled onto sand banks from water not more than 1 m deep. The small seine net was used to sample shallow littoral habitats in the lakes and in the channel. This seine net was pulled ten times in each of the lakes and three times in the Rondevlei-Langvlei channel during the autumn to spring surveys (**Fig 2.14**). The maximum number of seine net pulls in summer was influenced by algal growth and water levels. In Rondevlei and Langvlei, seven pulls were made but in Island Lake the full ten pulls were possible. It was not possible to use a seine net in the Serpentine channel as there were no suitable beaches on which to haul the net.

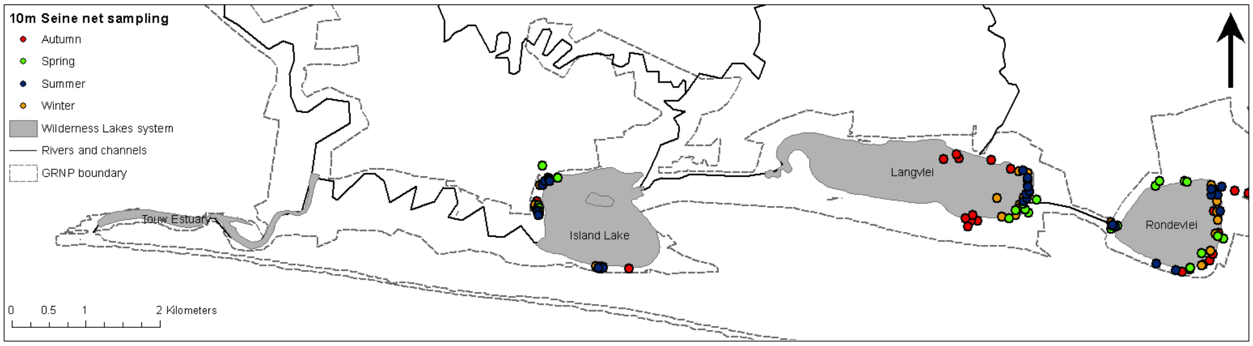


Figure 2.14. Seasonal 10 m seine net sampling points in Island Lake, Langvlei, Rondevlei-Langvlei channel and Rondevlei. Colours represent different seasons.

Sample treatment

Fish caught were identified to the lowest taxonomic level possible, counted and measured to the nearest millimetre fork length (FL) or total length (TL) depending on species, and released alive where possible. Sub-sampling was used in cases where the catch comprised large numbers of small fish (**Fig. 2.15**), from which total abundances were estimated. Sub-sampling involved measuring out two buckets or handfuls of fish to be measured and each fish species in the subsample was identified, counted and measured. The remainder of the fish were released and the number of buckets or handfuls released were counted. From this count, the total catches and numbers of each species was calculated. Identified species were grouped into their estuarine-dependence category as defined by Whitfield (1994a).

Figure 2.15. Example of a 30 m seine net catch with very high numbers of the shoaling pelagic species, *Gilchristella aestuaria* and *Atherina breviceps*, where sub-sampling was required.

Cyprinus carpio sampling

Cyprinus carpio were first sampled in the Wilderness Lakes in 2009 using fyke nets. In this study, low numbers of *C. carpio* were sampled in the 30m seine net during the 2010 surveys.

An extra fyke netting set was done throughout the system in autumn 2011 to give a three year autumn fyke netting analysis. The data from these gear types were used to show the progression of lengths over a two year period. *C. carpio* were sampled in Island Lake, Langvlei, Touw Estuary and the Serpentine channel and Rondevlei in the fyke nets and seine nets.

Angler records (including date, weight and photograph of each fish) for the period October 2010 to March 2011 for *C. carpio* caught in Island Lake were obtained from Heiko Feddersen and incorporated into the dataset (**Fig. 2.16**).

Figure 2.16. Example of *Cyprinus carpio* after it was caught by an angler in Island Lake. This record and others collected were added to the dataset (source: Heiko Feddersen).

Gambusia affinis sampling

While some *G. affinis* were sampled in the 30 m and 10 m seine nets, this was not adequate to determine the distribution of the species. In May 2011, a once-off scoop net sampling survey was conducted on the system to determine the abundance and distribution throughout the Wilderness Lakes system. In each of the lakes, interconnecting channels and the Touw Estuary, a selected number of sites was sampled with five scoops per site (**Fig. 2.17**). The sites used were selected to correspond to those in a study done by (Sloterdijk *et al*. 2011), in which the biology of *G. affinis* in the Wilderness Lakes system and surrounding estuaries was determined.

Where possible, each scoop covered approximately 5 m of shoreline in very shallow water along the littoral vegetation. Length frequency distribution data from the 30 m and 10 m seine nets were used for overall abundance and distribution calculations (Chapter 3) and the length frequency data from the scoop net data.

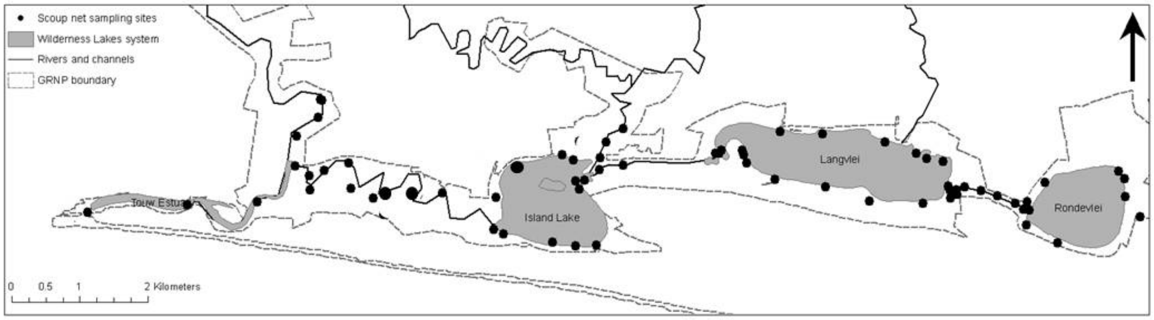


Figure 2.17. The location of the scoop net sampling points during the once-off scoop net sampling during May 2011 in each of the lakes, their interconnecting channels and the Touw Estuary.

Micropterus salmoides **sampling**

A once-off sampling in the Duiwe River was conducted in March 2011, using an electro fisher in the small pools to determine the presence of juvenile *M. salmoides*. At the time of sampling the water levels were low, creating small pools which provided the ideal opportunity to sample the shallow pools (**Fig 2.18, 2.19**). At each sampling site, three-pass electro fishing was conducted using a Samus© 725G backpack electro fisher, attached to a 12 V battery with settings for the electro fisher standardised at a duration of 0.3 milliseconds and a frequency of 80 Hz. Each pass covered the entire length of each pool. After each pass, fish caught on that pass were placed in a bucket with water, and the fisher returned to the pool to conduct the next pass.

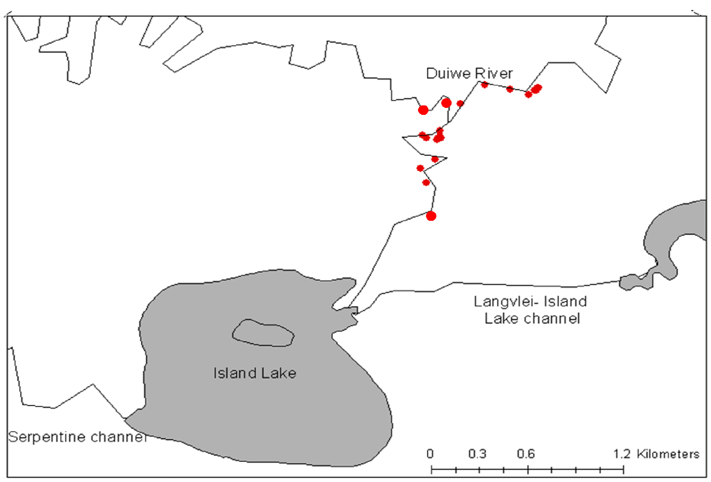


Figure 2.18. The location of the once-off electro fishing sampling sites in the Duiwe River which leads into Island Lake, Wilderness Lakes system.

Figure 2.19. Examples of electro fishing in the small pools along the Duiwe River, Western Cape.

Physico-chemical parameters

Various physico-chemical parameters (water temperature, pH, salinity, conductivity and dissolved oxygen) were measured during sampling trips. Measurements were taken each season at selected sampling sites, approximately 30 cm below the water surface using hand held instruments. A YSI 550A was used to measure dissolved oxygen, YSI Model 60 to measure pH and temperature and a YSI Model 30 to measure salinity, conductivity and temperature. Measurements were taken at fish sampling sites in each of the lakes, interconnecting channels and the Touw Estuary and the mean for each sampling season was used during analyses.

CHAPTER 3

DISTRIBUTION AND ABUNDANCE OF FISHES IN THE WILDERNESS LAKES SYSTEM

* Some of the results presented in this chapter have been published in Olds *et al*. (2011) (**Appendix 4**).

INTRODUCTION

Estuaries are the meeting place of rivers and the sea and are dynamic environments characterised by large fluctuations in environmental conditions (Harrison & Whitfield 2006; James *et al*. 2007; James & Harrison 2010). They are important nursery areas for several species of marine fishes, many of which are exploited in estuaries or in coastal fisheries later in their life cycles (Blaber & Blaber 1980; Wallace *et al*. 1984; James & Harrison 2010). Estuaries play an important role in promoting species richness in South Africa (Wallace *et al*. 1984). Fish that utilise estuaries can be divided into two broad groups according to their spawning locations. A marine spawning group which comprises mainly large species that enter estuaries primarily as juveniles and usually return to sea prior to attaining sexual maturity, and an estuarine group, made up predominantly of small species which have the ability to complete their life-cycles within estuaries (Whitfield 1998). The ichthyofauna composition in estuaries along the south coast of South Africa is fairly well known (Hall *et al*. 1987; Whitfield 1980, 1994b, 1998, 1999; Vorwerk 2000; James & Harrison 2010) and these studies, reviewed by James *et al*. (2007), show that in South African south coast estuaries the fish fauna are dominated by juvenile marine-dependent marine species with a strong component from the families Mugilidae and Sparidae and that native estuarine species are common and abundant in most estuaries (James & Harrison 2010) (**Table 3.1**).

Numerous studies have been conducted on the Wilderness Lakes system over the past 30 years, including hydrology (Allanson & Whitfield 1983), water chemistry (Russell 1999a), submerged macrophytes (Howard-Williams 1980; Weisser & Howard-Williams 1982), zooplankton community structure (Coetzee 1983), avian communities (Boshoff 1991) and fish community composition (Hall *et al*. 1987; Russell 1996; Russell 1999b; Olds *et al*. 2011). Hall *et al*. (1987) assessed the abundance of fishes throughout the system. An assessment by

James & Harrison (2008) was conducted on estuaries along the south coast of South Africa and it included the Touw Estuary. The ichthyofauna were sampled using a 30 m seine net and a fleet of gill nets (45, 75 and 100 mm mesh). Sampling comprised of six seine net hauls and five gill net sets (James & Harrison 2008).

The results from Hall (1985) and James & Harrison (2008) are typical of those in many South African estuaries. The dominant fish fauna was a marine/estuarine migratory component made up primarily of Mugilidae and Sparidae species (Hall 1985; James & Harrison 2008). The dominance of the Mugilid fishes has been recorded in other estuaries such as the Groot Brak estuary, Knysna estuary and Swartvlei estuary, which are all located in the warm-temperate bioregion (James & Harrison 2008). Native estuarine species (*A. breviceps* and *G. aestuaria*) were numerically dominant in all temporary open/closed estuaries. However, a direct comparison with James & Harrison (2008) could not be used to determine the differences in sampling procedure and only the Touw Estuary was sampled.

Hall (1985) and (Russell 1996, 1999b) sampled three alien freshwater fishes from the Wilderness Lakes. These were Mozambique tilapia *O. mossambicus*, mosquitofish *G. affinis* and largemouth bass *M. salmoides*. These species have been associated with a host of negative impacts on aquatic ecosystems. These impacts include competition with native biota (Canonico *et al*. 2005), *G. affinis* and *M. salmoides* alter invertebrate communities (Kumar & Hwang 2005; Pyke 2008; Weyl *et al*. 2010). *Micropterus salmoides* have been recorded preying on native estuarine fish (Weyl & Lewis 2006). *Cyprinus carpio* impact aquatic habitats as a result of their benthic foraging. These impacts include the resuspension of sediments, damaging macrophytes from feeding and decreasing zooplankton abundance (Khan *et al*. 2003; Miller & Crowl 2006; Britton *et al*. 2007) and introduction of fish diseases (de Moor & Bruton 1988).

Effective management of estuaries requires an understanding of relative abundances and species composition. Further, the present state of the fish community needs to be assessed to allow for spatial comparisons across estuaries and temporal assessments of change within an estuary. To monitor future impacts, the present state needs to be defined.

Table 3.1. Fish species commonly associated with temporary open/closed estuaries in the warm-temperate region, characterised by Harrison (2002) grouped by estuarine association (EA) categories described by Whitfield (1998). Reference numbers indicate studies that sampled the species in temporary open/closed estuaries.

Family	Species	EA	Reference
Atherinidae	*Atherina breviceps*	Ib	1;2;3
Clinidae	*Clinus superciliosus*	Ib	2
Clupeidae	*Gilchristella aestuaria*	Ia	1;2;3
Gobiidae	*Caffrogobius nudiceps*	Ib	1;3
	Caffrogobius gilchristi	Ib	2;3
	Glossogobius callidus	Ib	1;3
	Psammogobius knysnaensis	Ib	1;2;3
Syngnathidae	*Syngnathus acus*	Ib	1;3
Carangidae	*Lichia amia*	IIa	1;2;3
Elopidae	*Elops machnata*	IIa	1;2;3
Haemulidae	*Pomadasys commersonnii*	IIa	1;2;3
Monodactylidae	*Monodactylus falciformis*	IIa	1;2;3
Mugilidae	*Liza dumerili*	IIb	1;2;3
	Liza richardsonii	IIc	1;2;3
	Liza tricuspidens	IIb	1;2;3
	Mugil cephalus	IIa	1;2;3
	Valamugil cunnesius	IIa	3
Sciaenidae	*Argyrosomus japonicus*	IIa	1;2;3
Sparidae	*Lithognathus lithognathus*	IIa	1;2;3
	Rhabdosargus holubi	IIa	1;2;3
Ariidae	*Galeichthys feliceps*	IIb	1;2;3
Soleidae	*Heteromycteris capensis*	IIb	1;2;3
	Solea turbynei	IIb	1;2
Sparidae	*Diplodus sargus capensis*	IIc	1
	Sarpa salpa	IIc	1;2
Haemulidae	*Pomadasys olivaceum*	III	1;3
Cichlidae	*Oreochromis mossambicus*	IV	1;2;3
Mugilidae	*Myxus capensis*	Vb	1;2;3

1-James *et al*. 2007, 2- James & Harrison 2008, 3 – Vorwerk 2000

The aims of this chapter were to firstly assess the fish fauna from a qualitative (species composition) and quantitative (relative abundance) perspective in the Wilderness Lakes system and determine if the fish fauna in this system are representative of the species assemblage in other southern Cape estuaries. Secondly, to determine if the fish fauna along the Wilderness Lakes system are typical of the gradient (where freshwater species become less abundant and euryhaline marine species become more abundant up the system where salinity increases) from marine to freshwater shown in other South African studies.

In order to determine this, two hypotheses were tested. These were:

1) The fish fauna in the Wilderness Lakes system is representative of the species assemblage in other southern Cape estuaries

2) The fish fauna along the Wilderness Lakes system is typical of the gradient from marine to freshwater shown in other South African studies

METHODS AND MATERIALS

General

Fish were sampled using fyke nets, gill nets, 30 m and 10 m seine nets as described in Chapter 2. All fish caught were identified to the lowest taxonomic level possible, measured to the nearest millimetre fork length (FL) or total length (TL) depending on species, and released alive where possible. Sub-sampling was used in cases where the catch comprised large numbers of small fish, from which total abundances were estimated. Positively identified fish were grouped into their associated estuarine-dependence category as defined by Whitfield (1994a) (**Table 3.2**). Due to the complexity of the system and the species sampled, analyses were undertaken using estuarine association categories.

Table 3.2. The five major categories of fishes using South African estuaries, as described by Whitfield (1994a, 1998).

Categories	Description
Ia	Resident estuarine species only spawning in estuaries
Ib	Resident species spawning in estuaries, freshwater and marine environments
IIa	Euryhaline marine species which breed at sea but juveniles dependant on estuaries as nursery areas
IIb	Euryhaline marine species which breed at sea with juveniles occurring mainly in estuaries, but also found at sea
IIc	Euryhaline marine species which breed at sea with juveniles occurring in estuaries but are usually more abundant at sea
III	Marine species not dependant on estuaries
IV	Freshwater species
Va	Obligate catadromous species
Vb	Facultative catadromous species

Catch per unit effort (CPUE)

The overall community structure was determined using catch per unit effort (CPUE) where all catches were expressed as CPUE using the equation:

$$CPUE_i = C_i / E$$

where C_i is the catch of any species i and E is the effort expended to obtain i. For seine nets this was standardised as per haul while fyke effort was standardised to an overnight net deployment session and for gill nets which were standardised to a two hour net deployment session.

Catch was expressed as the number of fish caught and as mass of fish caught. Mass was estimated by converting length data using the length/weight relationship for each species (**Appendix 1**).

Catch composition

The complexity of the fish community on a species level necessitated the use of assessment methods that incorporate abundance, biomass and number of fish sampled. The index of relative importance (IRI) has been applied by several authors (Hart *et al*. 2002; Liao *et al.* 2001; Espinoza & Wehrtmann 2008), primarily in diet studies, but the index is commonly used in abundance studies (Pinkas *et al*. 1971; Caddy & Sharp 1986; Kolding 1989; Kumara & Amarasinghe 2008). For each gear type, IRI was calculated as:

$$IRI = (\%N + \%M) \times (\%FO)$$

where the frequency of occurrence (%FO) is the proportion of the samples (hauls, nets set) that a species was sampled in expressed as a % of the total nets set or hauls, %N is the relative number of individual fish per species expressed as a proportion of the total catch of all fishes, and %M is the relative gravimetric contribution of any species to the total mass in each gear type.

Statistical Analysis

A von Bertalanffy function was fitted to the number of species sampled and sampling effort to determine whether the fish community in the Wilderness Lakes system was sampled adequately in each of the gear types. The equation is shown below:

$$SE_t = SE_\infty \left(1 - e^{-k\,(t - t_0)}\right)$$

where, SE_t is the sampling effort, SE_∞ is the maximum number of species sampled in the sampling gear in that sampling area, k is the slope of the curve and t_0 is the number of species sampled at a sampling effort of 0.

Assumptions for normality and homogeneity of the variance were not met by the CPUE data by fish number or weight. Therefore the non-parametric, Kruskal-Wallis (H) ANOVA was used to compare the CPUE between four sampling seasons and between each of the sampling areas.

A chi square contingency table was used to assess whether the proportion of fish in each estuarine association category (Whitfield 1998) was dependant on the locality sampled. For fyke nets, gill nets and 30 m seine nets a four (sampling areas) × four (estuarine association category) contingency table was used and a three (sampling areas) × four (estuarine association category) contingency table was used for the 10 m seine net.

The species composition (grouped into estuarine association categories) in the Wilderness Lakes system was compared with four temporary open/closed estuaries located in the warm-temperate region (Harrison 2002) in the Western and Eastern Cape, using a five (temporary open/closed estuaries) × five (estuarine association categories) chi square contingency table.

RESULTS

A) Fyke nets

Seasonal CPUE

Seasonal fyke net CPUE, expressed as number of fish per net, is summarised in **Figure 3.1**. CPUE did not differ significantly by season in the Touw Estuary, Serpentine channel and in the Rondevlei-Langvlei channel (Kruskal-Wallis ANOVA; $p > 0.05$). In Island Lake, CPUE was significantly higher in autumn with CPUE increasing from 0.4 ± 0.1 fish.net.night^{-1} in spring to 78 ± 48 fish.net.night^{-1} in autumn. In Langvlei and Rondevlei, autumn and summer fyke net CPUE was significantly higher than winter and spring CPUE (**Fig. 3.1**).

Annualised CPUE

CPUE expressed as number of fish per net set differed significantly between sample sites (Kruskal-Wallis ANOVA; $H = 24.19$, $p < 0.05$). CPUE was significantly lower in Island Lake than in all other areas (**Fig. 3.2**). CPUE expressed as weight of fish per net set differed significantly between sample sights (Kruskal-Wallis ANOVA; $H = 25.7$, $p < 0.05$). CPUE was significantly lower in Island Lake and Rondevlei than in all other areas.

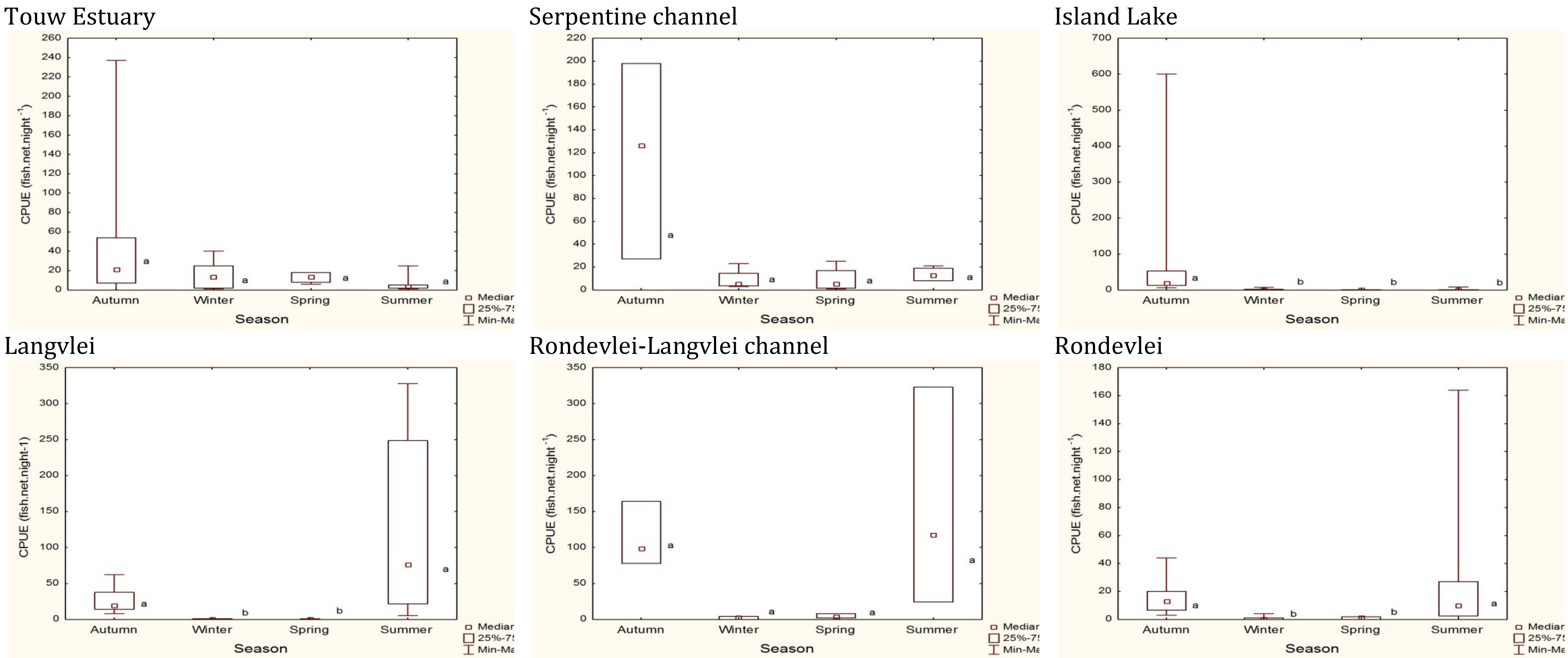
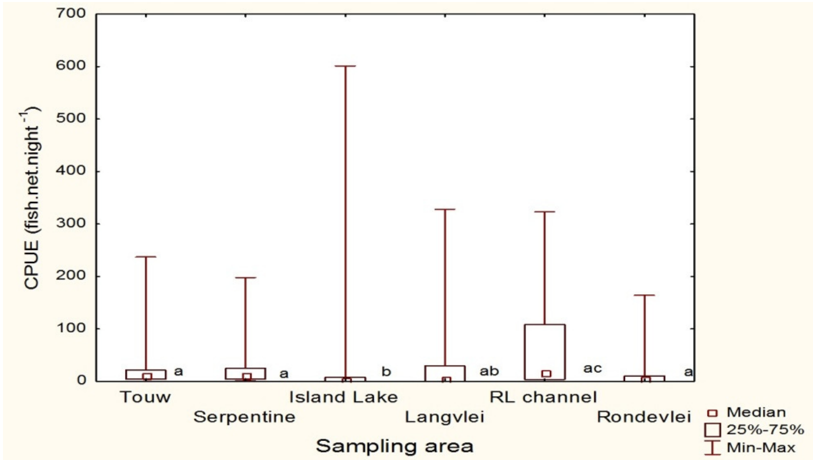


Figure 3.1. Seasonal box and whisker plot of fyke net Catch per Unit Effort (CPUE) expressed as Number of fish.net.night^{-1} in six areas of the Wilderness Lakes, Western Cape, South Africa. Sample size was 12 fyke net sets per season in Island Lake, Langvlei and Rondevlei, three in the Rondevlei-Langvlei channel, six in the Serpentine channel and nine in the Touw Estuary, over a one year period from May 2010 to May 2011. Different letters denote statistically significant differences between samples (Kruskal-Wallis ANOVA).

a)



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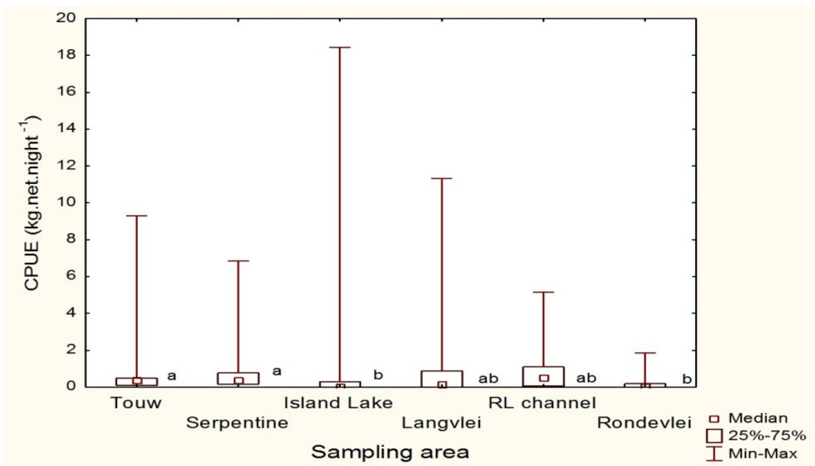


Figure 3.2. Box and whisker plots of fyke net Catch per Unit Effort (CPUE) expressed as a) Number of fish.net.night^{-1} and (b) kg.net.night^{-1} by sampling area in the Wilderness Lakes, Western Cape, South Africa. Sample size was 48 fyke net sets per lake, 12 in the Rondevlei-Langvlei channel, 24 in the Serpentine channel and 36 in Touw Estuary over a one year period from May 2010 to May 2011. Different letters denote statistically significant differences between samples (Kruskal-Wallis ANOVA).

Species Composition

A total of 5233 fishes from nine families were sampled using fyke nets and the relative importance of each of these species in each of the lakes and the Touw Estuary is summarised in **Table 3.3**. The fyke net samples were generally quite species poor, especially in Langvlei, Rondevlei and the Rondevlei-Langvlei channel. Generally species diversity was higher nearer the estuary mouth, with seven species sampled in both Island Lake and the Touw Estuary, and lower further up the system (Langvlei = two species; Rondevlei = two species). The number of fish in each of the estuarine association categories in each of the water bodies was dependent on sampling area (four categories × four sampling areas contingency table, χ^2 = 1257.07; d.f. = 9; $P < 0.001$) with the abundance of fish in the estuarine association categories I – II having the highest number of fish in Touw estuary. The abundance decreased higher up the system towards Rondevlei. Category IV showed the opposite trend, with the lowest number of fish in Touw Estuary (326 fish) and the highest number of fish in Langvlei (1916 fish).

At all sampling sites *O. mossambicus* was the most important species in terms of IRI. In the Touw Estuary, *O. mossambicus* was followed by *Caffrogobius gilchristi* (IRI = 36.2%) and *Monodactylus falciformis* (IRI = 2.6%). The other species all contributed less than 1% to the IRI (**Table 3.3**).

In Island Lake, *O. mossambicus* contributed 97.5% to the total IRI, *C. gilchristi* contributed 1.6% and other species less than 1% to the IRI.

In Langvlei and Rondevlei, *O. mossambicus* was the dominant species in terms of IRI, 99.99% and 99.98% respectively, with *M. falciformis* in Langvlei and *Psammogobius knysnaensis* in Rondevlei being only incidental catches.

The overall IRI of all species in the entire system is summarised in **Figure 3.3**. *O. mossambicus* dominated the catch composition with an IRI of 89.6%. *Caffrogobius gilchristi* was the second most important species (IRI = 9.4%) and the remainder of the species contributed a combined 1% of the total IRI.

Table 3. 3. Fish species sampled using fyke nets in the Touw Estuary, Island Lake, Langvlei and Rondevlei expressed as percent relative number of fish (%N), frequency of occurrence (%FO), percentage mass of fish (%M) and the index of relative importance (%IRI). n is indicated in parenthesis.

Species	EA	Touw Estuary				Island Lake				Langvlei				Rondevlei			
		%N (522 fish)	%FO (24 nets)	%M (19.9kg)	%IRI	%N (971 fish)	%FO (48 nets)	%M (33.1kg)	%IRI	%N (1917 fish)	%FO (48 nets)	%M (35.1kg)	%IRI	%N (538 fish)	%FO (48 nets)	%M (7.2kg)	%IRI
Caffrogobius gilchristi	Ib	35.4	83.3	17.2	36.2	2.0	20.8	2.0	1.56	0	0	0	0	0	0	0	0
Psammogobius knysnaensis	Ib	0	0	0	0	0	0	0	0	0	0	0	0	0.4	4.2	0.3	0.02
Galeichthys feliceps	IIb	0.6	12.5	1.0	0.17	0	0	0	0	0	0	0	0	0	0	0	0
Monodactylus falciformis	IIa	5.4	33.3	4.0	2.58	0.4	8.3	1.4	0.28	0.1	2.1	0.4	0.01	0	0	0	0
Liza richardsonii	IIc	0	0	0	0	0.1	2.1	0.4	0.02	0	0	0	0	0	0	0	0
Solea bleekeri	IIa	0.6	12.5	0.2	0.08	0.1	2.1	0.2	0.01	0	0	0	0	0	0	0	0
Rhabdosargus holubi	IIa	1.0	8.3	1.9	0.19	0.7	6.3	3.9	0.56	0	0	0	0	0	0	0	0
Oreochromis mossambicus	IV	62.5	54.2	73.6	60.7	96.6	27.1	91.7	97.5	99.9	60.4	99.6	99.99	99.6	72.9	99.7	99.98
Anguilla mossambica	Va	0.2	4.2	1.8	0.07	0.1	2.1	0.4	0.02	0	0	0	0	0	0	0	0

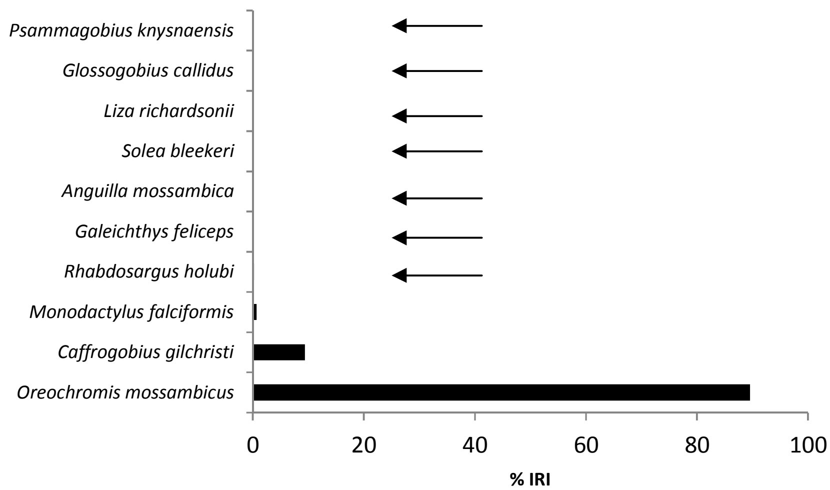


Figure 3.3. Species composition shown as index of relative importance (%IRI) of fish species caught in the fyke nets sampled throughout the Wilderness Lakes system from April 2010 to May 2011 (n = 195 nets, 5233 fish and 122 kg). Arrows indicate very low %IRI values which do not show on the graph.

B) 30 m seine nets

Seasonal CPUE

Seasonal 30 m seine net CPUE expressed as number of fish per net haul is summarised in **Figure 3.4**. CPUE did not differ significantly by season in the Touw Estuary, Island Lake and Rondevlei (Kruskal-Wallis ANOVA, $p > 0.05$). In mean Langvlei CPUE was significantly higher in autumn than in other seasons. In Langvlei, CPUE declined over the sampling period from 1415 ± 744 fish.net haul^{-1} during autumn 2010 sample to 28 ± 10 fish.net.haul^{-1} in winter (**Fig. 3.4**).

Annualised CPUE

Annual CPUE expressed as number of fish per net haul did not differ significantly between the sampling areas (Kruskal-Wallis ANOVA, $p > 0.05$) (**Fig. 3.5**). The CPUE ranged from 1225 ± 667 fish.net haul^{-1} in Rondevlei to 570 ± 233 fish.net haul^{-1} in the Touw Estuary.

In terms of biomass, CPUE was significantly higher in Island Lake, with a CPUE of 745 ± 15 kg.net haul^{-1}, than in the other three lakes (**Fig. 3.5**). Touw Estuary had the lowest CPUE, which differed significantly from Island Lake.

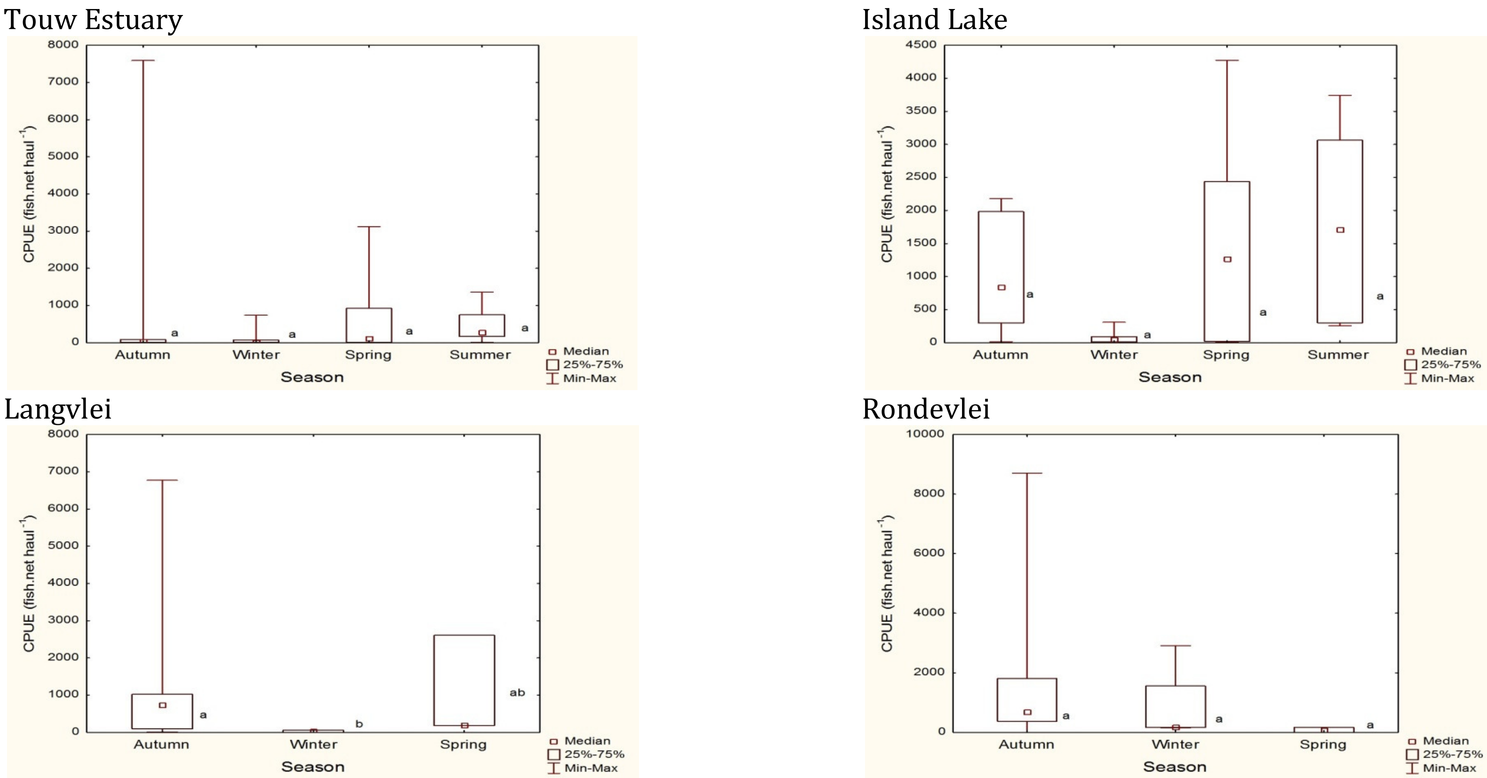


Figure 3.4. Seasonal box and whisker plot of 30 m seine net Catch per Unit Effort (CPUE) expressed as number of fish.net haul^{-1} by sampling area in the Wilderness Lakes, Western Cape, South Africa. Sample size: Touw Estuary - nine hauls, Island Lake – six hauls, Langvlei – nine hauls in autumn, 6 in winter and three hauls in spring, Rondevlei – six hauls in autumn, four hauls in winter and three hauls in spring over a one year period from May 2010 to May 2011. Different letters denote statistically significant differences between samples (Kruskal-Wallis ANOVA).

Species Composition

A total of 150055 fishes of 12 families and 17 species were sampled using the 30 m seine net. The relative importance of each of these species is summarised in **Table 3.4**. Generally species diversity in the 30 m seine net was higher nearer the estuary mouth, with 17 and 13 species sampled in Touw Estuary and Island Lake respectively, with lower diversity higher up the system (Langvlei = eight species; Rondevlei = seven species). The proportion of fish in each of the estuarine association categories was dependent on the water body (four categories × four sampling areas contingency table, $\chi^2 = 2475.32$; d.f. = 9; $P < 0.001$). The abundance of euryhaline marine and estuarine dependent category was highest in Touw Estuary and decreased up the system towards Rondevlei. The number of freshwater alien fish was highest in Island Lake (1962 fish) and lowest in Rondevlei (23 fish).

In the Touw Estuary, *A. breviceps* was the most dominant species in terms of IRI (IRI = 51%), followed by *G. aestuaria* (IRI = 33%), *Rhabdosargus holubi* (IRI = 7%), *Lithognathus lithognathus* (IRI = 4%) and *O. mossambicus* (IRI = 4%). The other species all contributed less than 2% to the IRI (**Table 3.4**).

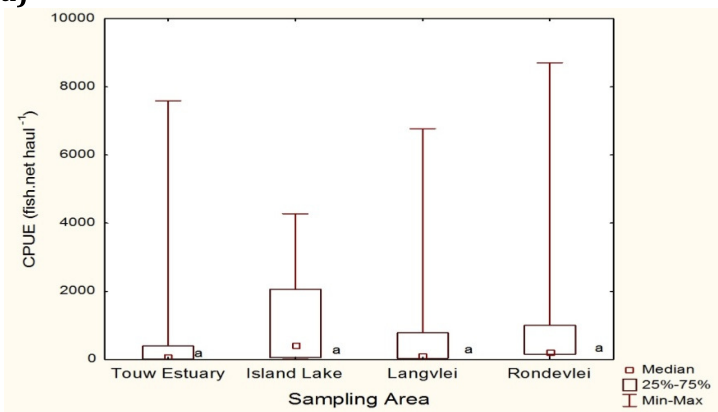
In Island Lake, *A. breviceps* contributed 47% to the total IRI, followed by *G. aestuaria* (IRI = 36%), *L. lithognathus* (IRI = 7%), *O. mossambicus* (IRI = 3%) and *Hyporhamphus capensis* (IRI = 2%) while the other species all contributed less than 3% to the IRI (**Table 3.4**).

In Langvlei, *G. aestuaria* was the most dominant species in terms of IRI (IRI = 56%), followed by *H. capensis* (IRI = 20%), *A. breviceps* (IRI = 10%), *O. mossambicus* (IRI = 8%) and *C. carpio* (IRI = 3%). The other species each contributed less than 1% to the IRI (**Table 3.4**).

In Rondevlei, the most important species in terms of IRI was *H. capensis* (IRI = 43%), followed by *G. aestuaria* (IRI = 34%), *A. breviceps* (IRI = 16%) and *O. mossambicus* (IRI = 6%) with other species each contributing less than 1% to the IRI (**Table 3.4**).

The overall importance of all species in the entire system is summarised in **Figure 3.6**. Three species dominated the catch composition. In order of importance these were *G. aestuaria* (IRI = 41%), *A. breviceps* (IRI = 28%) *H. capensis* (IRI = 21%) and lower catch rates of *O. mossambicus* (IRI = 5%) which together made up more than 90% of the total IRI. There were incidental catches of *Redigobius dewaali*, *C. gilchristi*, *L. amia*, *Myxus capensis* and *Mugil cephalus*, all of which contributed less than 1% to the IRI.

a)



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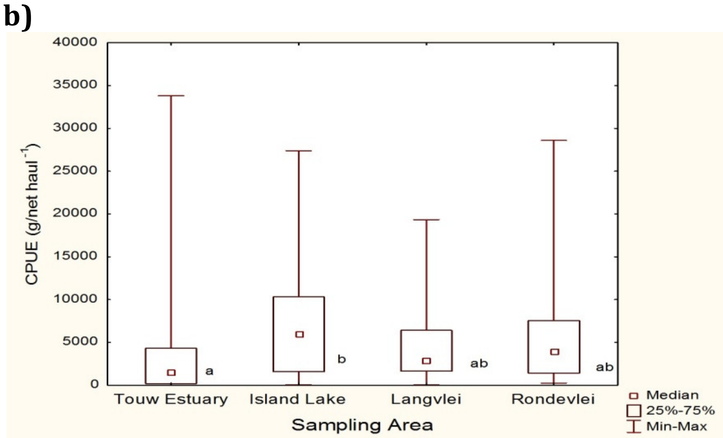


Figure 3.5. 30 m seine net Catch per Unit Effort (CPUE) expressed as (a) Number of fish.net haul^{-1} and (b) kg.net haul^{-1} by sampling area in the Wilderness Lakes, Western Cape, South Africa. Sample size was 36 hauls in Touw Estuary, 24 hauls in Island Lake, 18 hauls in Langvlei, 13 hauls in Rondevlei over a one year period from May 2010 to May 2011. Different letters denote statistically significant differences between samples (Kruskal-Wallis ANOVA).

Table 3. 4. Fish species sampled in the 30 m seine net in the Touw Estuary, Island Lake, Langvlei and Rondevlei expressed as percent relative number of fish (%N), frequency of occurrence (%FO), percentage mass of fish (%M) and the index of relative importance (%IRI). n is shown in parenthesis.

		Touw Estuary				Island Lake				Langvlei				Rondevlei			
Species	EA	%N	%FO	%M	IRI	%N	%FO	%M	IRI	%N	%FO	%M	IRI	%N	%FO	%M	IRI
		(51439 fish)	(36 hauls)	(158098g)		(58184 fish)	(24 hauls)	(179472g)		(18751 fish)	(18 hauls)	(79798g)		(21681 fish)	(13 hauls)	(80706g)	
Atherina breviceps	Ib	60.1	72.2	14.5	50.70	54.0	83.3	17.4	47.12	15.3	38.9	10.8	9.60	26.5	53.8	14.7	15.75
Gilchristella aestuaria	Ib	38.6	63.9	16.0	32.80	41.5	79.2	16.6	36.48	73.2	66.7	16.1	56.27	39.3	84.6	16.9	33.80
Caffrogobius gilchristi	Ib	0.002	2.8	0.01	0.0002	0	0	0	0	0	0	0	0	0	0	0	0
Psammogobius knysnaensis	Ib	0.4	63.9	0.2	0.33	0.2	50.0	0.2	0.14	0.01	5.6	0.001	0.0004	0.03	30.8	0.01	0.007
Redigobius dewaali	Ib	0.01	8.3	0.001	0.001	0.002	4.2	0.0004	0.0001	0	0	0	0	0	0	0	0
Hyporhamphus capensis	Ia	0.004	2.8	0.0001	0.0001	0.7	66.7	3.3	2.14	11.2	77.8	15.6	19.72	34.0	76.9	44.2	42.74
Syngnathus acus	Ib	0.1	30.6	0.02	0.03	0.02	25.0	0.01	0.01	0	0	0	0	0.02	15.4	0.01	0.004
Lichia amia	IIb	0.01	11.1	2.2	0.23	0.01	8.3	7.9	0.52	0	0	0	0	0	0	0	0
Monodactylus falciformis	IIa	0.03	22.2	0.6	0.12	0	0	0	0	0	0	0	0	0	0	0	0
Mugil cephalus	IIa	0	0	0	0	0.002	4.2	0.3	0.01	0	0	0	0	0	0	0	0
Liza richardsonii	IIc	0.04	11.1	7.7	0.80	0.03	12.5	6.1	0.61	0.1	22.2	11.2	2.35	0.1	30.8	9.78	2.154
juvenile Mugilidae	IIa	0.2	5.6	0.1	0.01	0	0	0	0	0	0	0	0	0	0	0	0
Lithognathus lithognathus	IIa	0.1	27.8	15.3	4.00	0.1	62.5	13.9	6.91	0	0	0	0	0	0	0	0
Rhabdosargus holubi	IIa	0.2	30.6	22.3	6.50	0.1	20.8	9.2	1.53	0	0	0	0	0	0	0	0
Oreochromis mossambicus	IV	0.2	22.2	20.9	4.40	3.4	25.0	13.9	3.42	0.1	38.9	22.5	8.31	0.1	53.8	14.4	5.543
Cyprinus carpio	IV	0.004	2.8	0.0003	0.0001	0.01	12.5	11.1	1.10	0.03	16.7	20.6	3.25	0	0	0	0
Gambusia affinis	IV	0.2	27.8	0.01	0.06	0	0	0	0	0	0	0	0	0	0	0	0
Myxus capensis	Vb	0.002	2.8	0.2	0.005	0	0	0	0	0.02	16.7	3.1	0.49	0	0	0	0

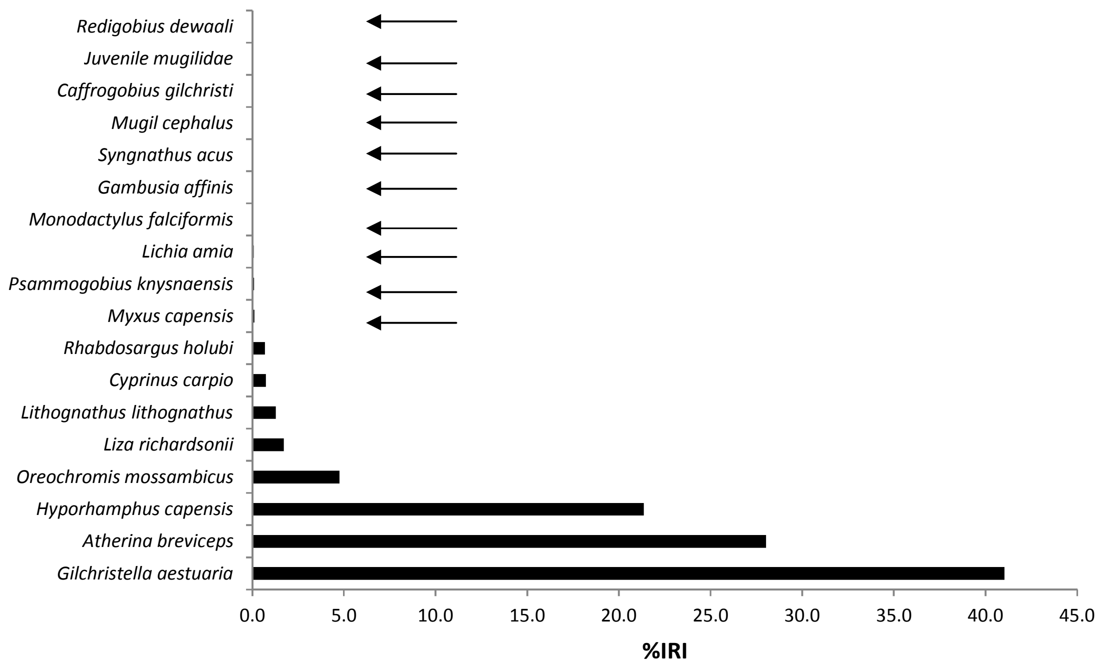


Figure 3.6. Species composition shown as relative importance (%IRI) of fish species caught in the 30 m seine net sampled throughout the Wilderness Lakes system from April 2010 to May 2011 (n = 91 hauls, 150055 fish and 417 kg). Arrows show species with %IRI values too low to appear on the graph.

C) 10 m seine nets

Seasonal CPUE

Seasonal 10 m seine net CPUE, expressed as number of fish per haul is summarised in **Figure 3.7**. CPUE did not differ significantly by season in the Rondevlei-Langvlei channel (Kruskal-Wallis ANOVA, $p > 0.05$). In Island Lake CPUE was significantly lower in winter and higher in spring and summer (Kruskal-Wallis ANOVA, $H = 15.32$; $p < 0.05$). In Langvlei CPUE was significantly higher in spring (Kruskal-Wallis ANOVA, $H = 18.74$; $p < 0.05$). In Rondevlei, CPUE declined from 229 ± 58 fish. net haul^{-1} in summer to 3 ± 1 fish. net haul^{-1} in autumn (**Appendix 3**).

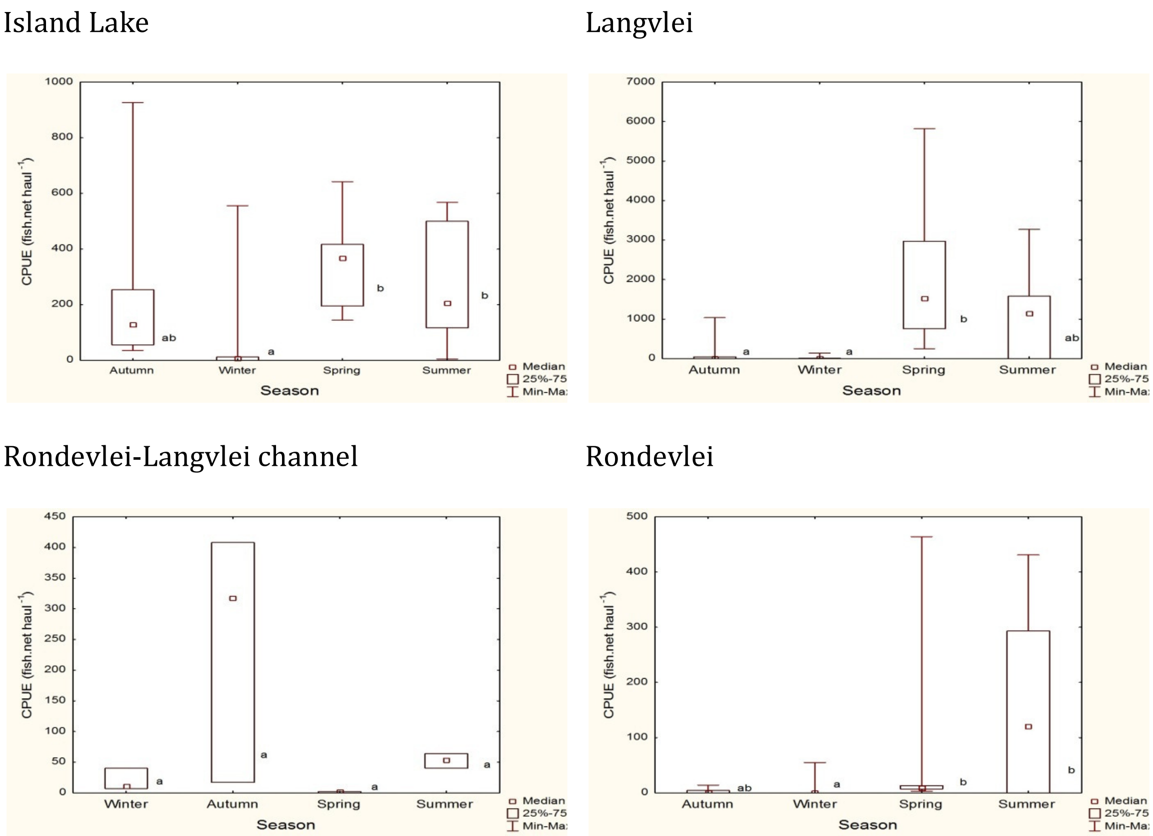


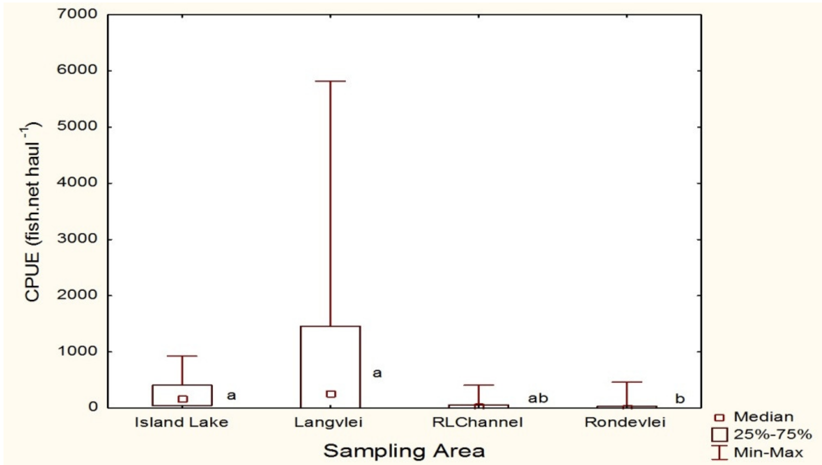
Figure 3.7. Box and whisker plot of seasonal 10 m seine net Catch per Unit Effort (CPUE) expressed as number fish.net haul^{-1} by sampling area in the Wilderness Lakes, Western Cape, South Africa. Sample size was ten hauls in Island Lake, ten hauls, ten hauls (seven in summer) in Rondevlei and Langvlei and three in the Rondevlei-Langvlei channel over a one year period from May 2010 to May 2011. Different letters denote statistically significant differences between samples (Kruskal-Wallis ANOVA).

<u>Annualised CPUE</u>

CPUE expressed as number of fish per haul, differed significantly between sampling areas (Kruskal-Wallis ANOVA, $H = 21.3$; $p < 0.05$) (**Fig. 3.8**). CPUE in Rondevlei was significantly lower than that in the other two lakes and the Rondevlei Langvlei channel where CPUE ranged from 80 ± 39 fish. net haul^{-1} in the Rondevlei-Langvlei channel to 901 ± 213 fish. net haul^{-1}in Langvlei.

In terms of biomass, CPUE was significantly lower in Rondevlei (0.2 ± 0.1 kg. net haul^{-1}) than in the other two lakes and channel. CPUE between Langvlei, Island Lake and the Rondevlei-Langvlei channel did not differ significantly (**Fig. 3.8**).

a)



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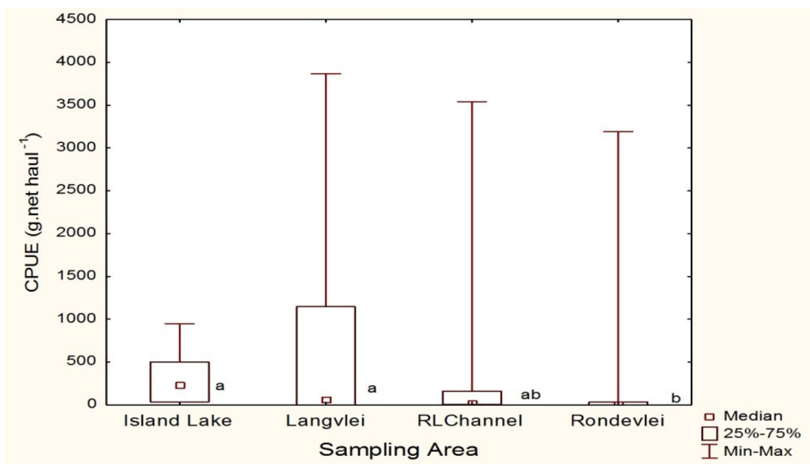


Figure 3.8. Box and whisker plot of 10 m seine net catch per unit effort (CPUE) expressed as (a) number fish.net haul^{-1} and (b) kg.net haul^{-1} by sampling area in the Wilderness Lakes, Western Cape, South Africa. Sample size was 40 hauls in Island Lake, 37 in Rondevlei and Langvlei and 12 in the Rondevlei-Langvlei channel over a one year period from May 2010 to May 2011. Different letters denote statistically significant differences between samples (Kruskal-Wallis ANOVA).

Species Composition

A total of 45888 fishes of eight species in eight families were sampled using the 10 m seine net. The relative importance of each of these species is summarised in **Table 3.5**. The species diversity was highest in Island Lake (seven species), with Rondevlei and Langvlei sampling six species each. The number of fish in each of the estuarine association categories in each of the water bodies was dependent on sampling area (four categories × four sampling areas contingency table, $\chi^2 = 7406.6$; d.f. = 4; $P < 0.001$).

In each of the lakes, the shoaling pelagic species dominated the catch composition numerically and were the most important species in terms of IRI.

In Island Lake, *A. breviceps* was the dominant species in terms of IRI (IRI = 49%), followed by *G. aestuaria* (44%) *O. mossambicus* (IRI = 4%) and *P. knysnaensis* (IRI = 3%). Other species all contributed less than 1% to the IRI (**Table 3.5**).

In Langvlei, *G aestuaria* was the most important species in the 30m seine net catches (IRI = 50%), followed by *A. breviceps* (IRI = 46%) and *O. mossambicus* (IRI = 3%). The remaining species contributed a combined 1% to the IRI (**Table 3.5**).

In Rondevlei, the most important species in terms of IRI was *A. breviceps* (IRI = 61%), followed by *O. mossambicus* (IRI = 35%) and *P. knysnaensis* (IRI = 3%) with the other species all contributing less than 2% to the IRI (**Table 3.5**).

The overall importance of all species in the entire system is summarised in **Figure 3.9**. Three species dominated the catch composition. In order of importance these were *A. breviceps* (IRI = 51%), *G. aestuaria* (IRI = 31%) and *O. mossambicus* (IRI = 14%) which together made up more than 95% of the total IRI.

Table 3. 5. Fish species sampled in the 10 m seine net in Island Lake, Langvlei and Rondevlei expressed as percent relative number of fish (%N), frequency of occurrence (%FO), percentage mass of fish (%M) and the index of relative importance (%IRI). n is shown in parenthesis.

		Island Lake				Langvlei				Rondevlei			
Species	EA	%N (9357 fish)	%FO (40 hauls)	%M (15.3kg)	%IRI	%N (33327 fish)	%FO (37 hauls)	%M (28.6kg)	%IRI	%N (2249 fish)	%FO (37 hauls)	%M (7.7kg)	%IRI
Atherina breviceps	Ib	49.0	77.5	34.2	49.03	44.1	75.7	28.9	46.18	53.1	54.1	22.9	61.18
Gilchristella aestuaria	Ib	45.9	72.5	33.2	43.60	53.9	54.1	57.4	50.28	1.6	10.8	0.3	0.31
Psammogobius knysnaensis	Ib	2.3	67.5	3.7	3.12	0.1	29.7	0.2	0.07	4.2	35.1	1.0	2.75
Hyporhamphus capensis	Ia	0.6	17.5	0.8	0.19	0.03	16.2	0.003	0.004	4.6	8.1	4.4	1.09
Syngnathus acus	Ib	0.5	37.5	0.1	0.15	0	0	0	0	0.5	16.2	0.1	0.14
juvenile Mugilidae	II	0.03	2.5	0.04	0.001	0	0	0	0	0	0	0	0
Oreochromis mossambicus	IV	1.5	17.5	27.8	3.91	1.8	27	13.5	3.47	36.0	21.6	71.3	34.53
Gambusia affinis	IV	0	0	0	0	0.02	10.8	0.02	0.002	0	0	0	0

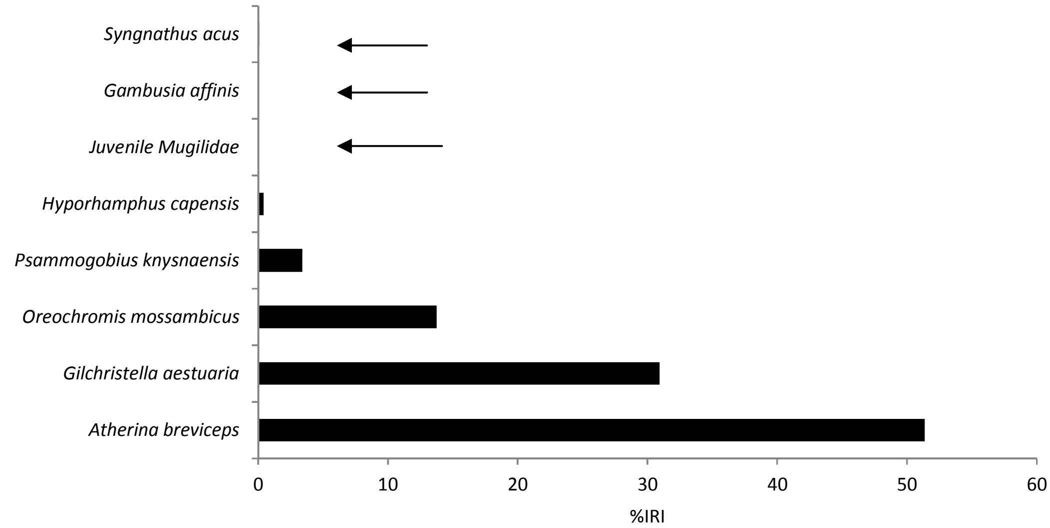


Figure 3.9. Species composition shown as relative importance (%IRI) of fish species caught in the 10 m seine net sampled throughout the Wilderness Lakes system. (n = 126 hauls, 45894 fish and 56 kg). Arrows show species with %IRI values too low to appear on the graph.

D) Gill nets

Seasonal CPUE

Seasonal gill net CPUE, expressed as number of fish per net set is summarised in **Figure 3.10**. CPUE did not differ significantly by season in the Touw estuary and in Island Lake (Kruskal-Wallis ANOVA, $p > 0.05$). In Langvlei CPUE was significantly higher in summer (Kruskal-Wallis ANOVA, $H = 10.8$; $p < 0.05$). In Rondevlei, CPUE declined for the sampling period from 30 ± 3 fish. net. night^{-1} set in the winter 2010 sample to 7 ± 2 fish.net. night^{-1} set during autumn and summer (**Fig. 3.10**).

Annualised CPUE

CPUE expressed as number of fish. net. night^{-1} differed significantly between sampling areas (Kruskal-Wallis ANOVA, $F = 32.4$; $p < 0.05$). CPUE in the Touw Estuary was significantly lower than that in the three lakes where CPUE ranged from 15 ± 2 fish. net. night^{-1} in Rondevlei to 24 ± 3 fish. net. night^{-1} in Langvlei (**Fig. 3.11**).

In terms of biomass, CPUE was significantly lower in the Touw Estuary (3 ± 1 kg. net. night^{-1}) than in the three lake sites (**Fig. 3.11**). Biomass CPUE between the three lake sites did not differ significantly.

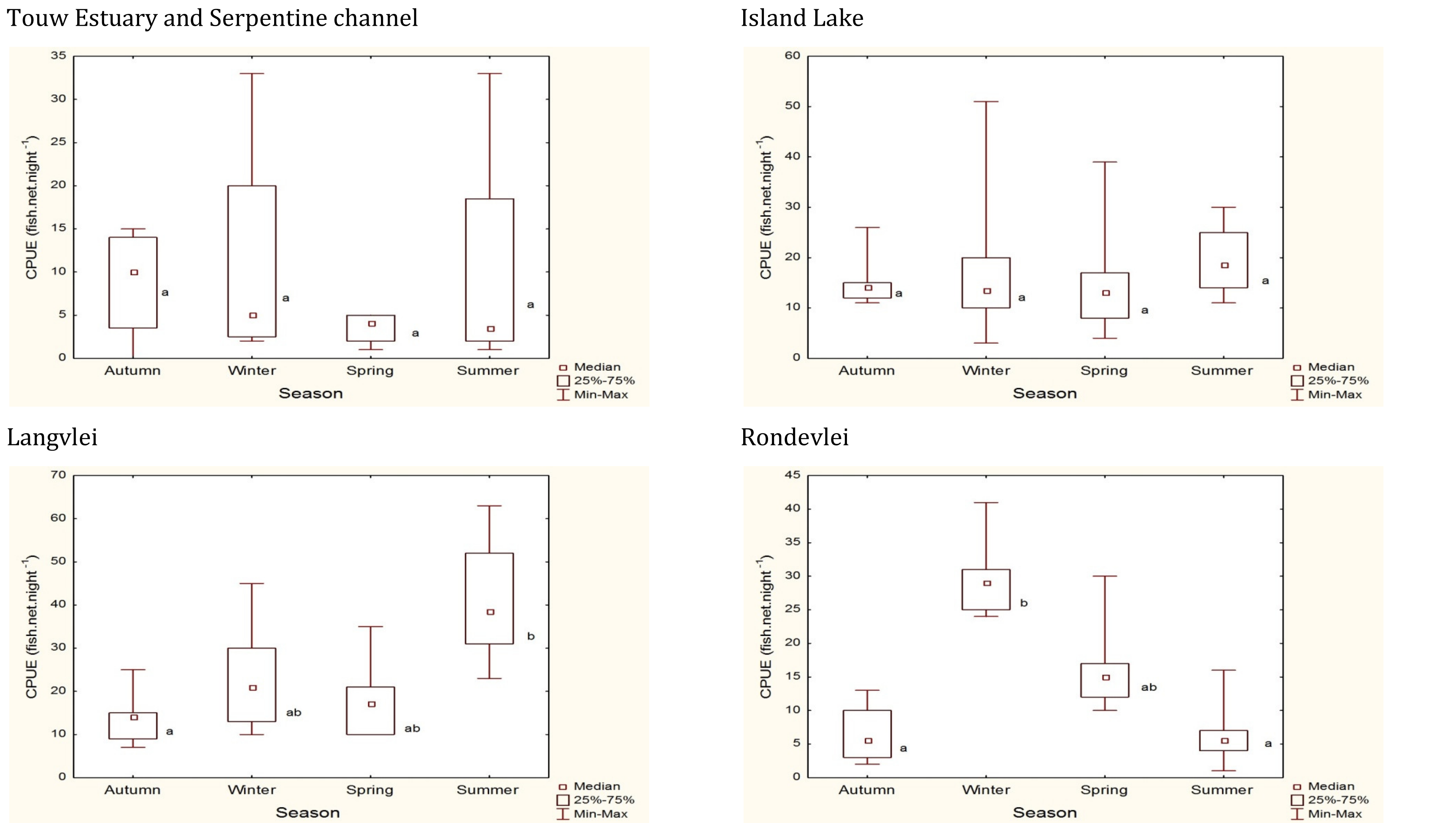
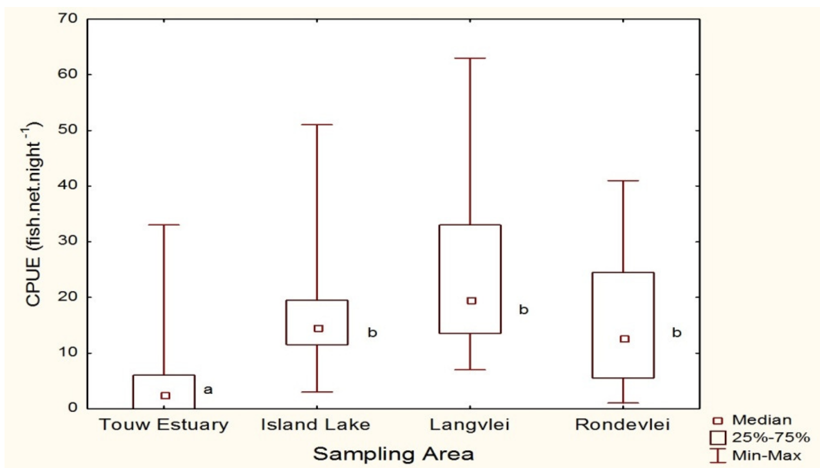


Figure 3.10. Seasonal box and whisker plot of gill net catch per unit effort (CPUE) expressed as number of fish.net^{-1} in four sampling sites of the Wilderness Lakes, Western Cape, South Africa. Sample size was six gill net sets per site over a one year period from May 2009 to May 2010. Different letters denote statistically significant differences between samples (Kruskal-Wallis ANOVA).

Species Composition

A total of 1478 fishes of 15 species in ten families were sampled using gill nets. The IRI of each of these species is summarised in **Table 3.6**. Generally species diversity in gill nets was higher nearer the estuary mouth, with 11 and 13 species sampled from the Touw Estuary and Island Lake respectively, and lower, higher up the system (Langvlei = seven species; Rondevlei = 10 species). The number of fish in each of the estuarine association categories in each of the water bodies was dependent on sampling area (four categories × four sampling areas contingency table, $\chi^2 = 430.35$; d.f. = 6; $P < 0.001$).

a)



b)

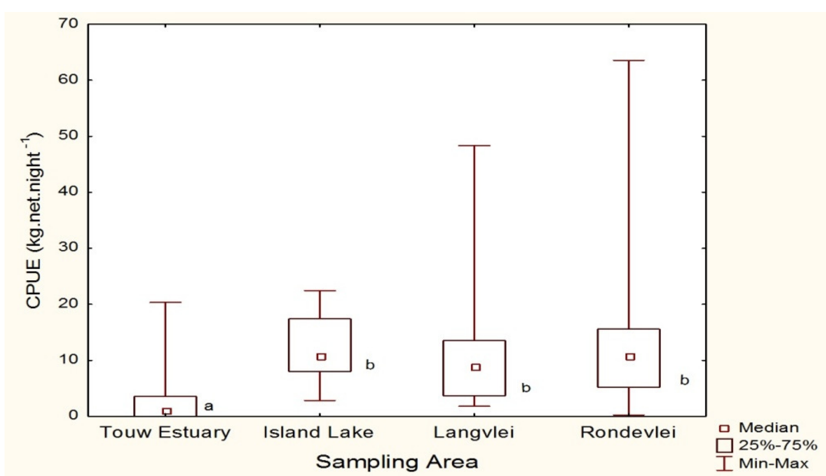


Figure 3.11. Box and whisker plot of gill net catch per unit effort (CPUE) expressed as (a) number of fish.net.night^{-1} and (b) kg.net.night^{-1} by sampling area in the Wilderness Lakes, Western Cape, South Africa. Sample size was 24 gill net sets in Island Lake, Langvlei and Rondevlei and 16 in the Touw Estuary site over a one year period from May 2010 to May 2011. Different letters denote statistically significant differences between samples (Kruskal-Wallis ANOVA).

Table 3. 6. Fish species sampled in the gill nets in the Touw Estuary, Island Lake, Langvlei and Rondevlei expressed as percent relative number of fish (%N), frequency of occurrence (%FO), percentage mass of fish (%M) and the index of relative importance (%IRI). n is indicated in parentheses.

Species	EA	Touw Estuary (n= 134)				Island Lake (n= 414)				Langvlei (n= 580)				Rondevlei (n= 355)			
		%N (134 fish)	%FO (16 nets)	%M (68.0kg)	%IRI	%N (414 fish)	%FO (24 nets)	%M (278.3kg)	%IRI	%N (580 fish)	%FO (24 nets)	%M (237.3kg)	%IRI	%N (355 fish)	%FO (24 nets)	%M (317.3kg)	%IRI
Galeichthys feliceps	IIb	39.6	18.8	4.4	14.53	0	0	0	0	0	0	0	0	0	0	0	0
Lichia amia	IIb	4.5	18.8	9.8	4.72	1.7	16.7	9.4	1.33	0	0	0	0	0.3	4.2	3.4	0.13
Elops machnata	IIa	0	0	0	0	0.2	4.2	0.5	0.02	0	0	0	0	0	0	0	0
Pomadasys commersonnii	IIa	14.9	56.3	26	40.59	15.9	75.0	22.1	20.4	0	0	0	0	0.3	4.2	1.4	0.06
Monodactylus falciformis	IIa	9.0	18.8	1.1	3.34	3.4	33.3	0.7	0.97	47.9	87.5	9.8	42.6	26.5	75.0	5.9	20.40
Mugil cephalus	IIa	0.7	6.3	1.7	0.27	3.1	25.0	6.3	1.70	0.5	12.5	2.2	0.29	2.3	25.0	5.3	1.58
Liza dumerili	IIb	1.5	12.5	0.7	0.47	20.8	75.0	8.9	16.00	0.2	4.2	0.1	0.01	0	0	0	0
Liza richardsonii	IIc	17.2	37.5	15.8	21.84	39.9	91.7	35.8	49.70	9.8	70.8	16.2	15.50	23.7	79.2	21.1	29.90
Liza tricuspidens	IIb	0	0	0	0	0.5	8.3	0.8	0.08	0	0	0	0	0.8	8.3	0.7	0.11
Argyrosomus japonicus	IIa	0	0	0	0	0.2	4.2	0.2	0.01	0	0	0	0	0	0	0	0
Lithognathus lithognathus	IIa	3.0	12.5	3.3	1.40	5.8	50.0	8.7	5.19	0	0	0	0	1.7	16.7	8.9	1.49
Rhabdosargus holubi	IIa	0	0	0	0	1.7	20.8	1.3	0.44	0.3	4.2	0.3	0.02	5.1	25.0	0.1	1.08
Oreochromis mossambicus	IV	4.5	12.5	29.9	7.59	0.2	4.2	0.03	0.01	36.2	50.0	53.5	37.8	8.2	45.8	23.3	12.10
Cyprinus carpio	IV	0.7	6.3	0.2	0.10	0	0	0	0	0	0	0	0	0	0	0	0
Myxus capensis	Vb	4.5	25	7.2	5.217	6.5	50.0	5.0	4.14	5.0	45.8	4.8	3.78	31.0	70.8	24.6	33.20

In the Touw estuary, *Pomadasys commersonnii* was the most important species in terms of IRI (IRI = 41%), followed by *Liza richardsonii* (IRI = 22%), *Galeichthys feliceps* (IRI = 15%), *O. mossambicus* (7.6%) and *M. capensis* (5%). The other species all contributed less than 5% to the IRI (**Table 3.6**).

In Island Lake, *L. richardsonii* was the most important species in terms of IRI (IRI = 50%), followed by *P. commersonnii* (IRI = 20%), *Liza dumerili* (IRI = 16%) and *L. lithognathus* (IRI = 5%) with other species contributing less than 5% to the IRI respectively (**Table 3.6**).

In Langvlei, *M. falciformis* was most important species in terms of IRI (IRI = 43%), followed by *O. mossambicus* (IRI = 38%) and *L. richardsonii* (IRI = 16%). Other species all contributed less than 5% to the IRI (**Table 3.6**).

In Rondevlei, the most important species in terms of IRI was *M. capensis* (IRI = 33%), followed by *L. richardsonii* (IRI = 30%), *M. falciformis* (IRI = 20%) and *O. mossambicus* (IRI = 12%) with other species contributing less than 5% to the total IRI (**Table 3.6**).

The overall importance of all species in the entire system is summarised in **Figure 3.12**. Seven species dominated the catch composition. In order of importance these were *L. richardsonii* (IRI = 28%), *M. falciformis* (IRI = 16%), *P. commersonnii* (IRI = 15%), *O. mossambicus* (IRI = 14%), *M. capensis* (IRI = 10%) *L. lithognathus* (IRI = 6%) and *G. feliceps* (IRI = 4%) which together made up more than 90% of the total IRI.

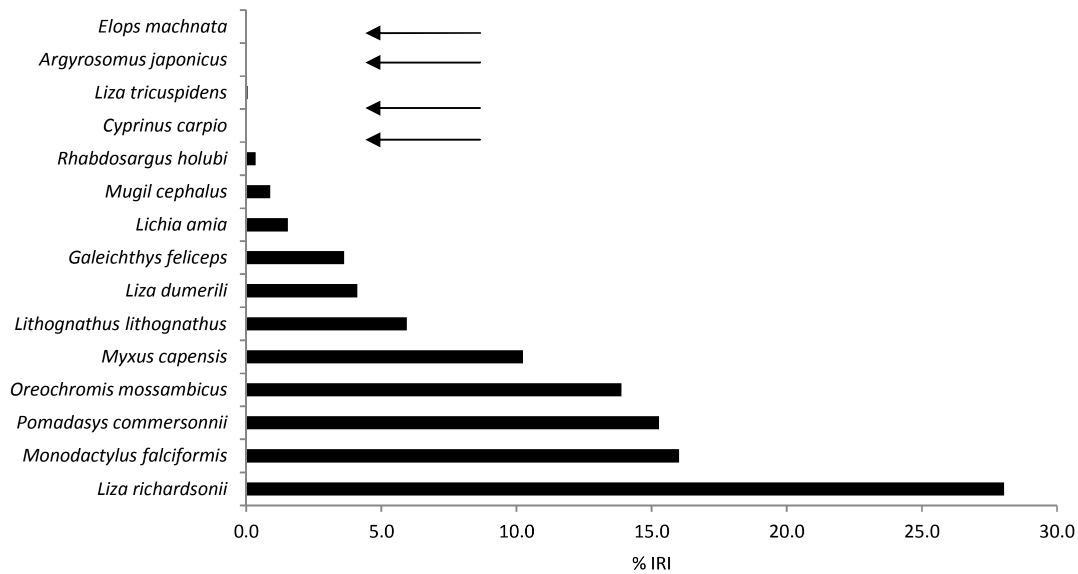


Figure 3.12. Species composition shown as relative importance (%IRI) of fish species caught in the gill nets sampled throughout the Wilderness Lakes system. (n = 88 hauls, 1483 fish and 901 kg). Arrows show species with %IRI values too low to appear on the graph.

E) Species diversity

The species saturation fitted from von Bertalanffy function curves are shown in **Figure 3.13**. Sampling reached saturation when the curve reached an asymptote where as sampling effort increased; the number of species sampled remained constant. In the gill net sampling, saturation was reached in Rondevlei after 11 gill net sets. In the Touw Estuary, saturation was reached after 13 net sets and in Island Lake and Langvlei the number of species sampled had not reached saturation yet. In the Touw Estuary, saturation in the 30 m seine net was reached, however in Island Lake, Langvlei and Rondevlei, saturation was not reached yet. A von Bertalanffy curve could not be fitted to the Langvlei 30 m seine net sampling due to a low number of species being sampled initially, however no new species were sampled after 16 hauls. In the 10 m seine net sampling, Langvlei reached saturation after seven hauls. In Rondevlei and Island Lake all species were sampled after 24 and seven hauls respectively with one species being sampled after asymptote was reached. In the fyke net sampling, Island Lake reached saturation after 46 fyke net nights and the Touw Estuary reached saturation after 34 fyke net nights. A von Bertalanffy curve could not be fitted to the Langvlei and Rondevlei fyke net sampling as one species was sampled for 36 and 29 fyke net set respectively before a second species was sampled. However the number of species sampled remained at two for the sampling period.

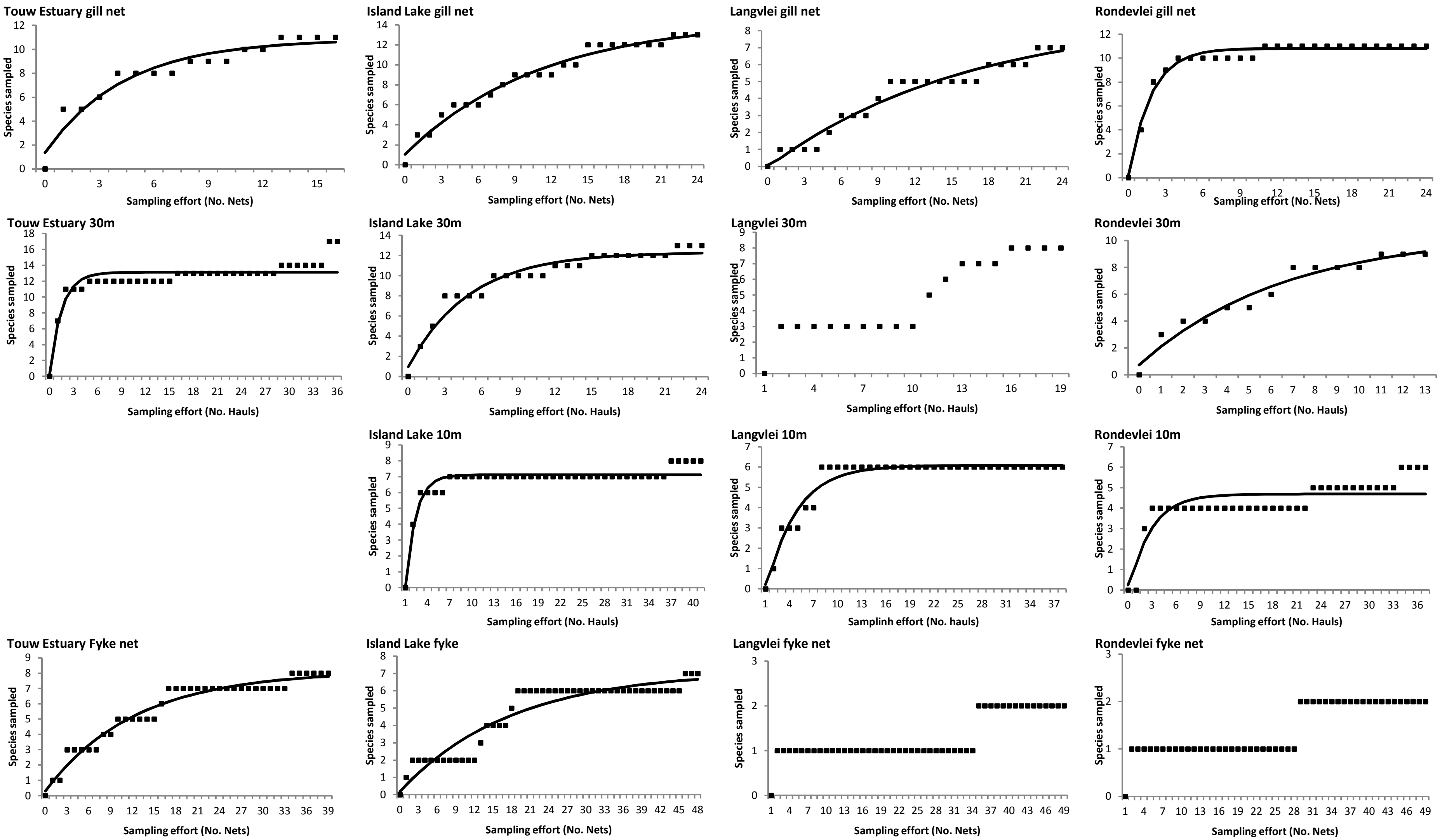


Figure 3.13. Observed species saturation and von Bertalanffy function curve for fyke nets, gill nets, 10 m and 30 m seine nets used in the Wilderness Lakes system during the sampling period May 2010 – May 2011.

The Wilderness Lakes system ichthyofauna comprises fishes in all but one (pure marine-category III) of Whitfield's (1998) estuarine categories, many of which are distributed throughout the system. The euryhaline marine category (category IIa-IIc) had the highest number of species (13 species) followed by native estuarine species (category Ia-Ib; eight species), alien freshwater (category IV, three species) and two catadromous (category V) species (**Table 3.7**). A total of 26 species representing 18 families was sampled throughout the Wilderness Lakes system.

Table 3.7. Summary table showing the overall distribution of fish species sampled from the Wilderness Lakes system, Western Cape, South Africa between autumn 2010 and summer 2011. EA = Estuarine Association category (after Whitfield 1998), Locality: TE = Touw Estuary; SC = Serpentine channel; IL = Island Lake; C = channel; LV = Langvlei, RLC = Rondevlei-Langvlei channel and RV = Rondevlei. Sampling gear type: F = Fyke, 10 m = 10 m seine net, 30 m = 30 m seine net, G = gill net, S = scoop net and A = angling. Black shading indicates confirmed distribution and grey indicates probable distribution due to a catch higher up the system.

Family	Species	EA	Locality							Gear type
			TE	SC	IL	C	LV	RLC	RV	
Anguilidae	*Anguilla mossambica*	Va								F
Ariidae	*Galeichthys feliceps*	IIb								G,F
Atherinidae	*Atherina breviceps*	Ib								10 m, 30 m
Carangidae	*Lichia amia*	IIa								G, 30 m
Cichlidae	*Oreochromis mossambicus**	IV								F, 10 m, 30 m, G
Clupidae	*Gilchristella aestuaria*	Ib								10 m, 30 m
Cyprinidae	*Cyprinus carpio**	IV								F, 30 m
Elopidae	*Elops machnata*	IIa								G
Gobiidae	*Caffrogobius gilchristi*	Ib								F, 30 m
	Glossogobius callidus	Ib								F
	Psammogobius knysnaensis	Ib								F, 30 m,10 m
	Redigobius dewaali	Ib								10 m, 30 m
Haemulidae	*Pomadasys commersonnii*	IIa								G
Hemiramphidae	*Hyporhamphus capensis*	Ia								10 m, 30 m
Monodactylidae	*Monodactylus falciformis*	IIa								F, 30 m, G
Mugilidae	*Mugil cephalus*	IIa								G
	Myxus capensis	Vb								G, 30 m
	Liza dumerili	IIb								G
	Liza richardsonii	IIc								F, G, 30 m
	Liza tricuspidens	IIb								G
	juvenile Mugilidae	II								30 m
Poeciliidae	*Gambusia affinis**	IV								30 m, 10 m, S
Sciaenidae	*Argyrosomus japonicus*	IIa								G
Soleidae	*Solea bleekeri*	IIa								F
Sparidae	*Lithognathus lithognathus*	IIa								30 m, G
	Rhabdosargus holubi	IIa								F, 30 m, G
Syngnathidae	*Syngnathus acus*	Ib								30 m, 10 m, G

* Alien fish species

The Touw Estuary and Island Lake sampled the highest number of species (23 and 24 species respectively). Langvlei sampled the lowest number of species at 15, made up of native estuarine and euryhaline marine species and three alien freshwater species. Species composition by estuarine association categories is shown in **Figure 3.14**. The proportion of species in each category was independent of sampling area (four categories × four sampling areas contingency table, $\chi 2 = 1.36$; d.f. = 9; $P > 0.01$).

Native estuarine species found in the Wilderness Lakes system comprised mostly pelagic shoaling species (*A. breviceps*, *G. aestuaria* and *H. capensis*) and benthic species such as gobies. The alien freshwater component of the species diversity was comprised of common carp (*C. carpio*), mosquitofish (*G. affinis*) and Mozambique tilapia (*O. mossambicus*) (**Table 3.7**).

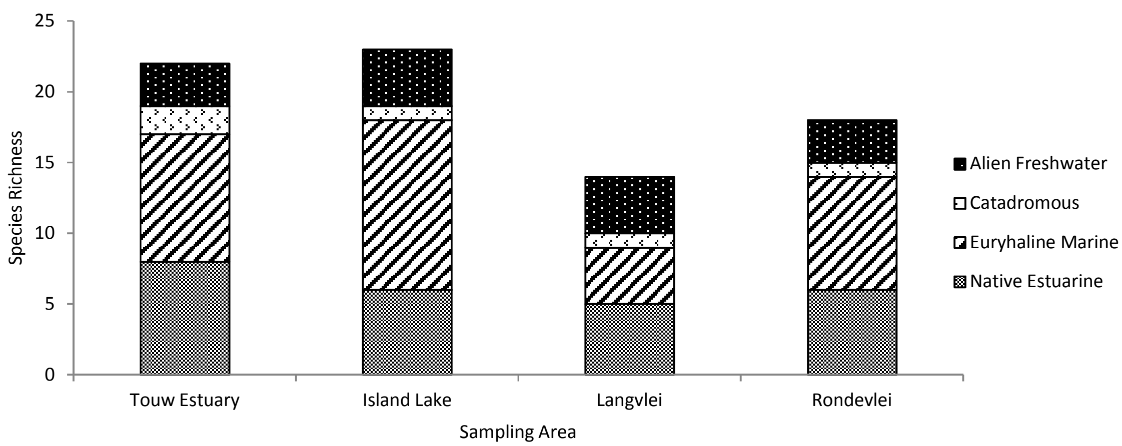


Figure 3.14. Species richness for the Wilderness Lakes system separated into estuarine association categories set out by Whitfield (1998). Alien freshwater represents category IV, Catadromous represents category V, Euryhaline marine represents category II and Native Estuarine represents category I.

The relative biomass of each group of species for each of the lakes and the Touw Estuary is shown in **Figure 3.15**. Langvlei had the highest biomass of alien fish species (52%) but the lowest biomass of euryhaline marine species (15.4%). Island Lake has the lowest biomass of alien species (15.3%) but the highest mass of euryhaline marine species (66.1%) (**Fig.3.15**). The native estuarine species contributed between 15% and 21% to each of the lakes and the Touw Estuary (**Fig. 3.15**). Rondevlei had the highest biomass (19.6%) and the Touw Estuary had the lowest biomass (2.2%) of catadromous species. The biomass of species in each of the estuarine association categories in each of the water bodies was dependent on water body (four sampling areas × four estuarine categories contingency table, $\chi^2 = 314.25$; d.f. = 9; $P < 0.001$)

mainly due to differences in the weight of euryhaline marine species, which were more abundant in Island Lake, and the higher weight of the alien freshwater species in Langvlei.

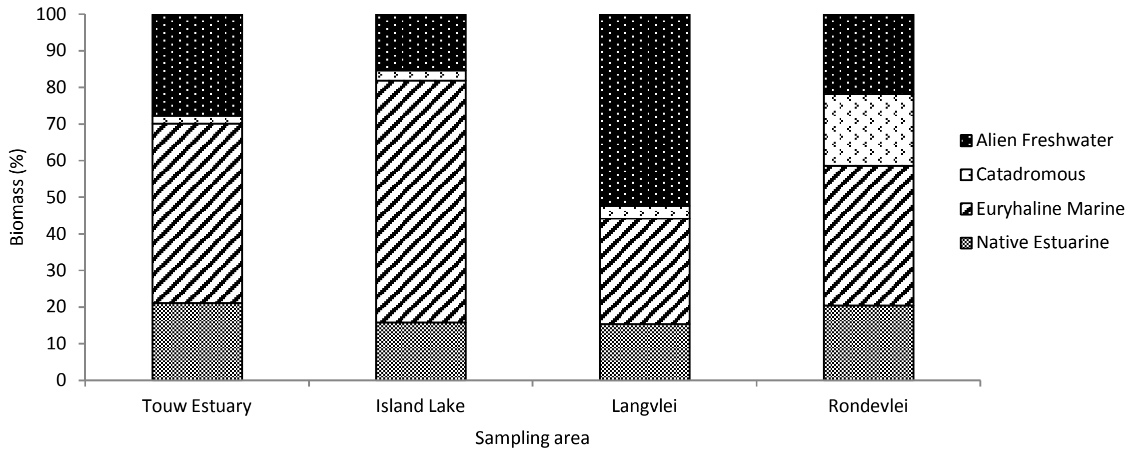


Figure 3.15. Accumulative biomass (%) of fish species combining catch from all gears in each of the lakes separated by the estuarine dependence categories as set out by Whitfield (1998). Alien freshwater represents category IV, Catadromous represents category V, Euryhaline marine represents category II and Native Estuarine represents category I. Effort for each gear type was relatively consistent.

F) Comparison with other estuaries

The species composition in the Wilderness Lakes system did not differ significantly (five categories × five estuaries contingency table, $\chi 2 = 13.63$; d.f. = 16; $P > 0.05$) from that in other temporary open/closed estuaries in the warm-temperate region when grouped into Whitfield's (1998) estuarine association categories (**Fig. 3.16**). The number of species sampled in the Wilderness Lakes system was higher than the other estuaries sampled, with Swartvlei Estuary, which is situated just north of the Wilderness Lakes, only sampling 20 species in comparison with 26 species sampled in the Wilderness Lakes. The species composition in terms of estuarine association was very similar between estuaries, with each estuary having the highest number of species from the euryhaline marine category (lowest: Blinde Estuary - six; highest: Wilderness Lakes - 13 species) and between one (Blinde Estuary) and eight (Wilderness Lakes) species in the native estuarine category (**Fig. 3.16**).

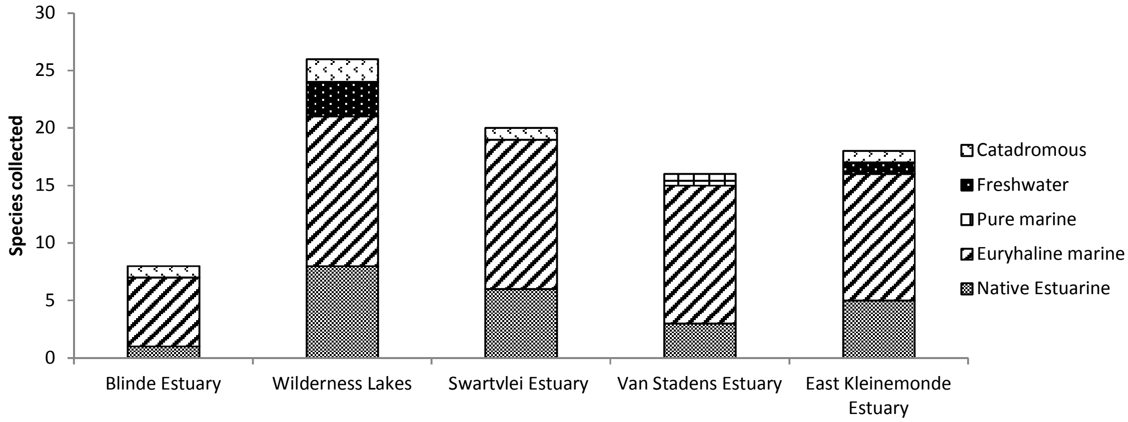


Figure 3.16. Comparison of species sampled in five temporary open/closed estuaries in the warm temperate region of South Africa grouped into their respective estuarine association categories (Whitfield 1998). The estuaries sampled were: Wilderness Lakes (this study), Blinde Estuary, Van Stadens Estuary and East Kleinemonde Estuary (James *et al*. 2007) and Swartvlei Estuary (James & Harrison 2008).

G) Length-frequency

Galeichthys feliceps, *Solea bleekeri, Elops machnata, A. japonicus, Glossogobius callidus, R. dewaali* and *Liza tricuspidens* were not sampled in high enough numbers to allow for meaningful representation of length-frequency distributions, and are summarised in **Table 3.8**. The length-frequency of the native estuarine species, *A. breviceps*, *P. knysnaensis*, *H. capensis*, *C. gilchristi*, and *G. aestuaria* indicate normally distributed population structure **(Fig. 3.17.)** with juveniles and adults being present in the population.

Table 3.8. Summary of length of fish species sampled in the Wilderness Lakes system. Sizes shown as minimum size (mm fork length), Maximum size (mm fork length) and the mean size (mm fork length) and number of fish sampled (N).

Species Name	EA	Min length (mm FL)	Max length (mm FL)	Mean length (mm FL)	N
Anguilla mossambica	Va	400	745	531.4	10
Galeichthys feliceps	IIb	143	186	157.6	56
Atherina breviceps	Ib	16	100	43.3	4336
Lichia amia	IIa	326	957	548	22
Oreochromis mossambicus	IV	9	476	81.7	5061
Gilchristella aestuaria	Ib	20	89	44.2	6081
Cyprinus carpio	IV	23	751	351.6	98
Elops machnata	IIa	-	572	-	1
Caffrogobius gilchristi	Ib	38	167	101.6	378
Glossogobius callidus	Ib	-	104	-	1
Psammogobius knysnaensis	Ib	15	100	430	629
Redigobius dewaali	Ib	28	40	34.2	5
Pomadasys commersonnii	IIa	390	790	505.1	78
Hyporhamphus capensis	Ia	26	220	121.8	2062
Monodactylus falciformis	IIa	13	228	182.4	449
Mugil cephalus	IIa	237	685	447.5	26
Myxus capensis	Vb	198	430	343.8	189
Liza dumerili	IIb	160	366	254.4	91
Liza richardsonii	IIc	78	483	314.8	413
Liza tricuspidens	IIb	332	406	378.8	5
juvenile Mugilidae	II	26	56	452	47
Gambusia affinis	IV	8	36	23.1	303
Argyrosomus japonicus	IIa	-	317	-	1
Solea bleekeri	IIa	77	141	97	7
Lithognathus lithognathus	IIa	123	746	382.4	116
Rhabdosargus holubi	IIa	22	420	202.3	180
Syngnathus acus	Ib	19	190	97.8	113

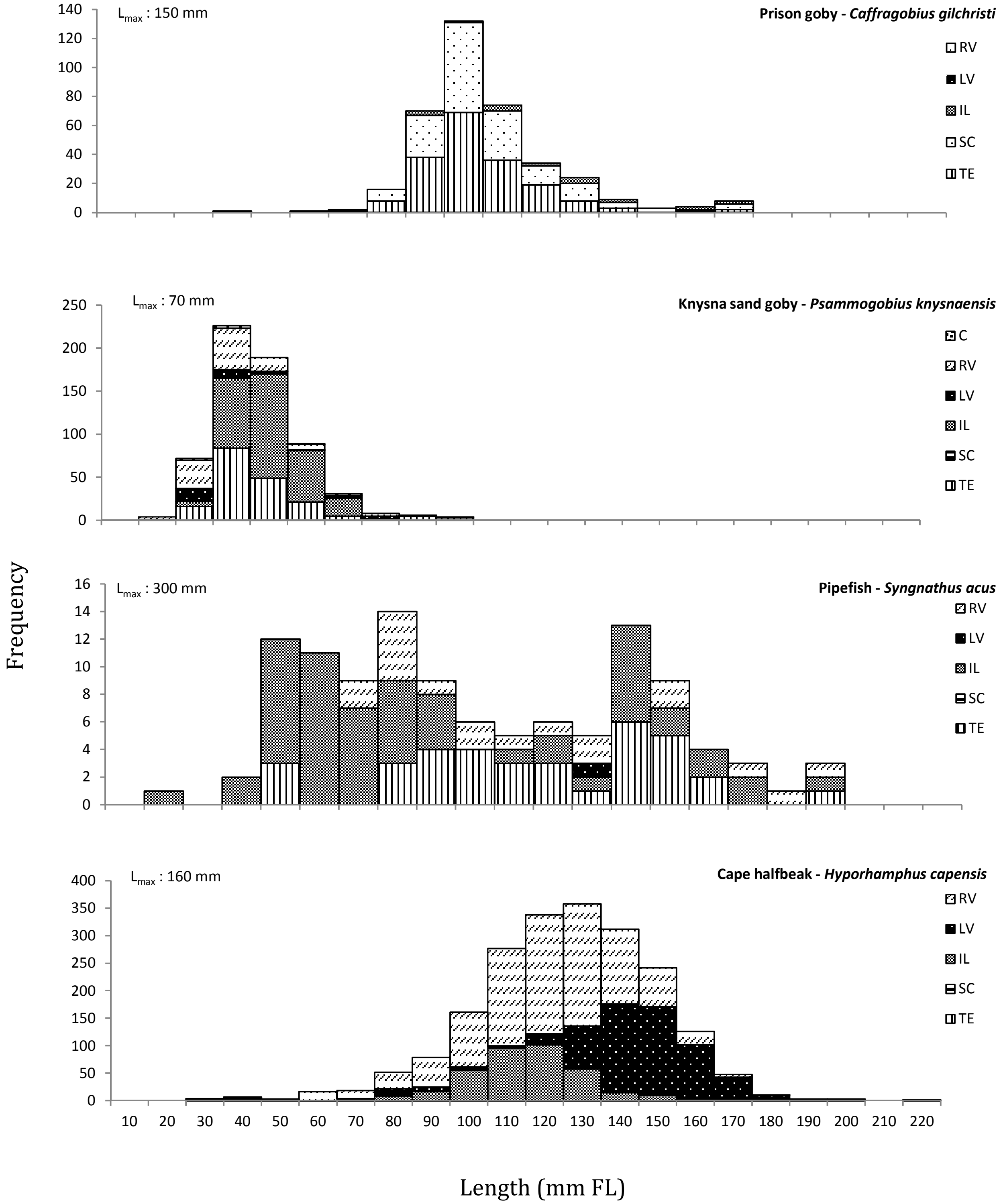


Figure 3.17. Length-frequency distribution of Whitfield's (1998) category I (Native Estuarine) species. Different colours represent different water bodies, TE = Touw Estuary, SC = Serpentine channel, IL = Island Lake, LV = Langvlei, C = Rondevlei-Langvlei channel and RV =Rondevlei. L_{max} is the maximum length recorded (Smith & Heemstra 1988).

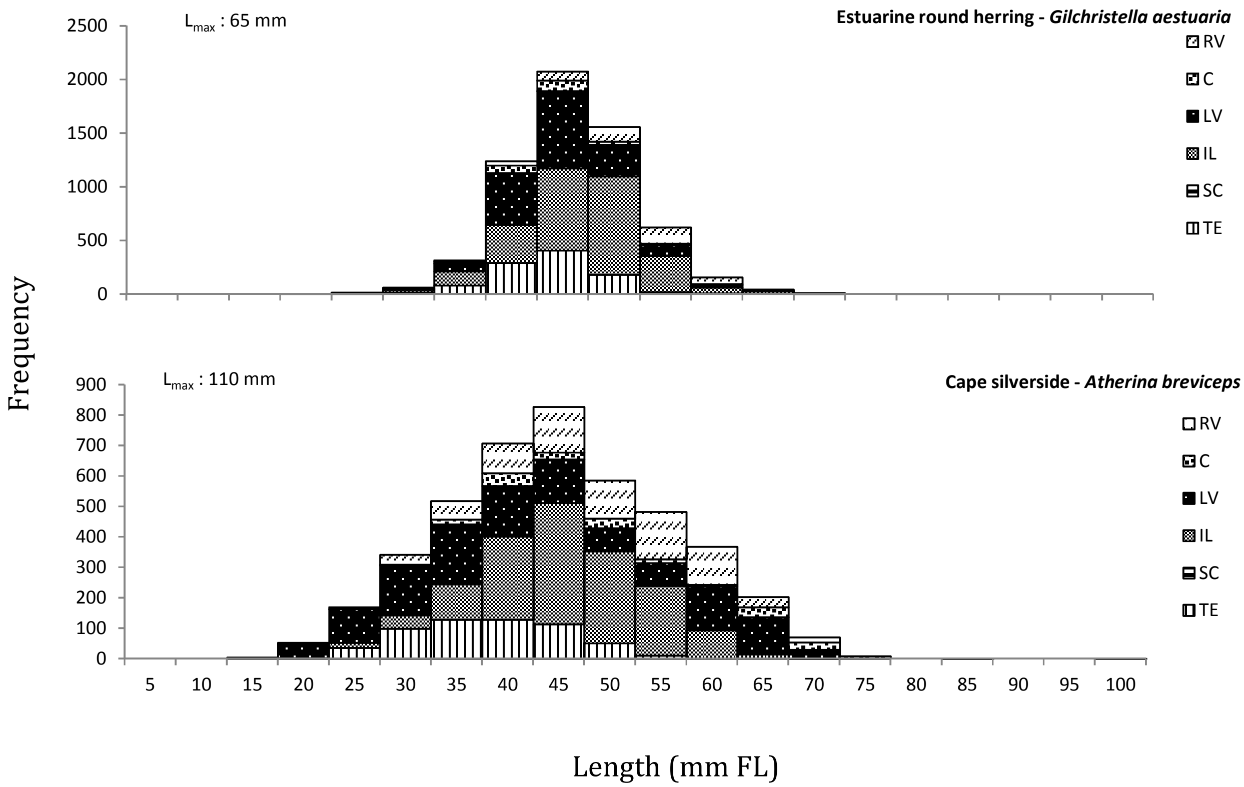


Figure 3.17. (Cont.) Length-frequency distribution of Whitfield's (1998) category I (Native Estuarine) species. Different colours represent different water bodies, TE = Touw Estuary, SC = Serpentine channel, IL = Island Lake, LV = Langvlei, C = Rondevlei-Langvlei channel and RV=Rondevlei. L_{max} is the maximum length recorded (Smith & Heemstra 1988).

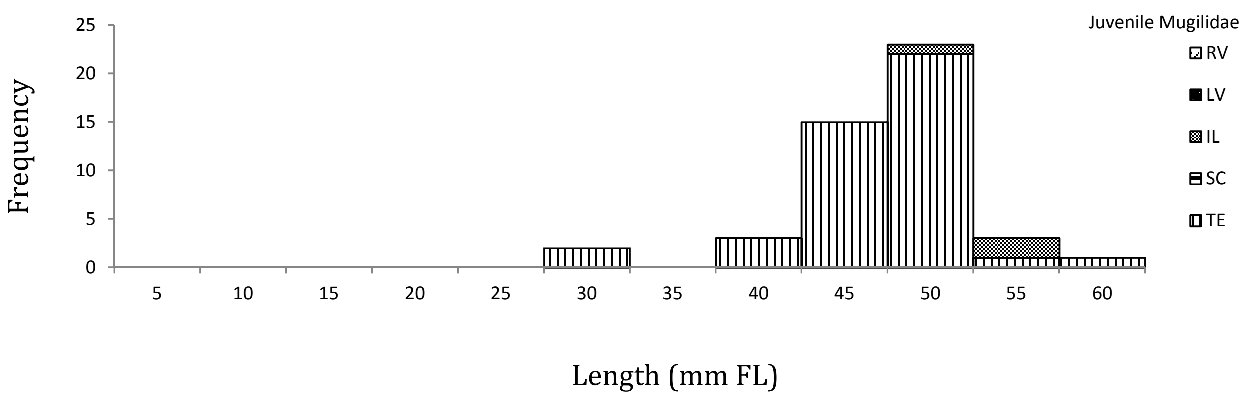


Figure 3.18. Length-frequency distribution of juvenile Mugilidae species which fall into Whitfield's (1998) category II (Euryhaline marine). Different colours represent different water bodies, TE = Touw Estuary, SC = Serpentine channel, IL = Island Lake, LV = Langvlei, and RV = Rondevlei.

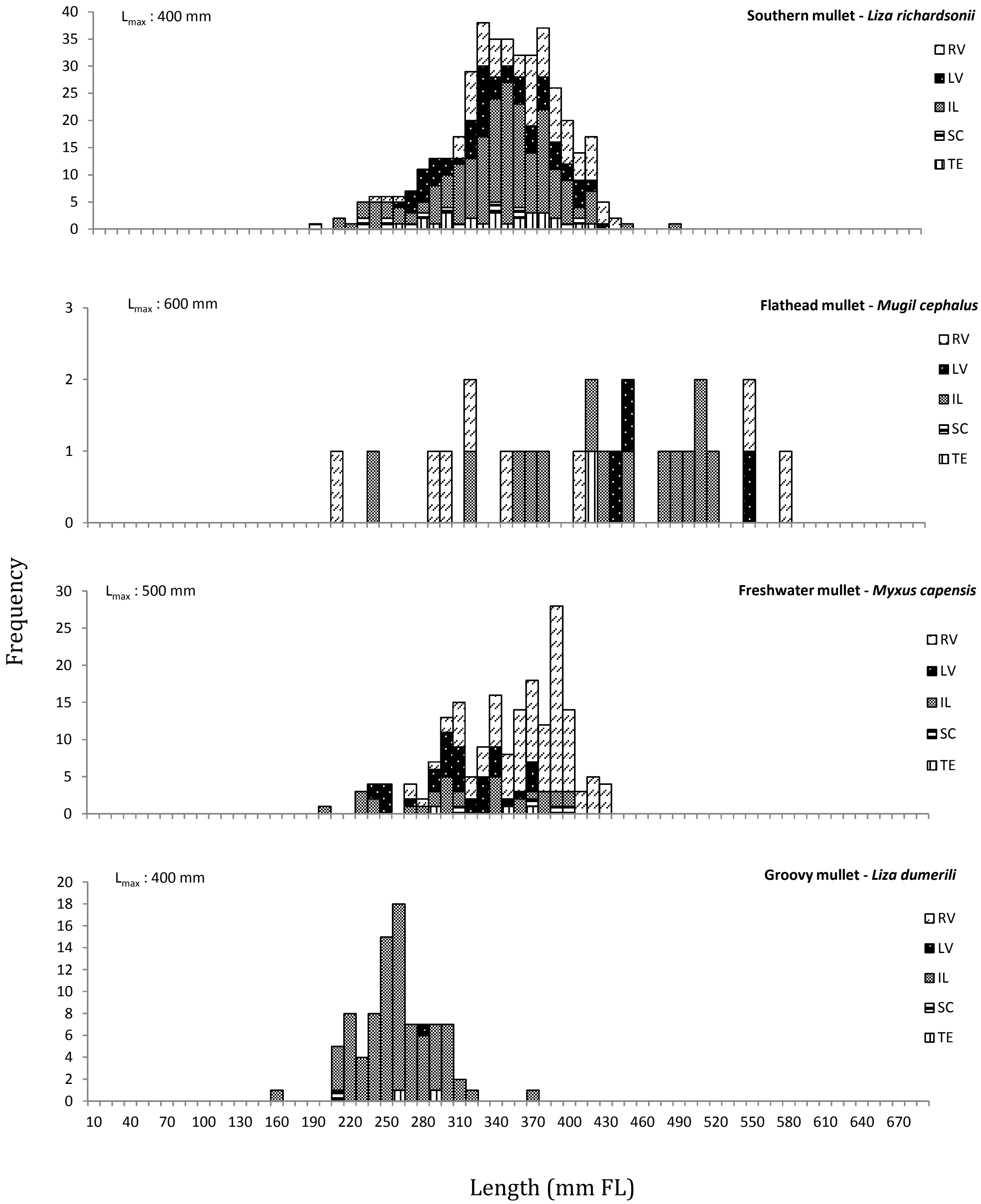


Figure 3.19. Length-frequency distribution of Mugilidae species which fall into Whitfield's (1998) category II (Euryhaline marine). Different colours represent different water bodies, TE = Touw Estuary, SC = Serpentine channel, IL = Island Lake, LV = Langvlei, and RV = Rondevlei. L_{max} is the maximum length recorded (Smith & Heemstra 1988).

Juvenile Mugilidae (30 - 60 mm FL) were sampled from the Touw Estuary and Island Lake only in early 2011 (**Fig. 3.18**). Four species of adult Mugilidae were sampled throughout the system, *L. richardsonii*, *L. dumerili*, *M. cephalus* and *M. capensis.* The length frequency was dominated by large fish with very few smaller (70 - 190mm FL) individuals being sampled (**Fig. 3.19**).

Six other marine spawning species (category II) were sampled in high enough numbers to allow for length-frequency assessments. *Monodactylus falciformis* was most abundant in the size classes between 180 - 230 mm FL and were sampled predominantly in Rondevlei and Langvlei, with smaller individuals (80 – 160 mm FL) being sampled in Touw Estuary and Island Lake (**Fig. 3.20**). One juvenile (13 mm FL) was sampled in early 2011 in the Touw Estuary. Size classes between 110 and 280 mm FL of *R. holubi* were sampled in Island Lake and Touw Estuary with large fish (300 – 380 mm FL) being sampled from Rondevlei. In early 2011, *R. holubi* juveniles (30 – 50 mm FL) were sampled from the Touw Estuary (**Fig. 3.20**).

The white sea catfish (*G. feliceps*) were only sampled in the lower Touw Estuary. All were juveniles between 150 and 190 mm FL. *Lithognathus lithognathus* were sampled from the Touw Estuary primarily between 260 and 380 mm FL and from Island Lake between 380 and 470 mm FL. Smaller fish (130 – 160 mm FL) were sampled from Island Lake in early 2011. Larger specimens (670 – 750 mm FL) were sampled from Rondevlei. No *L. lithognathus* were sampled from Langvlei (**Fig. 3.21**).

No *P. commersonnii* smaller than 390mm FL were sampled from the Wilderness Lakes system. *Pomadasys commersonnii* ranging from 390 - 650 mm FL were most common in Island Lake and one very large (790 mm FL) fish was caught in Rondevlei (**Fig. 3.21**).

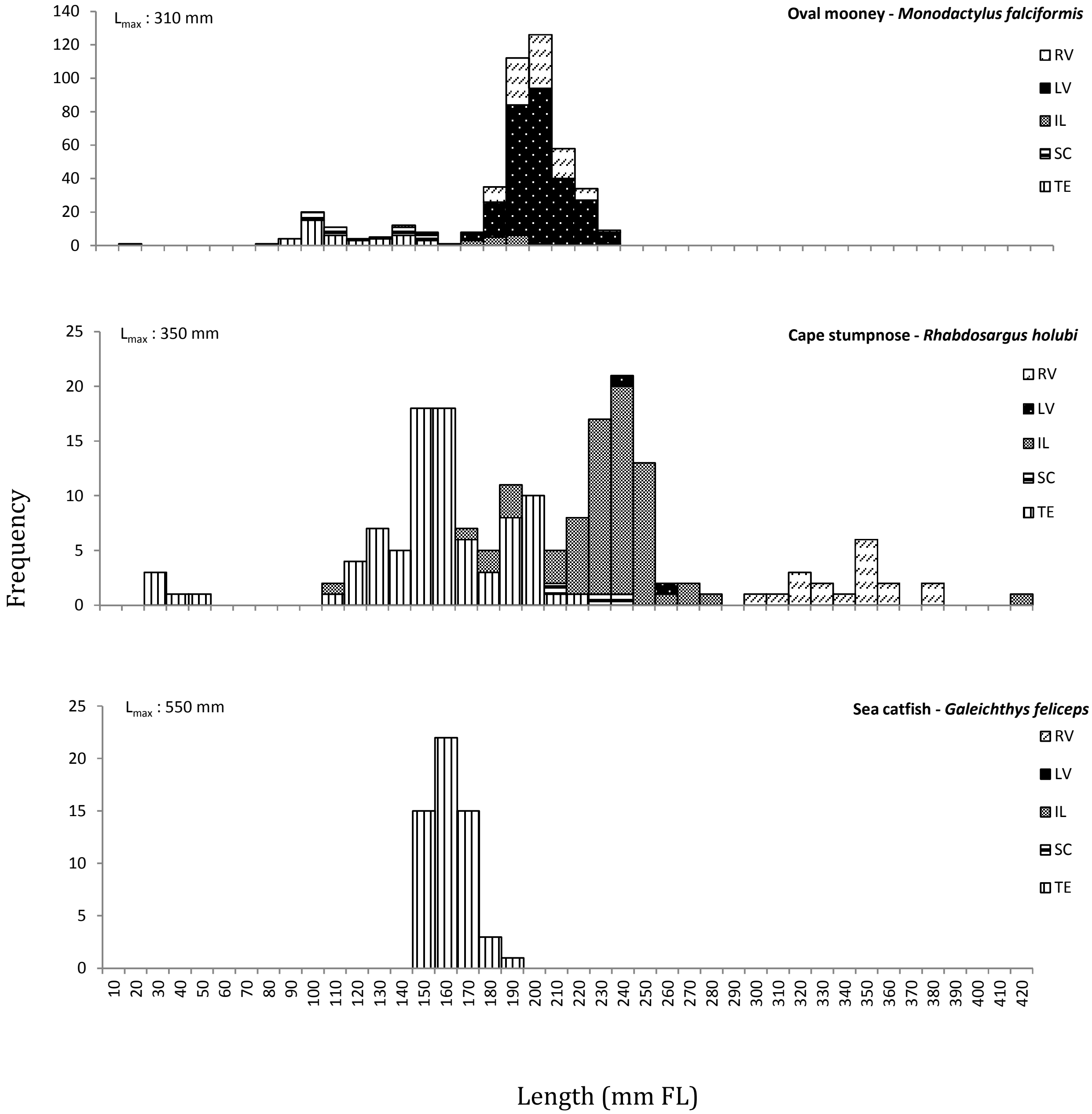


Figure 3.20. Length-frequency distribution of smaller species which fall into Whitfield's (1998) category II (Euryhaline marine) excluding Mugilidae family. Different colours represent different water bodies, TE = Touw Estuary, SC = Serpentine channel, IL = Island Lake, LV = Langvlei, and RV = Rondevlei. L_{max} is the maximum length recorded (Smith & Heemstra 1988).

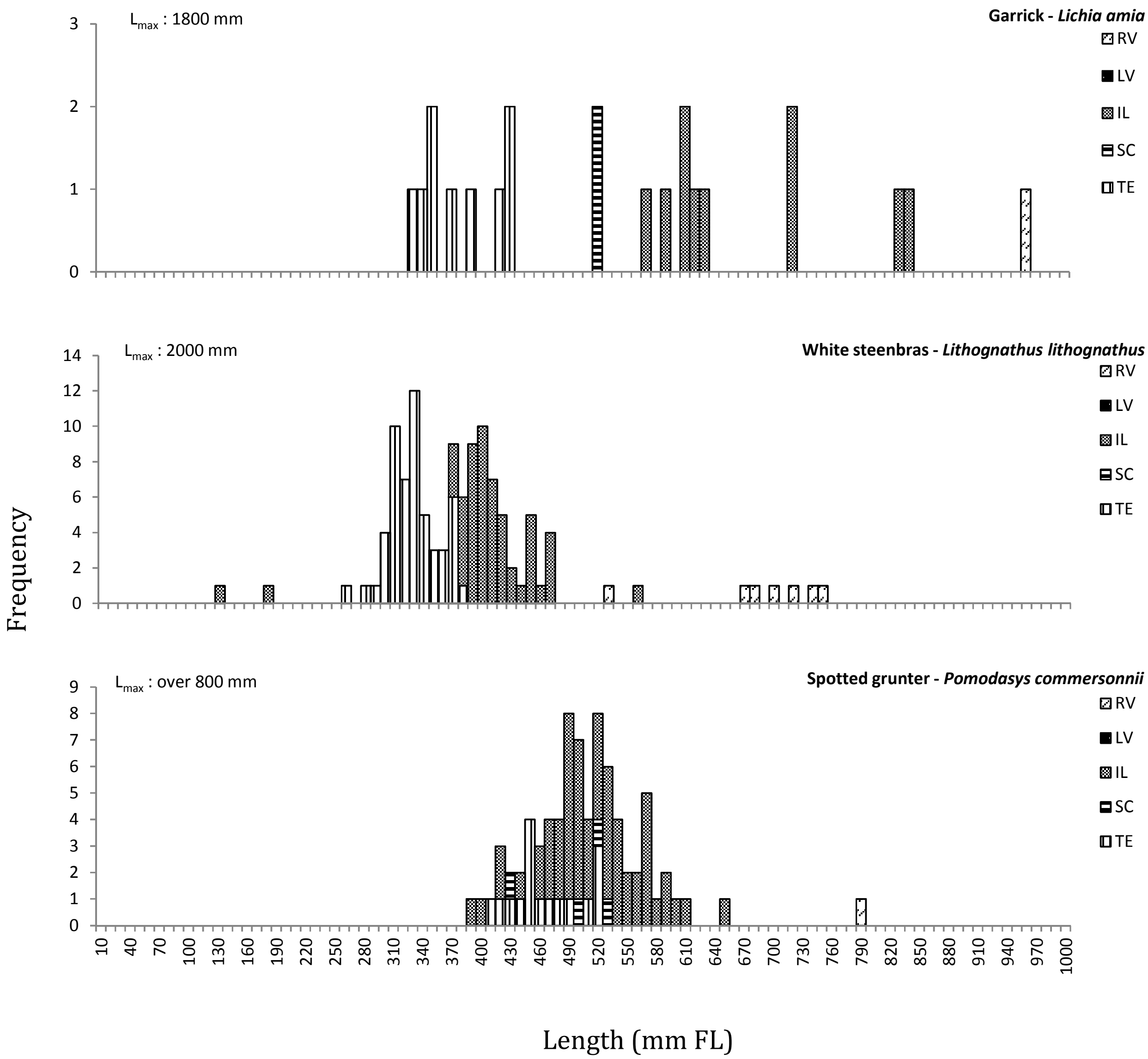


Figure 3.21. Length-frequency distribution of larger species which fall into Whitfield's (1998) category II (Euryhaline marine) excluding Mugilidae family. Different colours represent different water bodies, TE = Touw Estuary, SC = Serpentine channel, IL = Island Lake, LV = Langvlei, and RV = Rondevlei. L_{max} is the maximum length recorded (Smith & Heemstra 1988).

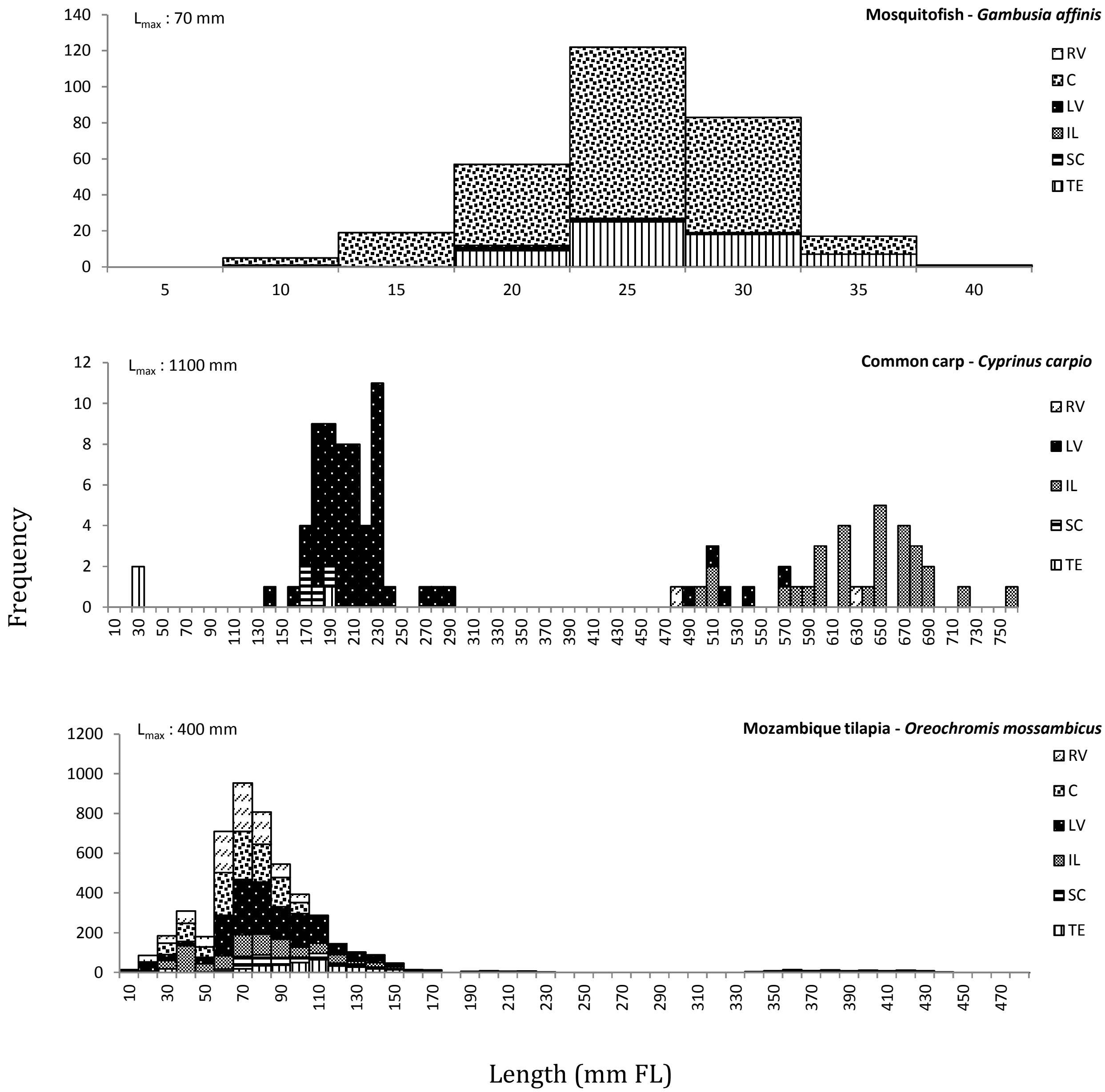
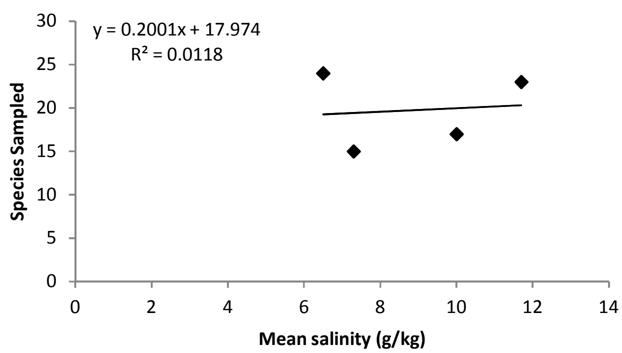


Figure 3.22. Length-frequency distribution of Whitfield's (1998) category IV (alien freshwater) species. Different colours represent different water bodies, TE = Touw Estuary, SC = Serpentine channel, IL = Island Lake, LV = Langvlei, C = Rondevlei-Langvlei channel and RV = Rondevlei. L_{max} is the maximum length recorded (Smith & Heemstra 1988; Froese & Pauly 1999).

A wide size range of *C. carpio* were sampled (**Fig. 3.22**). Larger fish were sampled in 30 m seine nets in Island Lake and smaller individuals were primarily sampled using fyke nets. *Oreochromis mossambicus* had a large number of fish (between 100 and 950 fish) in the smaller size classes (20 – 180 mm FL) (**Fig. 3.22**) which were sampled frequently in the fyke nets, while larger fish were sampled in the gill nets. *Gambusia affinis* were sampled from 8 mm SL to 40 mm SL with the highest abundance between 20 and 30 mm SL and the highest proportion of fish caught in the channel between Langvlei and Rondevlei (**Fig. 3.22**).

The relationship between the number of species sampled in each sampling area and the mean salinity and turbidity are shown in **Figure 3.23**. There was no relationship between salinity and the number of species sampled. There was no relationship (r^2 = 0.18, d.f. 1, 2, P = 0.57) between turbidity and the number of species sampled.

a)



b)

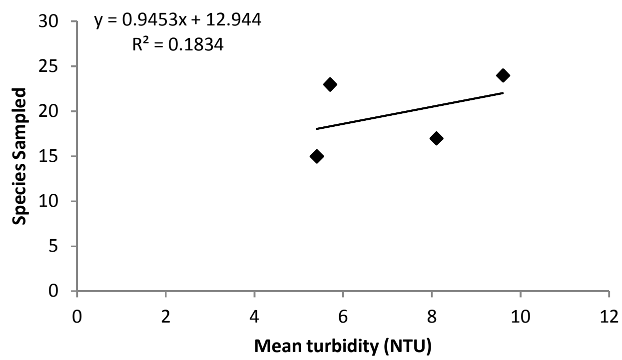


Figure 3.23. Relationship between the number of species sampled and a) the average salinity (g/kg) and b) turbidity (NTU) for each sampling area (Russell 1999a). TE = Touw Estuary, IL = Island Lake, LV = Langvlei and RV = Rondevlei.

DISCUSSION

The data collected in this study were highly variable with large variance temporally and spatially. As a result statistical assessment was difficult.

Species diversity

In common with most South African estuaries (Harrison & Whitfield 2006; James & Harrison 2008; James & Harrison 2010) the fish fauna of the Wilderness Lakes system is made up largely of a marine migratory component (13 species) with smaller proportions of native estuarine species (eight species), catadromous (two species) and alien freshwater (three species) species. Although overall, the fish community is made up predominantly of euryhaline marine species, the native estuarine fishes are numerically dominant with *A. breviceps* and *G. aestuaria* having the highest number of fish caught in the 10 m and 30 m seine nets. The numerical dominance by *A. breviceps* and *G. aestuaria* was also noted by Hall (1985), James & Harrison (2008) and Olds *et al*. (2011) and this is not uncommon in warm-temperate estuaries. Island Lake had the highest species diversity which is made up predominantly of euryhaline marine species, while Langvlei had the lowest species diversity. The low species diversity higher up the system was also noted by Hall (1985) with Langvlei and Rondevlei being categorized with low fish diversity. The fish community in these lakes can be divided into the gravimetrically dominant Mugilidae and Cichlidae, and the numerically dominant Atherinidae and Clupidae. The high diversity and occurrence of few juveniles (post mouth opening events) sampled within Island Lake and the Touw Estuary, confirms Hall's (1985) theory that these are the major nursery areas of the system.

Hall (1985) sampled 32 species throughout the system. The study by James & Harrison (2010) sampled 18 species from 11 families. While these latter workers sampled only the Touw Estuary and not the associated lakes, the species composition was very similar to that reported by Hall (1985) and that found in the current study. The species sampled by James & Harrison (2010) not sampled in this study was limited to *Heteromycteris capensis* (Cape sole). Hall (1985) sampled *Diplodus capensis capensis* (blacktail) and *Sarpa salpa* (strepie) in the Wilderness Lagoon, *Rhabdosargus sarba* (Natal stumpnose) in Island Lake and *Clinus superciliosus* (klipfish) in Rondevlei. None of these was sampled in this study. Species sampled by James & Harrison (2008) in Swartvlei that were not sampled in Touw Estuary include *D. capensis capensis*, *Diplodus hottentotus* (zebra), *C. superciliosus* and *S. salpa*. The species sampled by Hall (1985) not found in this study were all euryhaline marine species which could indicate limited recruitment into the Touw Estuary over the past 30 years. This could be attributed to the mouth opening state over this period (Chapter 2). In 2008 and 2009, the Touw Estuary mouth was open for 100% of each year, providing optimum recruitment for all marine spawning species. In 2010, the Touw Estuary mouth was recorded as open for a total of only 29 days: 10 in January, four in July and 15 in December. In 2011, the mouth was open for 13 days in January and 19 in May, after which the study concluded. These short open mouth periods provided very limited opportunities for juvenile marine species to move into the system. The periods of mouth opening coincided with major periods of recruitment into southern Cape estuaries for *R. holubi* (August to April), *L. lithognathus* (September to January) and *L. richardsonii* (all year) (Whitfield & Kok 1992), all of which recruited into the Wilderness Lakes system during the mouth opening periods. The opportunity for juvenile marine spawning species to move into the Wilderness Lakes system was limited in 2010 and early 2011. Species composition did not differ significantly from other temporary open/closed estuaries in the region. In each of the studies the euryhaline marine component dominated gravimetrically where the shoaling pelagic species dominated numerically (Vorwerk *et al*. 2001; James & Harrison 2008). This shows that the fish fauna in the Wilderness Lakes system is representative of temporary open/closed estuaries in the southern Cape.

Abundance and distribution

The distribution of species, but not the number of species sampled in the Wilderness Lakes system showed variation between the fresher and more saline regions in the system. This could be attributed to the temperature and salinity differences often seen in warm temperate estuaries (Harrison & Whitfield 2006; Childs *et al*. 2008). Harrison & Whitfield (2006) determined that the occurrence and abundance of fish fauna in South African estuaries are linked to temperature and salinity. The relationship between these factors is highly important in structuring fish assemblages.

Glossogobius callidus, *Anguilla mossambica* and *R. dewaali* were only sampled in the more fresh water areas between the Touw Estuary and Island Lake. The euryhaline marine and some native estuarine species which have wider environmental tolerances (Whitfield *et al*. 1981) were sampled throughout the system. The distribution of demersal fishes in the Kariega estuary was found to be linked to sediment type, with *G. callidus* being found predominantly in the middle and upper reaches of the estuary (Bennett & Branch 1990; Richardson *et al*. 2006). In these areas, the water was more turbid due to the muddier sediment. *Psammogobius knysnaensis* was found in the lower and mouth regions and this was attributed to its feeding behaviour requiring less turbid water (Bennett & Branch 1990; Richardson *et al*. 2006). Non-demersal species, such as members of the family Sparidae, Cichlidae, Clupidae and Atherinidae, which are less directly associated with sediment type were found throughout the system and are tolerant of wide fluctuations in turbidity, salinity and temperature (Richardson *et al*. 2006).

The native estuarine species (*A. breviceps* and *G. aestuaria*) were the most abundant throughout the system but were sampled only in the littoral zones using seine nets. Becker *et al*. (2011) found *A. breviceps* and *G. aestuaria* to be most abundant in low turbidity areas, which would make the formation of shoals easier. However, in the Wilderness Lakes system, Island Lake, which had the highest turbidity (9.6 NTU), also had the highest CPUE of both *A. breviceps* and *G. aestuaria* in the 30m seine net (1309 ± 5335 fish.net haul^{-1} and 1006 ± 1213 fish.net haul^{-1} respectively). The Touw Estuary and Langvlei exhibited the lowest turbidity (5.7 and 5.4 NTU respectively), but there was no correlation (r^2 = 0.18, d.f. 1, 2, P = 0.57) between the number of species in each sampling area and to the turbidity in that area (**Fig. 3. 21 b**). There appeared to be no effect of turbidity on the distribution of the shoaling pelagic species in the Wilderness Lakes system. Five Mugilid species (*M. cephalus*, *M. capensis*, *L. dumerili*, *L. richardsonii* and *L. tricuspidens*) were sampled in the system. *Liza richardsonii* was the most abundant Mugilid species in Touw Estuary, Langvlei and Island Lake and *M. capensis* was the most abundant Mugilidae species in Rondevlei. The high abundance of *M. capensis* higher up the system is expected as they are well known for long range migrations to the upper reaches of estuaries to spawn (Beckley 1984; Whitfield 1990). Bok (1979) also suggested that competition between Mugilid species may account for the segregation of species in estuarine systems.

Lithognathus lithognathus, L. amia and *P. commersonnii* were the most abundant in Island Lake and in the Touw Estuary (**Table 3.6**) and low numbers of very large fish were sampled from Rondevlei (**Table 3.6**). This decreasing abundance but increase in size was also reported by Hall (1985). The smaller fish found in Island Lake suggest that this lake serves as the system's initial nursery ground for the marine spawned species (Hall 1985). It is possible that the salinity gradient in the Wilderness Lakes may influence the fish's natural instinct to move towards high salinity to spawn. As salinity increases towards Rondevlei (Russell 1999a), this may confuse fish and if water levels are high enough they may move towards Rondevlei rather than down the salinity gradient towards the sea (Hall 1985; Russell 1999a). Once these fish reach Rondevlei it appears that they remain there, which accounts for the presence of very large euryhaline marine fishes in Rondevlei. Langvlei had the lowest number of euryhaline marine species and it appears that these species either stay in Island Lake or move through to Rondevlei. With few euryhaline marine species being sampled by both Hall (1985) and this study, it may be suggested that Langvlei acts as an intermediary or transition zone between the fresher Island Lake and more saline Rondevlei.

The longitudinal distribution of species within estuaries is indicative of the niche occupied by each species as well as the differing physico-chemical conditions along the estuary (Richardson *et al*. 2006). Physical and chemical conditions may determine the areas in which species will be sampled and their abundance in the catch composition in certain areas of the estuary (Ter Morshuizen & Whitfield 1994; Vorwerk *et al*. 2001). Although Vorwerk *et al*. (2001) found no clear evidence of an overall longitudinal fish distribution pattern in different warm-temperate estuaries on the south-eastern Cape coast, on an individual species basis there were noticeable trends. *Atherina breviceps* are described as being generally more abundant in the lower, more saline reaches of estuaries and *G. aestuaria* tended to be more abundant further upstream where the salinity was lower (Vorwerk *et al*. 2001). The opposite trend was seen in the 30m seine net catches in the Wilderness Lakes system, where *A. breviceps* and *G. aestuaria* were most abundant in Island Lake and least abundant up the system towards Rondevlei. While *A. breviceps* was more abundant than *G. aestuaria* in the Touw Estuary and Island Lake, the opposite was seen in Rondevlei and Langvlei (**Appendix 2**).

Mouth opening events are important in structuring fish communities and are strongly linked to environmental fluctuations such as timing, duration and frequency of mouth opening (Wallace & van der Elst 1975; Kok & Whitfield 1986; Russell 1996; James 2006). Migration to sea to spawn, and recruitment into estuaries, is highly dependent on mouth opening states and times which need to coincide with marine spawning events. Extended periods of mouth openings maximise the opportunities for fish to migrate between the ocean and estuaries. The lack of individuals sampled during this study can be related to the Touw Estuary mouth state. Between January 2010 and May 2011 the Southern Cape experienced a severe drought during which the Touw Estuary mouth remained closed for 90% of the time (Chapter 2) resulting in almost no recruitment into the Wilderness Lakes and no migration out to sea. After this extended closed period, the mouth was open for a 28 day open period in December 2010. After this, juvenile *R. holubi*, *L. lithognathus* and Mugilidae were sampled during the summer sampling period in the Touw Estuary and from Island Lake (**Fig 3.17; 3.18**).

Russell (1996) investigated factors affecting fish abundance in the Wilderness Lakes system and the Swartvlei Estuary and determined that no single environmental factor is responsible for changes in fish abundance in these systems. The species composition in the Wilderness Lakes system is representative of fish assemblages in other estuarine systems in the southern Cape, however their distribution in the system may differ from other systems due to the additional three lakes joined to the Touw Estuary. The distribution of fish species did exhibit a slight gradient in the system, with more freshwater species (*C. gilchristi*, *G. callidus*, *A. mossambica* and *R. dewaali*) only being sampled as high as Island Lake. The majority of the native estuarine species (*A. breviceps*, *G. aestuaria*, *P. knysnaensis* and *S. acus*) were sampled throughout the Wilderness Lakes system. A large proportion of the euryhaline marine species were sampled throughout the Wilderness Lakes system however, the more marine dominant species (*A. japonicus* and *E. machnata*) that are not dependent on estuaries were only sampled in the lower reaches of the Wilderness Lakes system.

Presence of alien species

As revealed in a pilot study by Olds *et al*. (2011), the freshwater aliens *O. mossambicus* and *G. affinis* were widely distributed and highly abundant throughout the Wilderness Lakes system. *Cyprinus carpio* were sampled in each of the lakes and the Touw Estuary but their abundances were low (n = 15) and they were caught predominantly with the 30m seine nets. *Micropterus salmoides* were not sampled in the Wilderness Lakes system during the pilot or this study besides from one sighting of two adult fish in Langvlei and they have been sampled in previous studies. The abundance, distribution and establishment success of the alien fishes is addressed in Chapter 4.

Conclusion

The fish species composition in the Wilderness Lakes system is typical of temporary open/closed estuaries in the warm-temperate biogeographic region of South Africa. The species diversity is higher than that in four other temporary open/closed estuaries, but the number of estuarine associated species does not differ significantly different from those in other estuaries. The Wilderness Lakes system is therefore typical of a temporary open/closed estuary with the same or higher species richness than is found in many other estuaries. The Wilderness Lakes system has the highest species diversity closer to the Touw Estuary mouth, with diversity decreasing higher up the system. The species composition shows some form of a gradient from freshwater to more marine species higher up the system, with common native estuarine species being abundant throughout the system. Four alien freshwater species are now reported from the system, two of which are abundant. The establishment success of the alien fishes is assessed in Chapter 4.

CHAPTER 4

THE ESTABLISHMENT SUCCESS OF ALIEN INVASIVE FISH IN THE WILDERNESS LAKES SYSTEM

INTRODUCTION

The process of species invasion can be divided into three stages: introduction, establishment and dispersal (Rosecchi *et al*. 2001). The success of each stage is determined both by environmental and biotic factors (Moyle & Light 1996; Rosecchi *et al*. 2001). While introductions of invasive species to new systems may occur quite frequently, establishment is not guaranteed. Where invasive species fail to become established despite repeated introductions, that failure is attributed to their inability to adapt to those systems and establish a successful breeding population (Moyle & Light 1996). In order to classify an introduced species as established, there must be evidence of reproduction and the existence of a self-sustaining population (Gozlan *et al*. 2010).

The biological characteristics of invasive species have been thought to play a large contributing role in establishment success (Moyle & Marchetti 2006). Ultimately success of a species in new environment depends on the suitability of its reproductive style to the environment, its ability to meet nutritional requirements and its ability to adapt to current abiotic conditions (Weyl *et al*. 2009). The biological characteristics of invasive species most often cited as associated with successful fish invasions are: high abundance and wide distribution in the native range, high physiological tolerances and an r-selected or opportunistic life history strategy (Winemiller & Rose 1992), rapid dispersal and generalist diet or habitat preferences (Moyle & Marchetti 2006).

Meta analysis for establishment

In an assessment of establishment, it is important to understand biological invasion as a process of overcoming barriers (Copp *et al*. 2005; Richardson *et al*. 2011). Introduction means that the fish species has overcome the geographical barrier of entering the system (**Fig 4.1**). Many introduced species survive as casuals; such taxa can reproduce sexually, but fail to maintain their populations over longer periods (Copp *et al.* 2005; Richardson *et al.* 2011). Casuals therefore must rely on repeated introduction for their persistence. Establishment only starts when environmental barriers (barrier B) (**Fig. 4.1**) do not prevent individuals from surviving and when various barriers to regular reproduction (barrier C) are overcome (**Fig. 4.1**). Invasion, i.e. spread into areas away from sites of introduction, requires that introduced fish also overcome barriers to dispersal within the new region (barrier D) and can cope with the abiotic environment and biota in the general area (barrier E) (**Fig. 4.1**). Therefore, a species can be considered successfully established after overcoming barriers A, B, C and D. At this stage populations are sufficiently large that the probability of extinction due to environmental unpredictability is low. Many then invade disturbed, semi natural communities. Invasion of stable, undisturbed communities usually requires that the alien species overcomes resistance posed by a different category of factors such as climatic factors (barrier F) (Richardson *et al.* 2011; Copp *et al.* 2005) (**Fig. 4.1**).

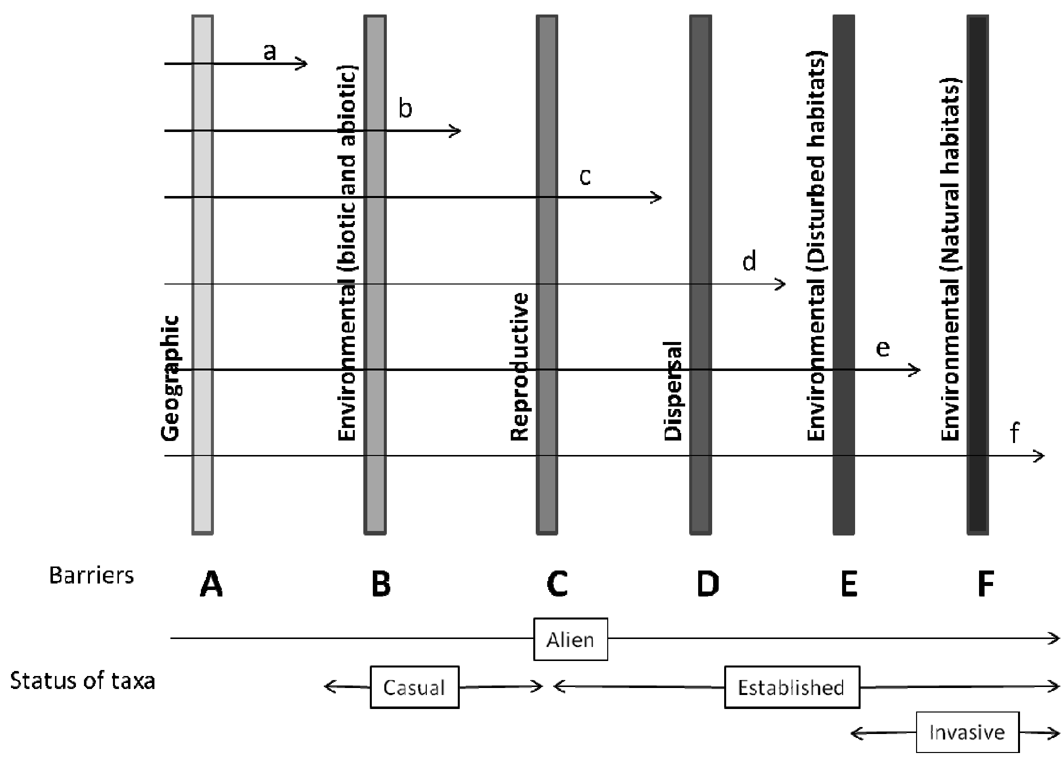


Figure 4.1. A schematic representation of the major barriers that limit dispersal of introduced fishes: (A) geographical; (B) environmental (abiotic and biotic factors in introduction area); (C) reproductive; (D) local or regional dispersal barriers; (E) environmental barriers in non-natural systems and (F) environmental barriers in natural systems. Pathways followed by taxa across barriers (from introduced to invasive) in natural systems are indicated by arrows, and these are reversible (Adapted from Richardson *et al.* 2011).

In this chapter, the establishment success of the four alien fishes, *O. mossambicus, G. affinis*, *C. carpio* and *M. salmoides* is investigated. The basis for determining whether a species is established is to test whether the species: (1) is widely distributed throughout the system; (2) comprises a large proportion of the fish fauna; and (3) has established a self sustaining population.

METHODS AND MATERIALS

Field sampling and analysis

Introduction pathways

To determine the potential introduction pathways and source populations, interviews with farm owners in the Wilderness Lakes system catchment were conducted. Semi-structured interviews were conducted followed by an open discussion on any additional information they thought would be important to the study. Two key conservation officials were interviewed to determine potential dates and corroborate potential sources of introduction. Electro fishing at 19 freshwater sites in the Duiwe River was conducted on a single occasion to determine presence/absence of invasive species.

Assessment criteria

Where possible, four assessment criteria were used as shown in **Table 4.1**. The abundance and distribution, size classes, spawning success and introduction pathways were assessed for each species to determine their establishment success.

Table 4.1. Assessment criteria and source of data for each of the invasive fish species in the Wilderness Lakes system, Western Cape.

Species	Assessment	Data source
Oreochromis mossambicus	• Abundance and distribution • Size classes • Spawning • Introduction pathways	- Fyke net, seine net and gill net sampling - Length data from sampling - Length data from sampling - Interviews with farmers, officials, literature
Cyprinus carpio	• Abundance and distribution • Size classes • Spawning • Introduction pathways	- Seine net, fyke net, gill net and angling - Length data from fish caught - Gonad development - biological examination, length-frequency distribution - Interviews with farmers, officials, literature
Gambusia affinis	• Abundance and distribution • Size classes • Spawning • Introduction pathways	- Once off scoop netting - Length data from scoop netting - Length data from scoop netting - Literature reviews
Micropterus salmoides	• Abundance and distribution • Size classes • Spawning • Introduction pathways	- Angling and electro fishing - Length data from fish caught - No data collected/available - Interviews with farmers, officials, literature

Distribution

Distribution data were collected using point data from abundance sampling. The data were collected from fyke net, gill net, 10m and 30m seine net and scoop net sampling as described in Chapter 2. A GPS point was taken at each site where an alien species was sampled (**Table 4.1**).

CPUE/Abundance

Abundance data, collected as CPUE, were collected by fyke nets, 10m and 30m seine nets, gill nets (sampling technique described in Chapter 2) and scoop nets. Additional data were collected from anglers targeting *C. carpio* and *M. salmoides* in Island Lake (**Table 4.1**). Fish sizes were estimated by converting the weight data using the length/weight relationship for each species (**Appendix 1**).

Maturity

Published literature on the age and growth of *O. mossambicus*, *C. carpio* and *M. salmoides* was consulted to obtain an estimate of the length at maturity to determine the maturity of sampled fish. Biological data taken from *O. mossambicus* and *C. carpio* were used to substantiate the literature reviews to determine length at maturity (**Table 4.1**). An independent assessment of the reproductive biology of *G. affinis* in the Wilderness Lakes ran concurrently with this study and the sex and maturity data were obtained from those results.

Statistical analysis

Distribution

The GPS points were taken at each site where an invasive species was sampled. These data points were used to plot distribution maps for each of the species in the GIS package from ESRI, ArcMap 10.

CPUE/Abundance

The CPUE of *O. mossambicus* in each sampling area for fyke net and gill net catches was correlated using a linear regression to the number species sampled in each sampling area and the Shannon-Weiner diversity index for each sampling area. The abundance of *G. affinis* in each sampling area was plotted on a box-and-whisker plot and a non-parametric Kruskal-Wallis

ANOVA (H) was used to determine the differences between the abundances in each sampling area, as assumptions for normality and homogeneity of the variance were not met.

Maturity

Length comparisons with maturity were made for each alien species. Length and age at maturity data were collected from publications on these species from Australia, Spain and Africa to determine an estimate of length at maturity.

Oreochromis mossambicus

The size at maturity of O. *mossambicus* is very variable between systems and is dependent on environmental factors linked to food and temperature (Weyl & Hecht 1999). For this reason, a range of maturities for different systems was used to determine the most accurate length at maturity. This was substantiated with biological data obtained from fish caught in the Wilderness Lakes system during this study (**Table 4.2**).

The mean length at sexual maturity for *O. mossambicus* was determined from 70 female and 74 male *O. mossambicus* collected during four months during 2010/2011 using the criteria described in **Table 4.3**. The proportion (P) of the sexually mature (developing, ripe and spent) individuals by length (L) was fitted to the logistic curve:

$$P = 1/\ (1 + \exp\ [-r\ (L - Lm_{50})])$$

where r is the slope of the curve and Lm_{50} is the mean length at sexual maturity.

Table 4.2. Length (L_{mat}) and age (A_{mat}) at maturity of *Oreochromis mossambicus* based on previous studies in Africa and Australia.1 - Booth & Khumalo (2009), 2 - Weyl & Hecht (1998), 3 - Bruton & Boltt (1975), 4 - James & Bruton (1992) and 5 - Arthington & Milton (1986)

		Male		Female		
Locality	GPS	L_{mat}	A_{mat}	L_{mat}	A_{mat}	Average
Mnjoli Dam, Swaziland[1]	26° 09' S 33° 16'E	205 mm TL	1 +	134 mm TL	0 +	170 mm TL
Lake Chicamba, Mozambique[2]	19° 08' S 33° 08' E	251 mm TL	3 +	223 mm TL	2 +	237 mm TL
Lake Sibaya, South Africa[3]	27° 25' S 32° 40' E	104 mm SL	1 +	69 mm SL	1 +	87 mm SL
Kowie Lagoon, South Africa[4]	26° 53' E 33° 36' S	223 mm SL	3 +	212 mm SL	3 +	218 mm SL
Rufanes Pool, South Africa[4]	Near 26° 53' E 33° 36' S	110 mm SL	1 +	118 mm SL	1 +	114 mm SL
Bradshaw's Mill Dam, South Africa[4]	Near 26° 53' E 33° 36' S	168 mm SL	2 +	186 mm SL	2 +	177 mm SL
Mill Farm Dam, South Africa[4]	Near 26° 53' E 33° 36' S	265 mm SL	2 +	263 mm SL	2-3 +	264 mm SL
Tingalapa Reservoir, Australia[5]	153° 10' E 27° 32'S	152 mm SL	1 +	180 mm SL	1 +	166 mm SL
North Pine Dam, Australia[5]	152° 55' E 27° 16'S	191 mm SL	1 +	174 mm SL	1 +	183 mm SL

Table 4.3. Descriptive criteria used to stage male and female *Oreochromis mossambicus* gonad development (from Weyl & Hecht 1998).

Stage of development	Description
Juvenile	Gonads not fully formed but present as two transparent threads of tissue. Sex not distinguishable macroscopically.
Resting	Ovary white or slightly yellowish. Oocytes macroscopically distinguishable. Testis distinguishable as small white strands.
Developing	Ovary enlarged oocytes readily visible and yellow. Testis broadened, distended and cream in colour.
Ripe	Oocytes of maximum size (2.5 – 3.4 mm along the long axis), readily extruded from female under abdominal pressure. Testis swollen to maximum size.
Spent	Ovary partly empty and flaccid with irregular oocyte size. Testis flaccid

Cyprinus carpio

The low number of *C. carpio* sampled in the Wilderness Lakes system did not allow for an age length investigation, for this reason length data from literature on the age and growth of *C. carpio* was used (**Table 4.5**). Fish sampled from the Wilderness Lakes system were sexed and staged using the criteria shown in **Table 4.4**.

Table 4.4. Descriptive criteria used to stage male and female *Cyprinus carpio* gonad development (from Winker *et al*. 2011).

Stage of development	Description
Juvenile	Not possible to visibly distinguish the sex. Gonads appear as translucent thin strips.
Resting	Ovary increased in size and translucent. Testis visible as a white and straight strip.
Developing	Enlarged ovary becomes opaque and is orange–red. Oocytes are visible. Testis increased in size and becomes lobular in shape.
Ripe	Ovaries turgid with oocytes filling the entire abdominal cavity. Oocytes are olive-green in colour. Blood capillaries are abundant. Testes creamy white, enlarged and fill more than a third of the body cavity. Sperm can be extruded from testes.
Spent	Ovary and testis flaccid and reddish in appearance. Ovary occasionally with a few vitellogenic oocytes present.

The study site nearest to the Wilderness Lakes system was Lake Gariep, Free State, where the average size of mature *C. carpio* was 316 mm FL (Winker *et al*. 2011) (**Table 4.5**). The smallest male caught measured 473 mm FL and was ripe (stage four) (**Fig. 4.12**) and the smallest ripe female measured 504 mm FL (**Fig. 4.12**). All but three fish (juveniles) sampled were larger than 316 mm FL and were therefore classified as mature adults.

Table 4.5. Length (L_{mat}) and age (A_{mat}) at maturity of *Cyprinus carpio* based on previous studies in Africa and Australia. 1 - Britton *et al*. (2007), 2 - Brown *et al*. (2005), 3 - Winker *et al*. (2011).

		Male		Female		
Locality	GPS	L_{mat}	A_{mat}	L_{mat}	A_{mat}	Average
Lake Naivasha, Kenya[1]	0° 46' S 36° 21' E	340 mm FL	1 +	420 mm FL	2 +	380 mm FL
Barmah Forest, Australia[2]	36° 00'S 145° 00'E	307 mm FL	2+	328 mm FL	2+	316 mm FL
Campaspe channels, Australia[2]	35° 17' S 149° 07' E	273 mm FL	2 +	287 mm FL	2 +	280 mm FL
Lake Gariep, South Africa[3]	34° 41' S 25° 40' E	296 mm FL	2 +	335 mm FL	2 +	316 mm FL

Gambusia affinis

The age and growth of *G. affinis* in the Wilderness Lakes system was investigated by Sloterdijk (2011). He established that mature females ranged from 14 mm TL to 38.5 mm TL, mature males ranged from 7.9 - 30.4 mm TL and immature fish measured between 5.05 and 13.95 mm TL. These lengths were used to show the population structure in the Wilderness Lakes system with fish measuring more than 8 mm TL being classified as adults.

Micropterus salmoides

The lack of sampling of *M. salmoides* did not allow any investigation into the maturity of these fish in the Wilderness Lakes system. Studies on the age and growth of *M. salmoides* in Eastern Cape impoundments have shown the length at maturity for *M. salmoides* to be 260 mm FL and 250 mm FL for male and female fish respectively. Fish sampled measuring over 255 mm FL would thus be classified as mature (**Table 4.6**).

Table 4.6. Length (L_{mat}) and age (A_{mat}) at maturity of *Micropterus salmoides* based on previous studies in Africa and Spain. 1 - Weyl & Hecht (1999), 2 - Rodriguez-Sánchez *et al*. (2009), 3 – Taylor (pers. comm. Department of Ichthyology and Fisheries Science, Rhodes University)

		Male		Female		
Locality	GPS	L_{mat}	A_{mat}	L_{mat}	A_{mat}	Average
Lake Chicamba, Mozambique[1]	19° 08'S 33° 08'E	305 mm FL	0+	290 mm FL	0+	298 mm FL
Primera de Palos ' lake, Spain[2]	37° 15'N 6° 56'W	182 mm TL	1+	212 mm FL	2+	197 mm TL
Wriggleswade Dam, Eastern Cape[3]	32° 35'S 27°33'E	260 mm FL	1+	250 mm FL	1+	255 mm FL

RESULTS

Due to differences in habitats occupied, breeding habits and sizes of the alien invasive fish species, the establishment of each of the invasive species is reported separately.

Oreochromis mossambicus establishment

Introduction pathways

The date and initial introduction into the Wilderness Lakes system is unknown but they were stocked into the nearby Groenvlei by nature conservation in 1976 (J. Huisamen, Ecological coordinator – CapeNature, pers. comm.). *Oreochromis mossambicus* were first sampled in the Wilderness Lakes system in 1983 by Hall (1985) where they were sampled in each of the lakes and the Touw Estuary (**Table 4.7**). The most likely introduction pathway is from farm dams in the catchment which flow into the Duiwe River and Langvlei Spruit. Their current distribution has remained constant to all the lakes, interconnecting channels and the Touw Estuary. *Oreochromis mossambicus* was not sampled in the Duiwe or Touw Rivers during the freshwater survey conducted by Russell (1999) nor during the electro fishing survey conducted in the Duiwe River in 2011 (**Fig. 4.3**). *Oreochromis mossambicus* therefore appears to be limited to estuarine parts of the system. It was most likely introduced into the Wilderness Lakes system between 1976 and 1983.

Table 4.7. History of *Oreochromis mossambicus* in the Wilderness Lakes system, Western Cape.

Date	Phase		Source
1976	Introduction	Fish stocked into Groenvlei to combat excessive weed growth in lake and dams in the catchment	Johan Huisamen (ecological coordinator, CapeNature)
1983	Dispersal	First record of species in Wilderness Lakes system, already widespread and abundant	Hall 1985
2009	Establishment	Fish sampled in fyke nets throughout Wilderness Lakes system	Rhodes University, SANParks
2010-2011	Establishment	Species widely distributed and highly abundant in system and established breeding population	This study

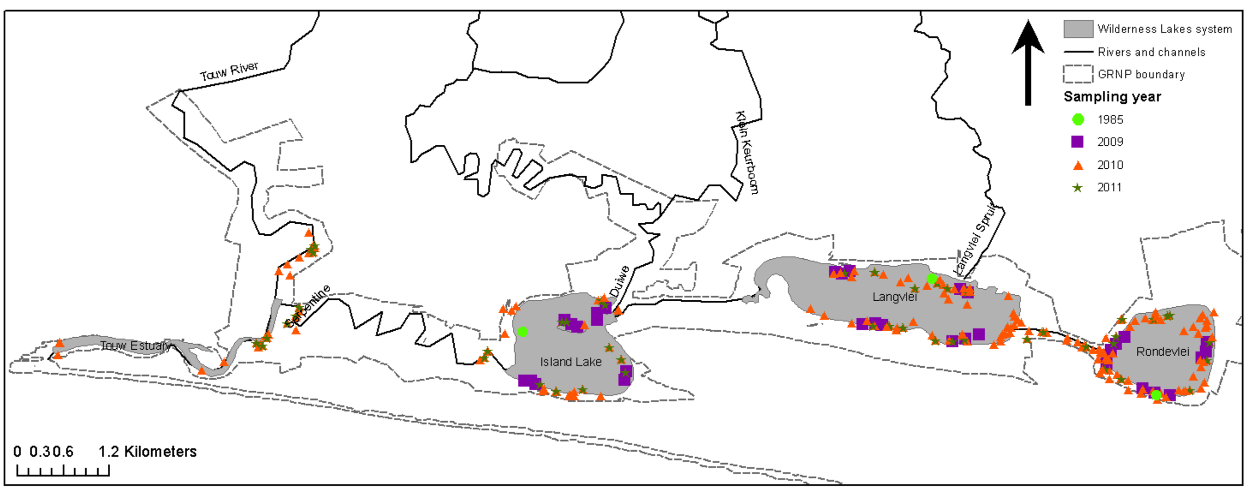


Figure 4.3. The historic (Hall 1985) and current distribution of *Oreochromis mossambicus* in the Wilderness Lakes system.

Reproduction

According to logistic ogives, males reach Lm_{50} at 210 mm TL and females reach Lm_{50} at 232 mm TL (**Fig. 4.4**). The small sample size of the separate sexes was not ideal to determine an accurate length at maturity so a value for the sexes combined (240 mm TL) was used to determine the maturity of the *O. mossambicus* population in the Wilderness Lakes system.

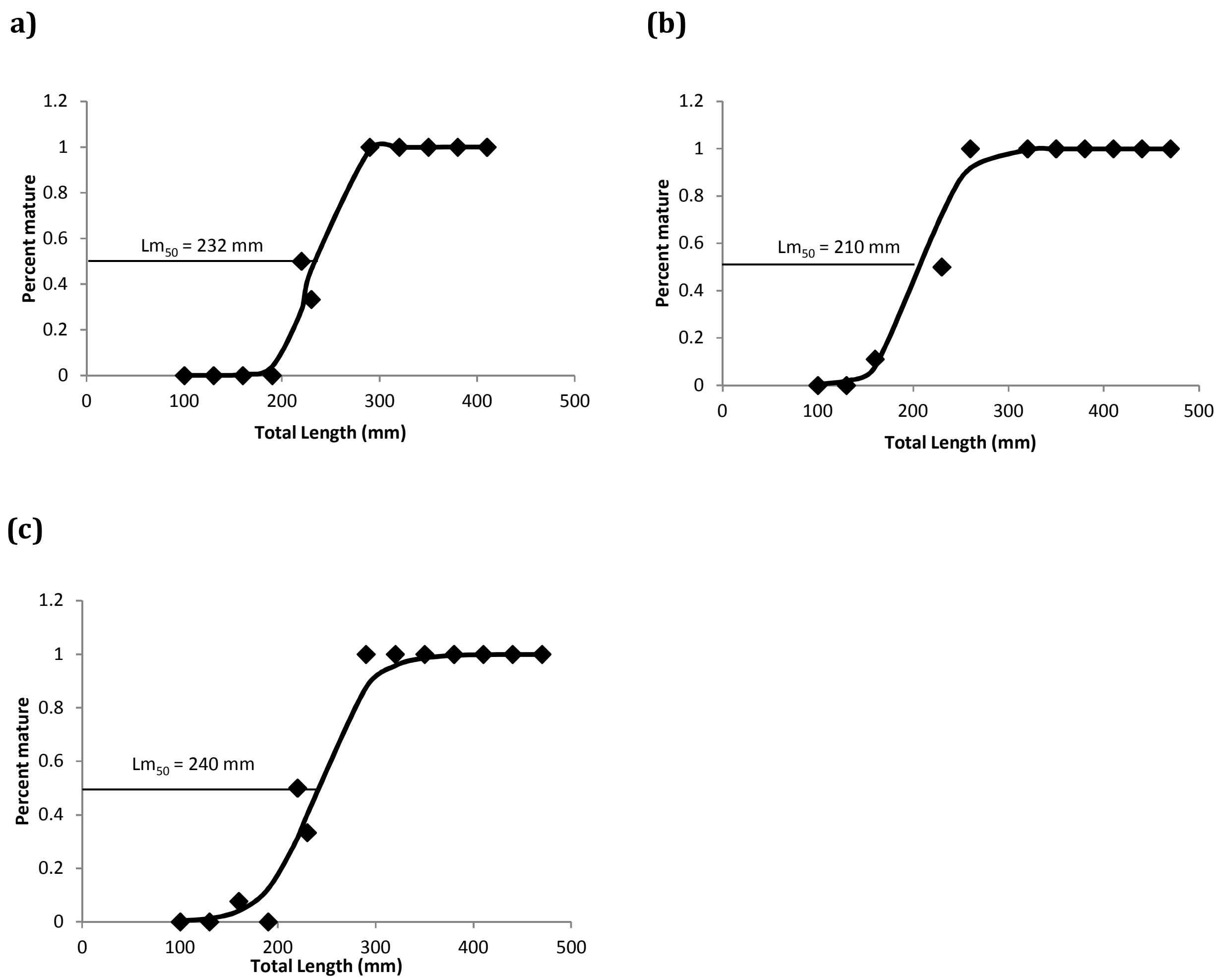


Figure 4. 4. Logistic ogives fitted to the percentage of sexually mature *Oreochromis mossambicus* (a) females, (b) males and (c) sexes combined. Lm_{50} = total length at 50% maturity.

Abundance and distribution

A total biomass of 594 kg of *O. mossambicus* was sampled from the Wilderness Lakes system in the fyke nets, 10 m and 30 m seine nets and gill nets. The percent biomass contribution of *O. mossambicus* in all the gears combined ranged from approximately 30% in Rondevlei (98 kg), Island Lake (182 kg) and Langvlei (182 kg) to more than 90% (62 kg) in the Rondevlei-Langvlei channel (**Fig. 4.5**). In the Rondevlei-Langvlei channel, this biomass was made up entirely of juvenile fish. In the study, large numbers of juvenile *O. mossambicus* were sampled throughout the system, with the highest abundances in autumn and summer (**Fig. 4.7**). The population structure indicates a larger proportion of juvenile fish with the highest numbers between 40 mm and 140 mm FL (**Fig. 4.6**). Few to no fish were sampled in the size classes between 240 mm and 340 mm FL. The length frequency distribution for each of the lakes, Rondevlei-Langvlei channel and the Touw estuary show the presence of both adults and juveniles, which indicates establishment (**Fig. 4.6** and **Fig. 4.7**).

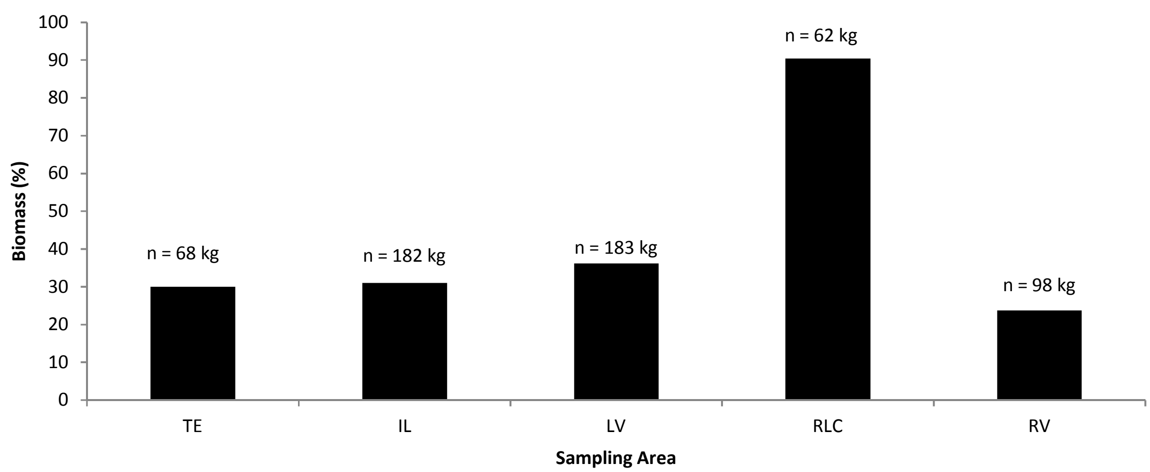


Figure 4.5. The accumulative biomass (%) contributed by *Oreochromis mossambicus* from fyke nets, 10m and 30m seine nets and gill nets in each of the lakes (RV = Rondevlei, LV = Langvlei, IL = Island Lake), the interconnecting channel (RLC = Rondevlei-Langvlei channel) and the Touw Estuary (TE).

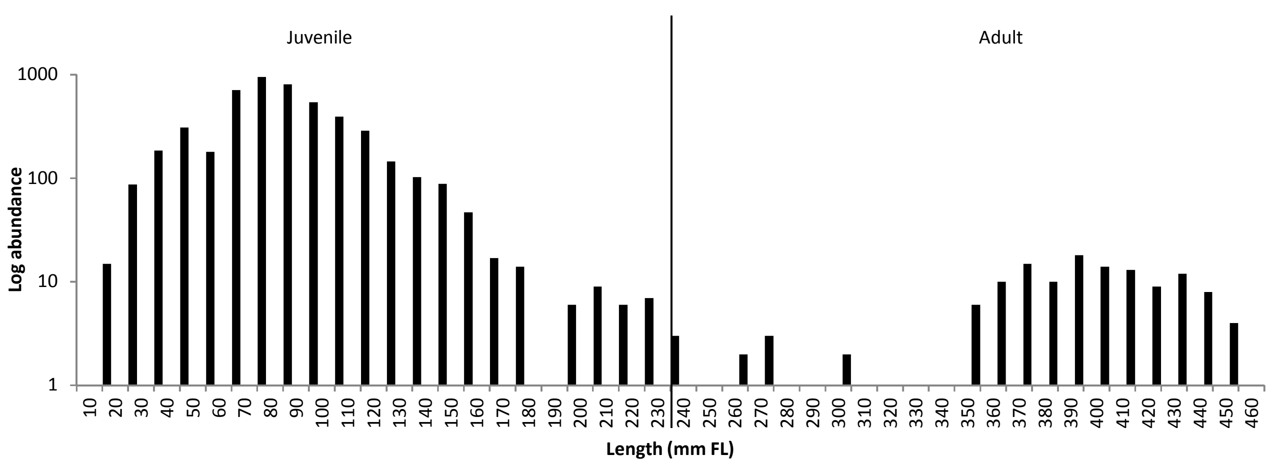


Figure 4.6. Log length-frequency distribution of *Oreochromis mossambicus* combining fyke net, 10m and 30m seine nets and gill nets throughout the Wilderness Lakes system sampled during the period May 2010 to May 2011. Different colours represent different sampling areas (n = 5057 fish).

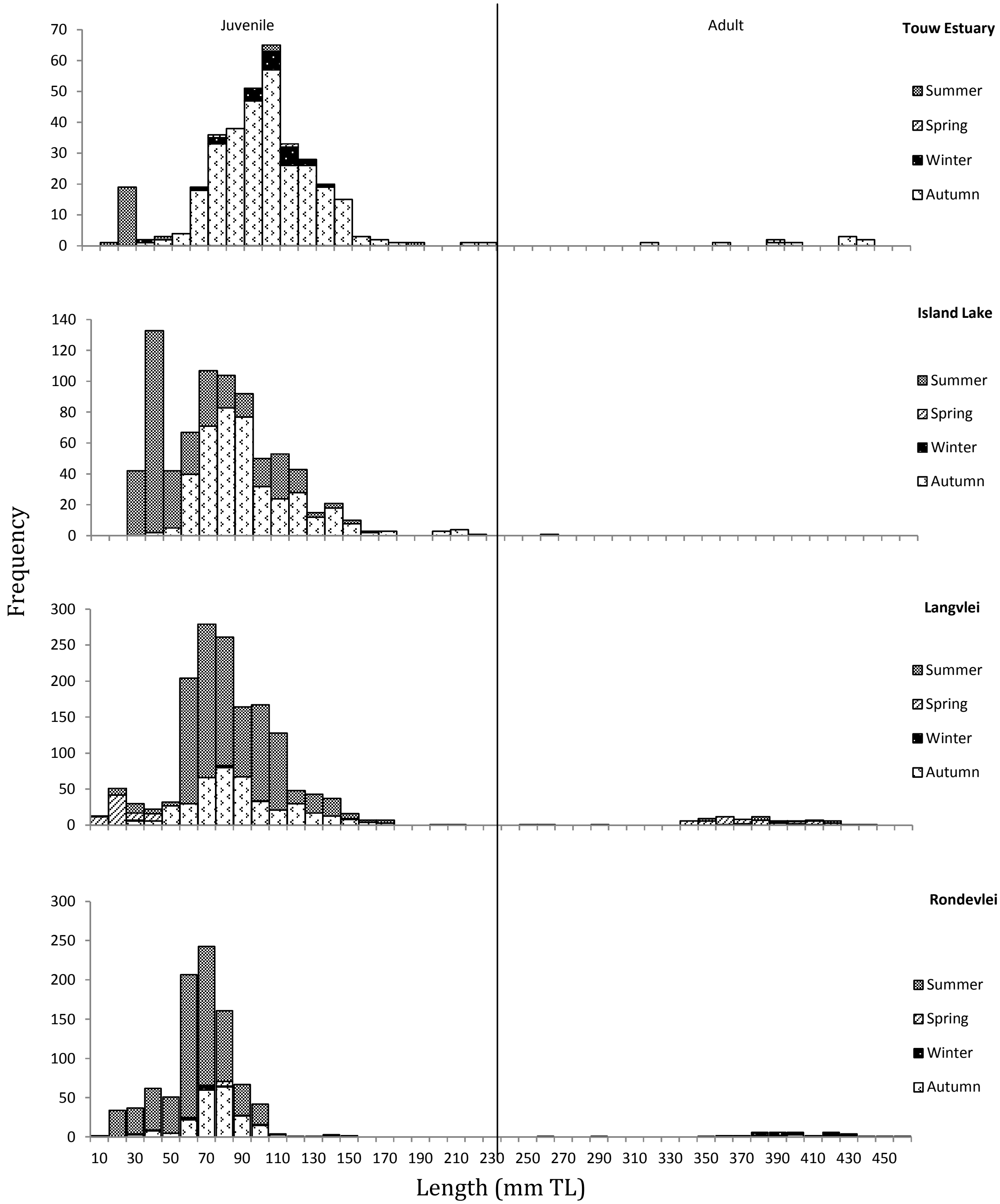


Figure 4.7. Seasonal size class distribution of *Oreochromis mossambicus* in each of the sampling areas in the Wilderness Lakes system, Western Cape, South Africa encorporating fyke net, 10 m and 30 m seine net and gill net catch data. Different fills represent different seasons sampled and vertical bar represents the difference between juvenile to adult fish.

Cyprinus carpio establishment

Introduction pathways

The official date of introduction into the Wilderness Lakes system is unrecorded. Illegal stocking into the nearby Groenvlei Lake occurred around 1997 and *C. carpio* were first suspected to be in the Wilderness Lakes system around 2003. *Cyprinus carpio* were first sampled in the system in 2009 in Langvlei. Since 2009, *C. carpio* have been sampled in each of the lakes, Serpentine channel and the Touw Estuary (**Table 4.6**).

Table 4.6. History of invasion of *Cyprinus carpio* into the Wilderness Lakes system, Western Cape.

Date	Phase		Source
1997	Introduction	*C. carpio* illegally introduced into Groenvlei - an adjacent fresh water lake the source is unknown	Johan Huisamen (ecological coordinator, Cape Nature)
2003	Introduction	Reports on *C. carpio* by anglers and SANParks staff from Island Lake	Heiko Feddersen (local angler), Ian Russell (aquatic ecologist, SANParks)
2009	Introduction	*C. carpio* first sampled in Langvlei, fish were initially between 150-300 mm FL	Olds et al. 2011
2010	Establishment and dispersal	*C. carpio* extended range to Rondevlei and Island Lake and are much larger (500 – 800 mm FL)	This study
2011	Establishment	*C. carpio* managed to breed with juveniles caught in Serpentine channel and Touw Estuary	This study

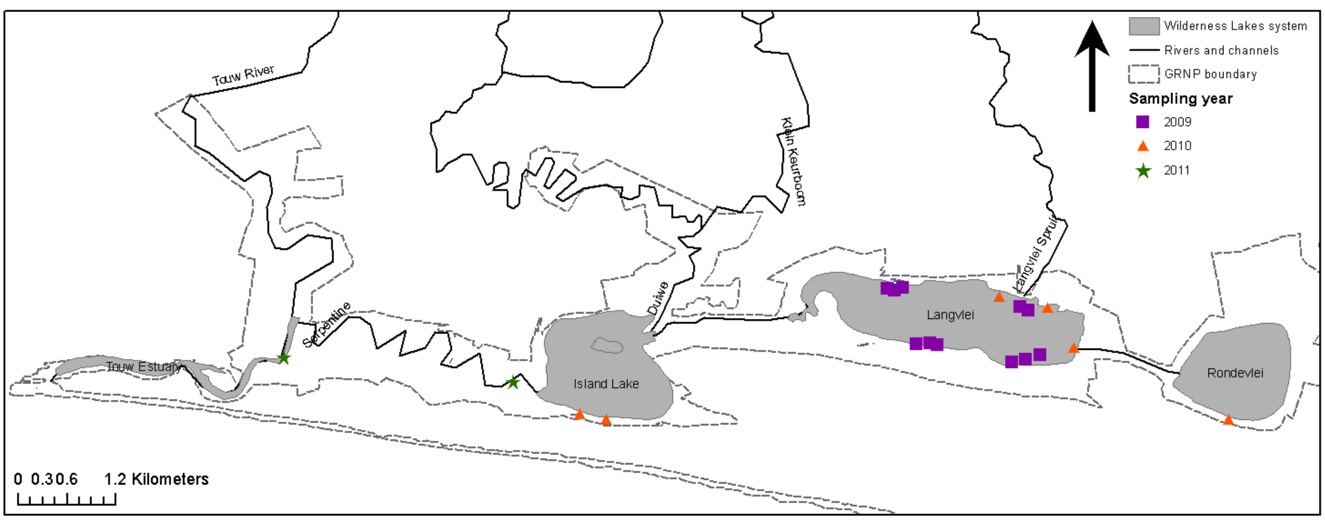


Figure 4.8. The distribution of *Cyprinus carpio* in the Wilderness Lakes system as sampled in 2009, 2010 and 2011.

Abundance and distribution

The first formal record of *C. carpio* in the Wilderness Lakes system was in May 2009, with fish measuring between 150 – 300 mm FL. These fish were sampled from Langvlei using fyke nets

(Olds *et al*. 2011) (**Fig. 4.8**). In 2010, larger (500 – 800mm FL) specimens were sampled in Rondevlei (n = 2), Langvlei (n = 5) and Island Lake (n = 5) in the 30 m seine net. None were sampled in fyke nets. Fish caught by anglers ranged in size from 560 – 750 mm FL (**Table 4.7**). Between January and March 2011, two juveniles (24 mm and 25 mm FL) were sampled in the 30 m seine net near the Touw Estuary mouth and one was sampled from the middle of the Touw Estuary (184 mm FL) in the gill net. The autumn 2011 fyke net sampling caught four juveniles in the Serpentine channel near the entrance to Island Lake ranging from 167 – 187 mm FL. The data show a progression in size classes from 2009 to 2011 and a confirmed and successful spawning event occurring in the Wilderness Lakes system (**Fig. 4.9 and 4.10**).

Table 4.7. Length and weight data for *Cyprinus carpio* caught by Heiko Feddersen, Edric Bruwer Johan Klopper, Dawie Blignout, Roco Meiring, Marius Botha and Jackie Feddersen (all local anglers) in Island Lake, Wilderness Lakes system between October 2010 and March 2011.

Date Caught	Sampling Area	Length (mm FL)	Weight (kg)	Source
2010/10/14	Island Lake	582	4.50	Johan Klopper
2010/10/14	Island Lake	648	6.28	Edric Bruwer
2010/10/14	Island Lake	670	6.94	Edric Bruwer
2010/10/14	Island Lake	571	4.23	Edric Bruwer
2010/10/14	Island Lake	566	4.10	Edric Bruwer
2010/10/14	Island Lake	614	5.31	Edric Bruwer
2010/10/23	Island Lake	637	5.95	Heiko Feddersen
2010/10/23	Island Lake	598	4.87	Heiko Feddersen
2010/10/23	Island Lake	665	6.79	Edric Bruwer
2010/10/24	Island Lake	713	8.46	Danie Blignout
2010/10/24	Island Lake	675	7.10	Heiko Feddersen
2010/10/24	Island Lake	646	6.21	Heiko Feddersen
2010/10/24	Island Lake	681	7.32	Roco Meiring
2010/11/20	Island Lake	648	6.26	Jackie Feddersen
2010/11/20	Island Lake	591	4.70	Jackie Feddersen
2010/11/20	Island Lake	751	9.90	Jackie Feddersen
2010/11/20	Island Lake	641	6.05	Jackie Feddersen
2010/11/20	Island Lake	619	5.44	Heiko Feddersen
2011/03/05	Island Lake	615	5.34	Heiko Feddersen
2011/03/05	Island Lake	666	6.83	Marius Botha
2011/03/05	Island Lake	673	7.07	Heiko Feddersen

Reproduction

The size class distribution of *C. Carpio* in the Wilderness Lakes system is shown in **Figure 4.10** and **Figure 4.11**. There were two distinct size classes between 2009 and 2010. In 2009 juvenile fish were sampled in Langvlei ranging from 130 - 290 mm FL. In 2010 the fish sampled were all adults ranging from 490 - 760 mm FL. In 2011 both juvenile and adult fish were sampled, indicating reproduction in the system and demonstrating that this species is establishing.

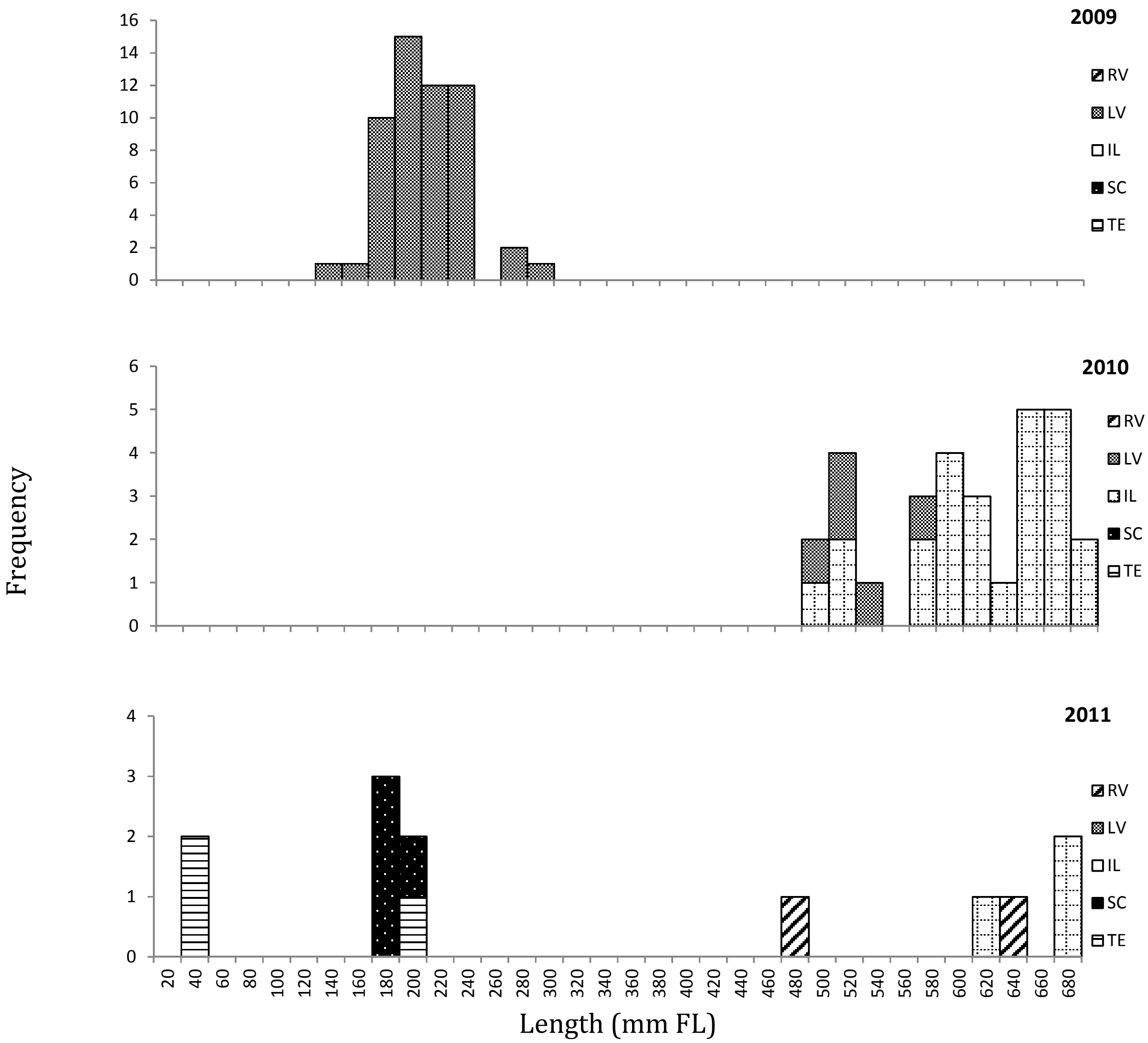


Figure 4.9. The size class distribution of *Cyprinus carpio* over the sampling period May 2009-May 2011 in fyke nets, 30 m seine nets, gill nets and angler records. Different colour bars represent different sampling areas. RV = Rondevlei, LV = Langvlei, IL = Island Lake, SC = Serpentine channel, TE = Touw Estuary.

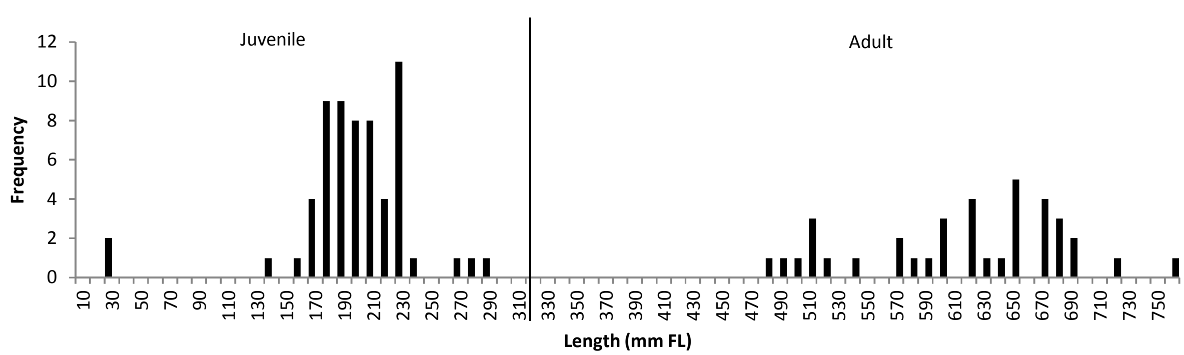


Figure 4.10. Length-frequency distribution of *Cyprinus carpio* combining fyke net, 30 m seine nets, angling and gill net data throughout the Wilderness Lakes system sampled during the period May 2009 to May 2011.

Examples of mature *C. carpio* sampled in the Wilderness Lakes system are shown in **Fig. 4.11**. The fish were sexually mature and were ready to spawn, which would indicate a minimum age of two years for the male and one year for the female. All adult fish caught were recorded as ripe, with one female recorded as spent (**Table 4.4**). These ripe fish sampled in August are likely to have spawned and resulted in the young of the year fish sampled in early 2011.

(a)

(b)

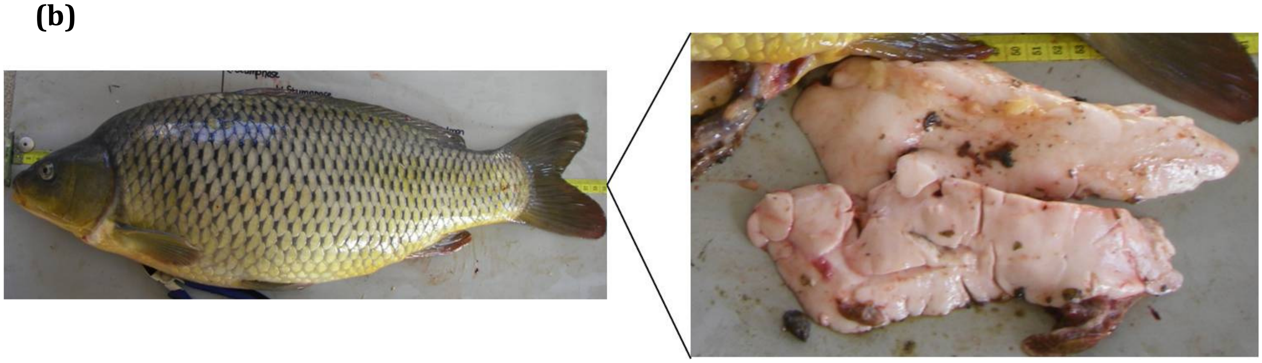


Figure 4. 11. Adult *Cyprinus carpio* showing **a)** ripe female measuring 504 mm FL caught in Langvlei in August 2010 and **b)** ripe male measuring 525 mm FL and caught in Langvlei in August 2010

Gambusia affinis establishment

Introduction pathways

Gambusia affinis were first reported in the Garden Route region (Groenvlei) around 1970 for the purpose of fodder fish for the introduced *M. salmoides* (de Moor 1988). The introduction date into the Wilderness Lakes system is unknown as they were not sampled by Hall (1985) but were sampled by Russell (1999b). *Gambusia affinis* was first sampled in the Wilderness Lakes system in the lower reaches of the Touw (n = 6) and Duiwe (n = 7) rivers in low numbers (**Table 4.8**).

Table 4.8. Potential introduction and spread of *Gambusia affinis* in the Wilderness Lakes system, Western Cape.

Date	Phase		Source
1970	Introduction	Introduced to Groenvlei as fodder fish	de Moor & Bruton (1988)
1972	Introduction	First sampled in the Karatara River near Sedgefield	de Moor & Bruton (1988)
1999	Introduction	Fish sampled in Duiwe River and Touw River	Russell 1999
2010	Dispersal	Fish sampled throughout the wilderness Lakes	Olds *et al.* 2011
2011	Establishment	Fish established a breeding population throughout Wilderness Lakes system	Sloterdijk 2011, this study

GPS point data from seine net and scoop net sampling between 2010 and 2011 provided the current distribution of *G. affinis*. The distribution had spread to the Touw Estuary, Serpentine channel, Island Lake, Island Lake-Langvlei channel, Langvlei, Rondevlei-Langvlei channel and Rondevlei (**Fig. 4.12**).

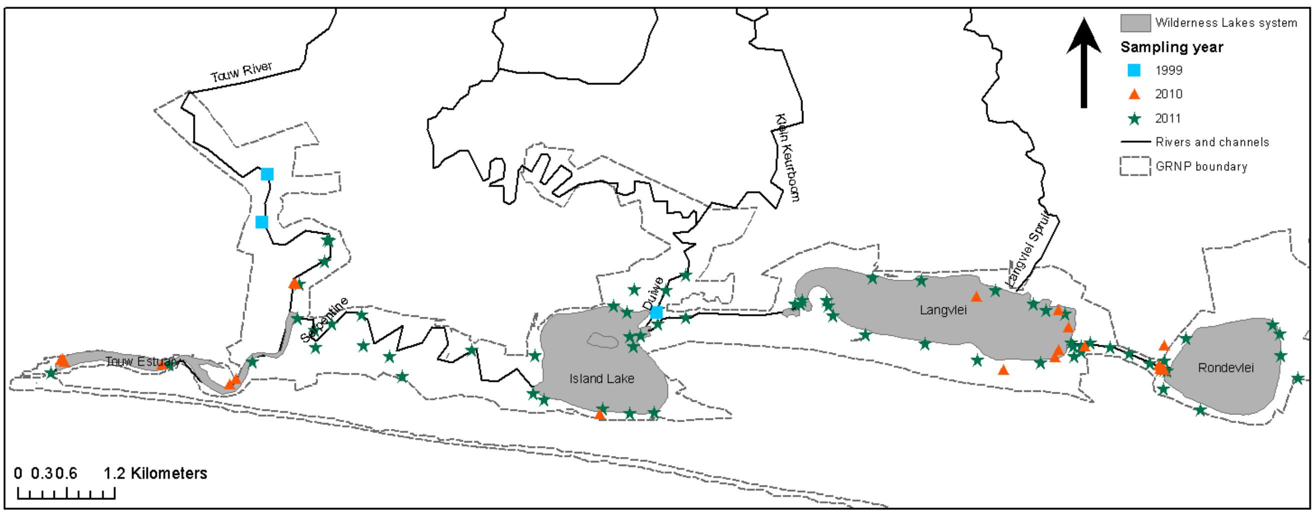


Figure 4.12. Historical (Russell 1999b) and current distribution of *Gambusia affinis* in the Wilderness Lakes system, Western Cape, South Africa.

Abundance and distribution

Gambusia affinis were sampled in each of the lakes, interconnecting channels and the Touw Estuary (**Fig. 4.12**). The CPUE throughout the system was variable, with the Serpentine channel and the Touw Estuary (81 ± 67 fish.scoop^{-1}) having a significantly higher CPUE (Kruskal-Wallis ANOVA; $H = 16.22$, $p < 0.05$) (**Fig 4.13**). CPUE was consistently the highest in the Langvlei-Island Lake channel (98.3 ± 199 fish.scoop^{-1}) and the lowest in Island Lake (13.7 ± 15.3 fish.scoop^{-1}). The number of fish caught per site ranged from 0 fish in deeper sampling sites in Rondevlei, Langvlei and Island Lake to 2701 fish in the Langvlei-Island Lake channel.

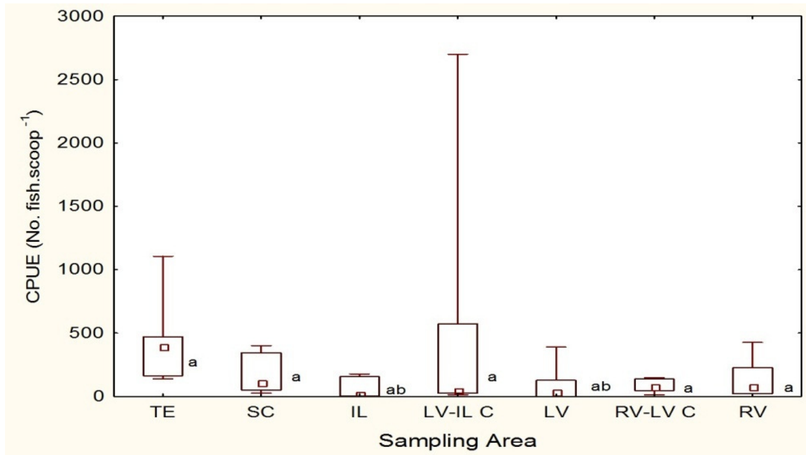


Figure 4.13. Box and whisker plot of Catch Per Unit Effort (CPUE) of *Gambusia affinis* in each of the water bodies sampled during the once off scoop net sampling in May 2011 (RV = Rondevlei, RV-LV C = Rondevlei-Langvlei channel, LV = Langvlei, LV-IL C = Langvlei-Island Lake channel, IL = Island Lake, SC = Serpentine channel, TE = Touw Estuary), Different letters denote statistically significant differences between samples (Kruskall-Wallis ANOVA).

Reproduction

The length frequency distribution from the Wilderness Lakes system shows a range of fish sizes from 8 - 36 mm SL (**Fig. 4.14**). While males mature smaller than females, maturity was taken at the smallest mature female according to Sloterdijk (2011). This measurement is 14 mm SL taken from the May 2011 sample. Adult fish comprised 76% of the population, with 24% of the population comprised young of the year stock (**Fig. 4.14**).

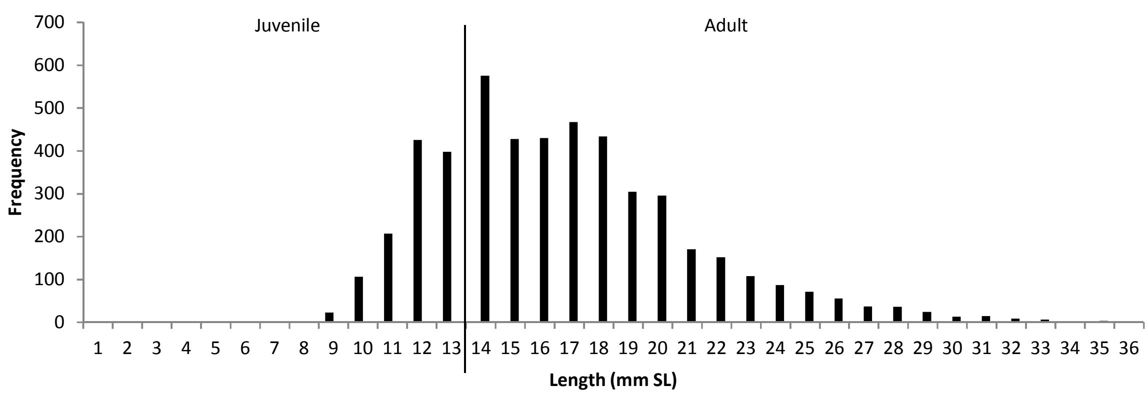


Figure 4.14. Length frequency distribution for *Gambusia affinis* sampled using scoop net data collected in Rondevlei, Rondevlei-Langvlei channel, Langvlei, Langvlei-Island Lake channel, Island Lake, Serpentine channel, Duiwe River and the Touw Estuary between the sampling period May 2010 – May 2011.

Micropterus salmoides establishment

Introduction pathway

Micropterus salmoides was introduced into Groenvlei in 1934 for angling purposes (de Moor & Bruton 1988) however the date of introduction of *M. salmoides* into the Wilderness Lakes system is unknown (**Table 4.10**). Along with *O. mossambicus*, *M. salmoides* was first sampled by Hall (1985) in Langvlei and Island Lake. In Langvlei, five juveniles were sampled in a 10 m seine net with a combined weight of 4 g and six were sampled in Island Lake (**Fig. 4.16**). One specimen weighing 145 g was sampled in the gill nets and two with a combined weight of 308 g were sampled in the 30 m seine net. Three were sampled in the 10 m seine net with a combined weight of 2101 g.

Table 4.10. History and introduction of *Micropterus salmoides* into the Wilderness Lakes system, Western Cape.

Date	Phase		Source
1934	Introduction	Fish stocked into Groenvlei for angling purposes	De Moor & Bruton (1988), Johan Huisamen (ecological coordinator, CapeNature)
1983	Introduction	Juveniles and adults sampled in Langvlei and Island Lake	Hall 1985
1999	Introduction	Sampled in Duiwe River which leads into Island Lake	Russell (1999b)
2010	Dispersal	Adult fish seen in Langvlei	This study
2011	Dispersal	Juvenile fish sampled in Duiwe River in early 2011	This study

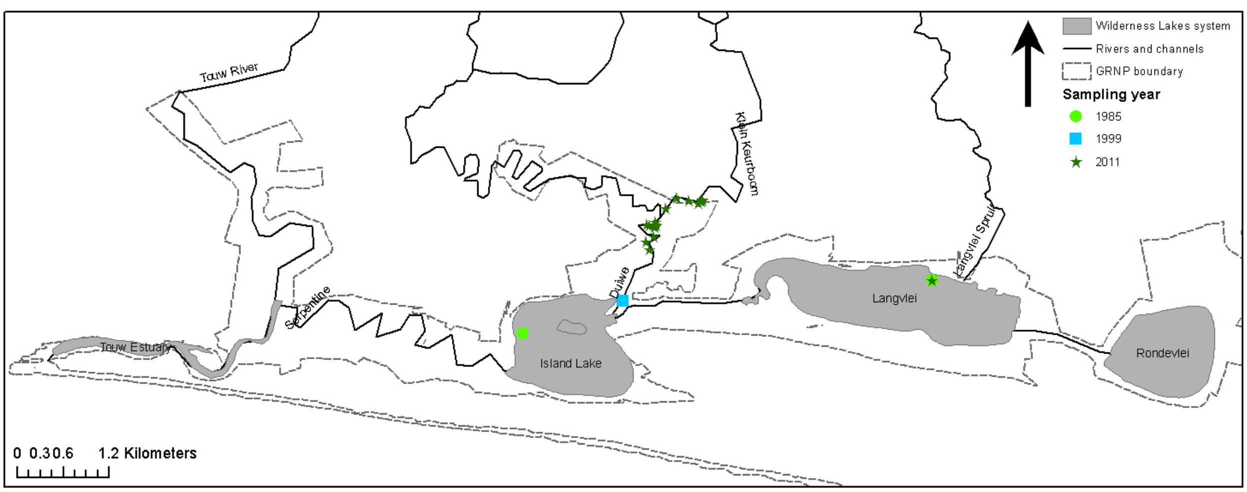


Figure 4.15. Historic (Hall *et al*. 1985, Russell 1999b) and current distribution of *Micropterus salmoides* in the Wilderness Lakes system, Western Cape, South Africa.

Abundance and distribution

The freshwater study by Russell (1999b) sampled six *M. salmoides* in the lower reaches of the Duiwe River in deeper pools (< 30 cm) by trolling a lure. During this study, 40 juvenile *M. salmoides* were sampled in the Duiwe River in February 2011 (**Fig. 4.18)** and were sampled in shallow pools, ranging from 75 - 120 mm FL (**Fig. 4.17**). The distribution of *M. salmoides* in 2010 has been limited to one confirmed sighting of two adults in the shallows of Langvlei and reports from anglers in Island Lake.

A positive confirmation of *M. salmoides* in Island Lake was provided by two local anglers (Armand van der Harst, **Fig. 1.3** and Jerian Spaans, **Fig 4.16**). Interviews with local anglers reported catches of *M. salmoides* ranging in sizes from approximately 300 g to 2 kg with the most abundant size class being 1.2 to 1.5 kg and fewer larger sizes. Catch rates were reported to fluctuate over fishing sessions but one angler reported catch rates during one fishing session to be as high as 15 fish caught in three hours.

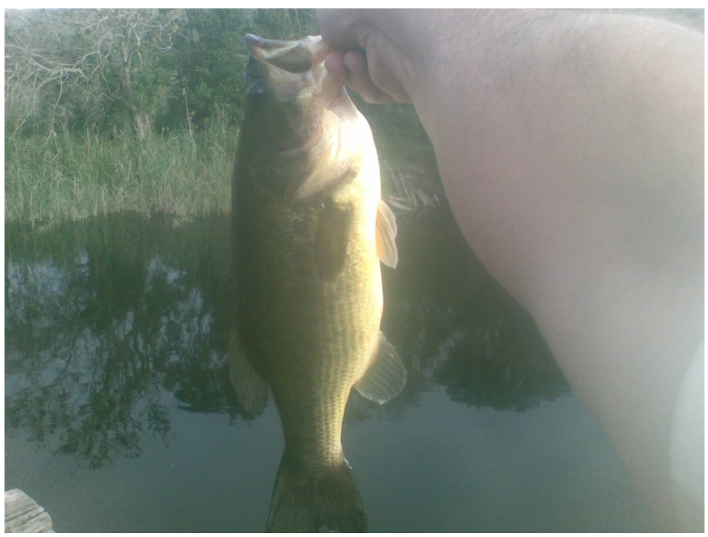


Figure 4.16. Adult *Micropterus salmoides* caught in Island Lake, Wilderness Lakes system, South Africa, by a local angler (Jerian Spaans) on 2010/11/01).

Reproduction

Extensive sampling throughout the Wilderness Lakes did not sample any *M. salmoides*; however, juveniles ranging in sizes from 75 – 120 mm FL (**Fig 4.17**) were sampled at 17 electro fishing stations in the Duiwe River (**Fig 4.18**). While juveniles were sampled in the tributaries into the Wilderness Lakes system, in the lakes themselves, no evidence of spawning was observed.

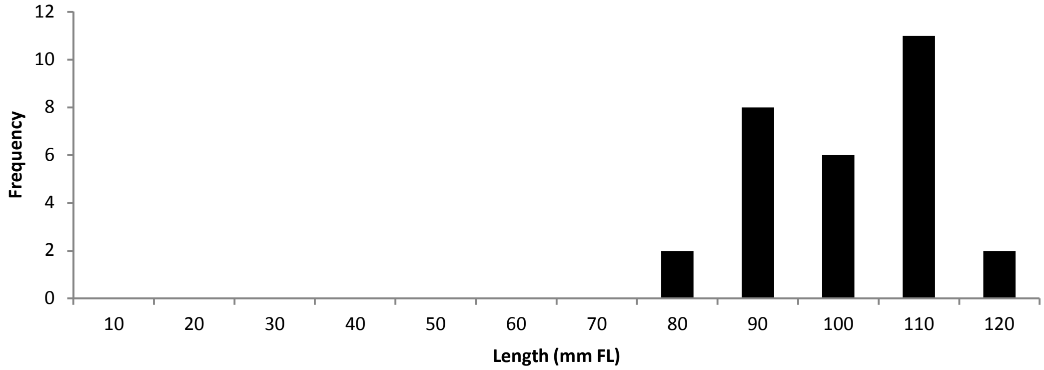


Figure 4.17. Length frequency distribution of juvenile *Micropterus salmoides* sampled in the Duiwe River which feeds into Island Lake, Wilderness Lakes system.

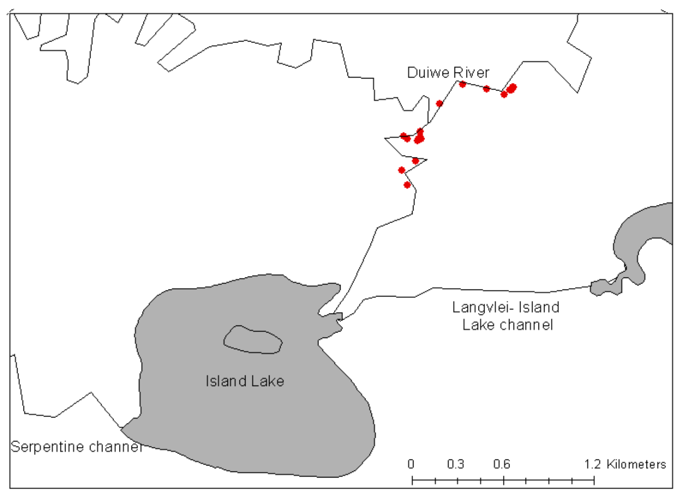


Figure 4.18. Location of electro-fishing sampling sites along the Duiwe River adjoining the Wilderness Lakes system where juvenile *Micropterus salmoides* were sampled.

Introduction pathways

There are no formal records of introduction dates and source populations into the Wilderness Lakes system. Interviews with farm owners with farm dams in the catchment revealed potential source populations of *M. salmoides*, *C. carpio* and *O. mossambicus*. Various farm dams were stocked with *M. salmoides* and *O. mossambicus*, which feed into the Duiwe River and Langvlei Spruit, between 1950 and 1980 (**Fig. 4.19; Table 4.10**). Dam number five (**Fig. 4.19; Table 4.10),** which is located towards the uppermost reaches of the Duiwe River, was stocked with *M. salmoides*, *O. mossambicus* and *C. carpio* in the 1950s. Farm dams lower in the catchment, which are connected to each other via overflow pipes to avoid flood damage, also contain *M. salmoides* and *O. mossambicus*. It is possible that the fish moved downstream via these overflow pipes into the Duiwe River, Langvlei Spruit into the Wilderness Lakes system.

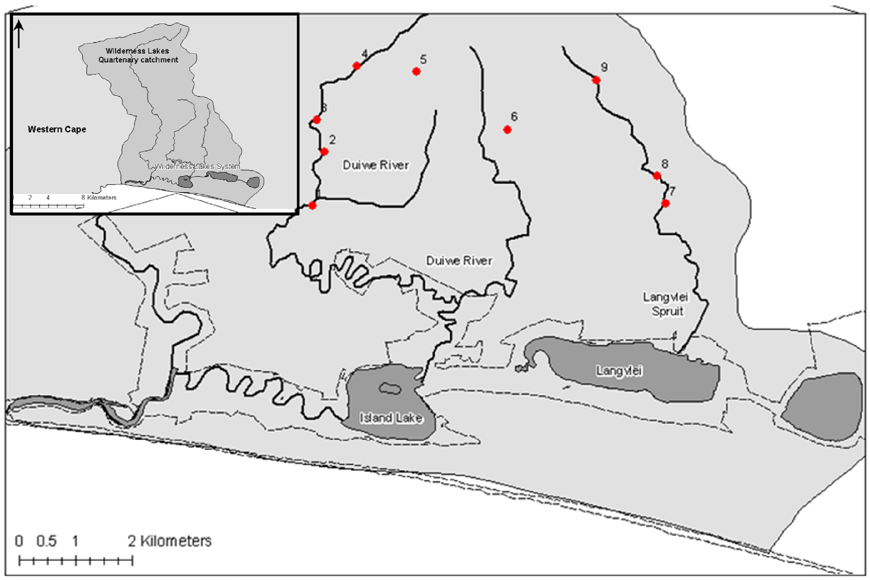


Figure 4.19. Location of farms in the Wilderness Lakes quarternary catchment where farmers were interviewed for potential source populations and introduction dates.

Table 4.10. Summary of interviews with selected farm owners with dams stocked with alien species in the Wilderness Lakes system catchment.

Dam No.	Date stocked	Species	Summary
1	1960s	*O. mossambicus*, *M. salmoides*	Did not stock dams directly, were introduced via upstream overflow, all farm dams connected via overflow pipes
2	Data lacking		
3	Pre-1984	*M. salmoides*	Owners did not stock fish into dams and rarely attempt to catch fish, data limited
4	1960s	*M. salmoides*, *C. carpio*, Salmonid spp	Currently *M. salmoides* and *C. carpio* in dams, trout died out. Dams on farm connected to dams lower in catchment
5	Pre-1998	*O. mossambicus*	Previously stocked with *M. salmoides* but a drought in 2009/2010 killed the population
6	Data lacking		
7	Pre-1994	*O. mossambicus*	Previously stocked with *M. salmoides* but drought in 2009/2010 killed the population

DISCUSSION

Biological invasions are complex processes consisting of a sequence of phases from introduction to a new system to the establishment, dispersal and impact of these species on native fauna (Rosecchi *et al*. 2001; García-Berthou 2007). Determining when species have established cannot be based on a standard set of criteria as life history strategies vary greatly between species. Life history traits may represent a critical aspect of species ecology because life history strategies are considered to have evolved from constraints among traits that have consequences for reproduction and fitness in different environments (Olden *et al*. 2006). Invasive species which fall into the Winemiller & Rose (1992) opportunistic strategy, with traits such as small adult size, high fecundity, rapid growth and early maturity, are often considered to have an increased likelihood of invasion success because of their ability to rapidly colonize new areas once introduced (Winemiller & Rose 1992; Moyle & Marchetti 2006). However this suggests that these traits are an advantage to a fish invader only if individuals also live long enough to survive periods of unfavourable conditions (Moyle & Marchetti 2006). With different life histories, and having entered the Wilderness Lakes system at different times, the stages of establishment of the four alien species are different.

Oreochromis mossambicus

Introduction pathways

Oreochromis mossambicus was introduced into the system at least 30 years ago. While there is no published record of its introduction into the Wilderness Lakes system, the species was first stocked into the nearby Groenvlei Lake to control algal growth (J. Huisamen, ecological coordinator, CapeNature, pers. comm.). This species has been widely translocated primarily for recreational fishing and aquaculture, and on a smaller scale as a forage fish for *M. salmoides,* an agent for the biocontrol of macrophytes and chironomids, and to augment commercial fisheries (de Moor & Bruton 1988; Costa-Pierce 2003; Canonico *et al*. 2005). Introductions for angling purposes for this species are common, and this may be one of the major pathways into the Wilderness Lakes system, as numerous dams in the catchment are stocked with *O. mossambicus* for angling purposes. Although tilapiine fishes have been widely distributed for aquaculture purposes (Costa-Pierce 2003; Maddern *et al*. 2007), it is unlikely that this would have occurred in the Wilderness Lakes as no dressing and packing facilities are located nearby. There is no evidence of direct stocking into the Wilderness Lakes system and the most likely source of introduction of these fish is as a result of dispersal from further up the system.

Establishment

Once in the Wilderness Lakes system, this species established rapidly and dispersed throughout the system. It is now widespread and abundant. Within Australian waters *O. mossambicus* have been shown to spread rapidly and establish breeding populations within a few years (Maddern *et al*. 2007). Rapid invasions by *O. mossambicus* and closely related *O. niloticus* have been reported in numerous countries, with negative impacts on native biota including decreased abundance and extinction of native species resulting from habitat and trophic overlaps and competition for spawning sites, habitat destruction and water quality changes (Weyl 2008). Introduction of *O. niloticus* into the Kafue and Zambezi river catchments for aquaculture purposes in the 1960s resulted in the distribution of this species out of its intended range and it subsequently established which resulted in hybridisation with the native *O. mossambicus*. This has resulted in a loss of the genetic integrity of *O. mossambicus* (Canonico *et al.* 2005).

Dispersal

The historic and current distribution of *O. mossambicus* in the Wilderness Lakes system has remained the same. They are widely distributed throughout the Wilderness Lakes system, with the highest abundance in Langvlei, Rondevlei and their interconnecting channel. Juveniles ranging from 10 - 190 mm TL were very abundant in the fyke net and 10 m seine net catch in the Rondevlei-Langvlei channel showing a preference for shallow, more sheltered habitats. Adults were sampled primarily in the gill nets set in deeper water. During summer, large numbers of adults were seen in the shallow water guarding nests (pers. obs.); this behaviour was also noted by James & Bruton (1992) and Ellender *et al*. (2008) in the Eastern Cape, South Africa. Although this species can tolerate seawater salinities (Whitfield & Blaber 1979), Potter *et al*. (1990) showed that during mouth opening events of seasonally closed estuaries they generally retreated to the upper reaches of the estuary. The current study did not show a change in distribution during periods of higher salinity and it was consistently one of the most important fish species in all sampling gears.

Cyprinus carpio

Introduction pathways

Cyprinus carpio were initially introduced into South Africa as ornamental fish but over time it has been introduced into South African water systems primarily for angling purposes. This is one of the major pathways of invasion. It was stocked into one farm dam in the catchment so there were potential invasion opportunities. As with the illegal stocking of *C. carpio* into Groenvlei, it is assumed that the introduction of the species was either from an illegal introduction for angling purposes or through dispersal from a farm dam in the catchment (I Russell, aquatic ecologist, SANParks, pers. comm.).

Establishment

Cyprinus carpio were sampled in each of the lakes, Serpentine channel and the Touw Estuary. While it has become widespread, the abundances appear to be low. The expansion in range and the progression in sizes between 2009 and 2011 and the confirmed spawning in early 2011 may indicate that this species is establishing in the Wilderness Lakes system. Once it establishes in a system, the population size can increase rapidly dominating the system (Koehn 2004; Britton *et al*. 2007). This species is Australia's most invasive freshwater fish, contributing more than 90% of fish biomass in many areas of south-east Australia (Koehn 2004). Another African system that has been invaded by *C. carpio* where the species has established rapidly is Lake Naivasha, Kenya (Britton *et al*. 2007). Analysis at a global scale shows that they have most of the attributes expected for a successful invasive species (Koehn 2004). They have a well documented history of successful invasion with wide distribution and abundance, wide environmental tolerances, rapid growth, high reproductive capacity, broad diet, early sexual maturity and short generation times (de Moor 1996; Koehn 2004; Oyugi *et al.* 2011; Winker *et al*. 2011).

Dispersal

A survey of estuaries in the Eastern Cape found that in the Great Fish estuary, *C. carpio* contributed 5.2% to the catch composition (James & Harrison 2010). The range expansion of freshwater species (*O. mossambicus* and *C. carpio*) into estuaries during periods of increased riverine flow occurs in the Bot Estuary (Bennett 1989). This may also influence the distribution of *C. carpio* in the Wilderness Lakes system. Juvenile *C. carpio* were only sampled in the fresher Touw Estuary and in Island Lake, and it seems possible that the higher salinities restricted successful reproduction in Langvlei and Rondevlei during the spawning season. Winker *et al*. (2011) relied heavily on angling competitions for gathering *C. carpio* samples (provided 86% of *C. carpio* sampled) as seine net and gill net sampling was not adequate in collecting samples because *C. carpio* often develop avoidance behaviour against sampling gear (Arlinghaus & Mehner 2003).

Gambusia affinis

Introduction pathways

Gambusia affinis was initially stocked into aquatic systems for the biocontrol of mosquitoes and as forage fish for *M. salmoides* (de Moor & Bruton 1988). This species was introduced into the

Wilderness Lakes region around 1972 into the Karatara River near Sedgefield, and later (pre-1977) into Groenvlei. No official record of when *G. affinis* was introduced into the Wilderness Lakes system exists but the species was first sampled in the Touw and Duiwe Rivers by Russell (1999b). Since then their distribution has spread to all sampling areas in the Wilderness Lakes.

Establishment

Gambusia affinis is a widespread and abundant species in the Wilderness Lakes system. The large range expansion between 1999 and 2011 indicates that it has the ability to expand into new water systems quickly and effectively. Short generation times allow it to colonise new areas rapidly and become rapidly abundant (Pyke 2008). Sloterdijk (2011) has shown that this species is successfully breeding in the system.

Dispersal

Sampling aimed at the densely vegetated, shallow regions of this system produced high abundances of *G. affinis*. *Gambusia affinis* generally occurs in shallow, slow moving or still water that is densely vegetated (Casterlin & Reynolds 1977; Arthington & Lloyd 1989; Skelton 2001; Pyke 2005, 2008). This would corroborate the very high CPUE (98 ± 3.4 fish.scoop^{-1}) in the Langvlei-Island Lake channel, which is very narrow and densely vegetated. While *G. affinis* was sampled in the majority of sites sampled, areas that were deeper than approximately 1m with sparse vegetation did exhibit lower CPUE. Abundance typically varies seasonally, with numbers being lowest prior to, and highest after the breeding season (Pyke 2008; Sloterdijk 2011). An increase in the catches of *G. affinis* was noted in summer, especially in the Touw Estuary from 0.2 ± 0.7 fish.scoop^{-1} in winter to 11 ± 25 fish.scoop^{-1} in summer but catches were not high enough to conclusively say there were seasonal changes. Sloterdijk (2011) found seasonal differences in catches in the Wilderness Lakes system and linked this to low temperatures, possibly making *G. affinis* more susceptible to disease and predation than they would be at higher temperatures.

Micropterus salmoides

Introduction pathways

Micropterus salmoides was originally introduced to South Africa for angling purposes in 1928 (Skelton & Weyl 2011) and into Groenvlei Lake in 1934 for the same purposes (J. Huisamen, ecological coordinator, CapeNature, pers. comm). Stocking in farm dams in the catchment occurred between 1950 and 1960. The most likely source of introduction is via farm dams further up the system where the majority of the dams are connected via overflow pipes. The introduction pathways and date of introduction of the alien invasive species is not conclusive but farm dams in the catchment are a possible source of introduction for *M. salmoides* and *O. mossambicus*. This conclusion is aided by the presence of *M. salmoides* in the upper reaches on the Duiwe River. These areas may not be accessible to the adult fish in Island Lake due to low water levels and a V-notch weir in the lower reaches of the Duiwe River (**Figure 4.20**). Juvenile *M. salmoides* can move downstream into Island Lake during periods of higher water flow in the Duiwe River, which would result in repeated introductions into the Wilderness Lakes system.

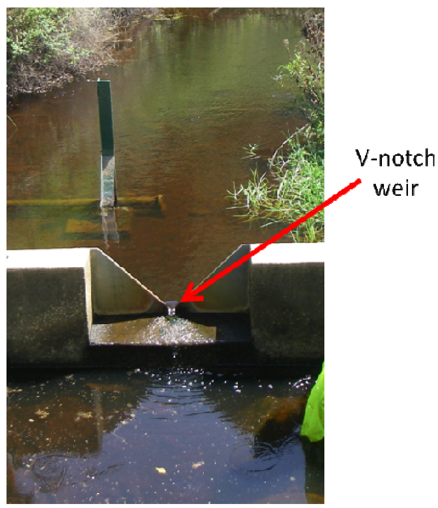


Figure 4.20. V-notch weir in the lower reaches of the Duiwe River which is the freshwater source to Island Lake, Wilderness Lakes system, Western Cape.

Establishment

The distribution of *M. salmoides* appears to have remained unchanged over the past 28 years. Hall (1985) sampled it in Langvlei and Island Lake where Russell (1999) sampled juveniles in the Duiwe River. Gill net and seine net sampling in Lake Chicamba, Mozambique was adequate in sampling *M. salmoides* (Weyl & Hecht 1999), which would indicate that these sampling gears would have sampled *M. salmoides* in the Wilderness Lakes system if they were abundant. The lack of samples caught in the numerous sampling gears used may be an indication of low abundances in the Wilderness Lakes system. *Micropterus salmoides* are intensely piscivorous, with an ontogenetic shift from eating invertebrates when fish are juveniles to fish in the adult stages (Jang *et al*. 2006; Weyl & Lewis 2006). In estuarine systems, *M. salmoides* have been reported to feed on native estuarine fish species (Weyl & Lewis 2006)

Dispersal

The distribution of *M. salmoides* appears to be limited to low numbers of fish in Langvlei and Island Lake. Angler records were therefore important in determining the distribution of this species. Anglers have been catching *M. salmoides* in Island Lake for approximately 30 years with sizes ranging from 300 g to 2 kg (C. White, local angler, pers. comm.). Reports from anglers indicate that catches can occasionally be high as 15 fish in 3 hours. Fish were generally caught in areas with lower salinities. In their native range, *M. salmoides* move to areas of lower salinity when salinity increases outside their preferred tolerances (Norris 2007). The habitat and salinity preferences for *M. salmoides* may limit their distribution in the Wilderness Lakes system to areas near the freshwater sources. This is seen quite clearly in Island Lake, with fish congregating at the Duiwe River inlet to Island Lake and on rocky substrates.

Life history

While *M. salmoides* appears to be an established species due to its extended period in the system, its distribution in the system might be limited by environmental factors, mainly salinity. It is possible that the higher salinities in the Wilderness Lakes system inhibit spawning in *M. salmoides*, and juveniles from farm dams which enter the system via the Duiwe River may replenish stocks annually, making this a casual species. Richardson *et al*. (2011) described alien species that do not form self-replacing populations in the invaded region and whose persistence depends on repeated introductions into the system as casual species.

Conclusion

Life span of fishes, adequate sampling and repeated introductions all affect species abundances and distributions over time (García-Berthou 2007). For many studies, the variables most identified as determining establishment are reproductive modes (parental care), diet preferences and environmental tolerances (García-Berthou 2007).

The invasive species found in the Wilderness Lakes system all appear to be in different stages of establishment (**Table 4.12**).

Table 4.12. Establishment status assessed by distribution, abundance and breeding success of four alien invasive fish species in the Wilderness Lakes system, Western Cape

	Distribution	**Abundance**	**Breeding**	**Status**
Oreochromis mossambicus	Widespread	High	Yes	Established
Gambusia affinis	Widespread	High	Yes	Established
Cyprinus carpio	Widespread	Low	Yes	Establishing
Micropterus salmoides	Limited to Island Lake, Duiwe River inflow and Langvlei	Very low	No	Casual

Gambusia affinis and *O. mossambicus* are established in the Wilderness Lakes system and are both strong components of the fish fauna in the system. *Cyprinus carpio* appears to be in the establishing phase where it is widespread but not yet abundant and has recently spawned in the system (**Table 4.12**). *Micropterus salmoides* has been in the Wilderness Lakes system for an extended period, but the abundance and distributions remain low and it appears that repeated introductions from upstream sustain the population, indicating a casual species (**Table 4.12**). The factors that may play a role in controlling the establishment of the alien species may be abiotic. Therefore, an assessment of the environmental factors for establishment in the Wilderness Lakes system is made in Chapter 5.

CHAPTER 5

FACTORS ALLOWING THE ESTABLISHMENT OF ALIEN SPECIES IN THE WILDERNESS LAKES SYSTEM

INTRODUCTION

Climatic conditions control the potential range that any given species can occupy at a specific time (Jackson *et al*. 2001). Whether or not a species occupies certain areas within its potential range depends on a combination of historic/biogeographic conditions defining the area; such as previous opportunities to colonize the area and contemporary factors at a smaller scale such as predation or environmental gradients (Jackson *et al*. 2001). Roughgarden & Diamond (1986) envisioned that abiotic factors play an important role in community ecology by delimiting a pool of species that could possibly occupy a specific area or habitat. Abiotic factors and physiological responses to those factors determine where a species can survive and persist, and segregation among species is due to the interaction between physiological responses to abiotic factors and competitive abilities (Dunson & Travis 1991; Akin *et al*. 2005).

Large-scale patterns in the distribution of species in estuarine systems result primarily from species responses to the physical environment. Abiotic factors may act like a physiological sieve, thereby playing an important role in the structuring of species in a community (Martino & Able 2003). Investigations into invading fishes in California, America suggests that if abiotic factors are favourable for an alien species, that species is likely to successfully invade, regardless of the biota already present (Moyle & Light 1996). When exotics fail to become established despite repeated invasions, that failure is best attributed to their inability to adapt to abiotic conditions rather than to biotic resistance on the part of the recipient community (Moyle & Light 1996).

The hypotheses of biotic acceptance and biotic resistance address the establishment and range expansion stages of alien invasion. Biotic acceptance is a theory which argues that ecosystems accommodate the establishment and coexistence of alien species despite the presence and abundance of native species (Fridley *et al*. 2007; Leprieur *et al*. 2008; Richardson *et al*. 2011). This theory predicts that the establishment of non-native species will be greatest in areas rich in native species and with optimal environmental conditions for growth. The opposite of this is biotic resistance, where native species are resistant to the establishment of invasive species (Leprieur *et al*. 2008; Richardson *et al*. 2011). Biotic resistance predicts that species-poor communities will host more non-native species than more species-diverse communities, the latter being highly competitive and hence readily impede the establishment of non-native species, leading to a negative relationship between native and non-native species (Moyle & Marchetti 2006; Leprieur *et al*. 2008). In a global context, Leprieur *et al*. (2008) found that biotic resistance could not adequately explain the observed pattern of fish invasions as there were no negative relationships between native and non-native species richness. This suggests that regional species-rich communities are not necessarily a barrier against the establishment of non-native species.

To determine whether biotic or abiotic factors affect the establishment success of alien species three hypotheses needed to be tested. These hypotheses are:

1) Abundance of alien fishes decreases with increased native fish diversity
2) The abiotic environment in the Wilderness Lakes system limits the distribution of alien fishes in the Wilderness Lakes system
3) The abundance of alien fishes can be correlated with temperature and salinity in the Wilderness Lakes system

METHODS AND MATERIALS

To test whether abiotic or biotic factors limit the distribution of the four alien fishes, their distribution and relative abundance determined from seine netting, gill netting and fyke netting (Chapter 2) were correlated with abiotic factors (temperature, salinity, conductivity, dissolved oxygen and pH) and with the biotic factors, species richness and diversity.

Water quality sampling and analysis

Salinity, temperature, conductivity, dissolved oxygen and pH were measured during each sampling trip. These data were grouped into the four sampling seasons (June to August - winter, September to November – spring, December to February - summer and March to May - autumn) using the methods described in Chapter 2. Two permanent water temperature loggers

(HOBO Pro v2 water temperature loggers (U22-001) and Hobo Pendant Temperature/light loggers (UA-002-64)) were placed in each of the lakes and the Touw Estuary and one in the Rondevlei-Langvlei channel. These loggers were placed between 1.5 and 2 m below the water surface and logged temperatures every three hours for a period of one year (2 February 2010 – 11 February 2011). One reading, at 10 am from one of the loggers, was used to plot the temperature changes in each of the lakes, the Touw Estuary and the Rondevlei-Langvlei channel. Additional long term temperature and salinity data were obtained from SANParks who have monitored water quality throughout the Wilderness Lakes system on a monthly basis from January 1991 to October 1999 and then on a quarterly basis from January 2000 to January 2011. During these SANParks surveys, measurements were taken at 30 cm depth where water temperature and salinity were measured using a YSI Model 33 S-C-T meter. In each sampling event five measurements were taken in each of the lakes and between seven and nine measurements were taken in the Touw Estuary depending on water levels.

Statistical analysis

Physico-chemical parameters

A cluster analysis was used to explore the degree of similarity between the six sampling areas based on the physico-chemical parameters (temperature, salinity, conductivity, dissolved oxygen and pH) collected between April 2010 and May 2011 using PRIMER 6. A Bray-Curtis similarity measure was used on non standardised and untransformed data and from these data were grouped into a cluster analysis based on the degree of similarity. This similarity matrix was used to group sampling areas according to their similarities to determine which areas in the Wilderness Lakes system would be likely to limit establishment.

Assumptions for normality and homogeneity of the variance were not met by the physico-chemical data, therefore the non-parametric Kruskal-Wallis ANOVA (H) was used to assess for seasonal differences between sampling areas. The non-parametric Kruskal-Wallis ANOVA was also used to determine the differences between physico-chemical parameters in each sampling area to determine whether distributional differences in each of the aliens in the sampling areas could be attributed to this.

Factors affecting species abundance and distribution

To test whether the distribution of alien fishes within the Wilderness Lakes system was influenced by certain biotic and abiotic factors a linear regression was used. Abundance and distribution (using CPUE) of *G. affinis* was determined from a once-off scoop netting survey, as described in Chapter 2. Distribution and abundance data were obtained from gill net, seine net, fyke net sampling and angling for *O. mossambicus* and *C. carpio*. The number of species sampled and the Shannon-Wiener Diversity index (methods described in Chapter 6) for each sampling area were correlated to the CPUE of *O. mossambicus* and *G. affinis* to determine whether native species diversity affected the distribution and abundance of the alien species. To assess the relationship between alien fish abundance and the physical environment, a linear regression using non-standardized data between CPUE of *O. mossambicus, G. affinis* and total number of *C. carpio* and each of the physico-chemical parameters (temperature, salinity, conductivity, dissolved oxygen and pH) was used.

Meta analysis of the environmental tolerances of the alien species

To test the suitability of the environment in the Wilderness Lakes system for the alien species, their physiological tolerances were reviewed and compared to the physiological environment in the system.

Results of meta analysis

Oreochromis mossambicus

The environmental tolerances of *O. mossambicus* are summarised in **Table 5.1**. This species prefers water temperatures of 15 – 40^{o}C, but grows optimally at 25 – 37^{o}C (Jobling 1981). Feeding ceases below 16^{o}C and reproduction only takes place at temperatures above 20^{o}C (Costa-Pierce 2003). Onset of cold stress has been reported at water temperatures of 15^{o}C and reported lethal minimum temperatures ranged between 5.5 and 12^{o}C (Costa-Pierce & Rackocy 2000). At temperatures below 15^{o}C, *O. mossambicus* have increased susceptibilities to disease, parasites and predators (Costa-Pierce & Rackocy 2000). In Port Elizabeth, South Africa, the lower lethal temperature has been recorded as 10.5^{o}C (Allanson *et al*. 1971).

Table 5.1. Physiological tolerance range of *Oreochromis mossambicus* for various physico-chemical parameters.

Parameter	Tolerance	Source
Temperature	Lower limit: 10.5°C Optimal range: 10 – 38°C Upper limit: 40°C	Allanson *et al* 1971;Chmilevskii 1998; Jobling 1981
Salinity	Optimal range: 6 – 14 ‰ Upper limit: 120 ‰	Jobling 1981; Whitfield & Blaber 1979
Oxygen	Lower limit – 2.5 mg/l	De Moor & Bruton 1988
pH	Lower limit: below 3 Optimal: above 4	Villegas 1990

Oreochromis mossambicus is the most tolerant of the tilapiine fishes (Whitfield & Blaber 1979; Costa-Pierce & Rackocy 2000) and it adapts especially well to saline environments, particularly where salinity changes gradually (Chmilevskii 1998; Costa-Pierce 2003). Under experimental conditions, *O. mossambicus* can tolerate 59 ‰ (Fiess *et al.* 2007) and in estuarine environments in South Africa it has been reported to tolerate a wide range of salinities from 0 – 120 ‰ (Whitfield & Blaber 1979). Optimal growth rates for are between salinities of 6 and 14 ‰ (de Moor & Bruton 1988). Increased salinities, may cause rapid reductions in fish growth rates (Costa-Pierce 2003). *Oreochromis mossambicus* may avoid stressful salinities by moving to more osmotically favourable areas (Costa-Pierce & Rackocy 2000).

Oreochromis mossambicus tolerates low dissolved oxygen levels (minimum of 2.5 mg/l) (McVeigh 1980; de Moor & Bruton 1988). Acute exposure to water of pH 3 proved to be lethal for *O. mossambicus*. In the field, *O. mossambicus* rarely occur in water with a pH of 4 or lower (Villegas 1990).

Cyprinus carpio

The environmental tolerances of *C. carpio* are summarised in **Table 5.2**. The thermal preference for *C .carpio* is between 12 and 38°C but it can tolerate temperatures as low as 0.5°C and as high as 35°C (de Moor 1996). The upper lethal temperature of *C. carpio* is 40.6°C (Jobling 1981).

Table 5.2. Physiological tolerances of *Cyprinus carpio* for various physico-chemical parameters.

Parameter	Tolerance	Source
Temperature	Lower limit:0.5°C Optimal range: 12 – 38°C Upper limit: 40.6°C	Jobling 1981; De Moor 1996
Salinity	Lower limit: 0.0 ‰ Optimal range: 0.5 – 2.5 ‰ Upper limit: 14.5 ‰	Wang *et al*. 1997
Oxygen	7% saturation	Koehn 2004
pH	Lower limit: 5.2 Optimal range: 5.5 – 10.5	Koehn 2004; Oyen *et al*. 1991

Cyprinus carpio is widely regarded as a freshwater fish with a high growth rate in waters with salinities of 0.5 ‰ or less (Wang *et al*. 1997). The optimal salinity range is 0.5 – 2.5 ‰ but it can tolerate salinities between 7 and 10 ‰ (Wang *et al.* 1997). Studies have shown that *C. carpio* has a lower oxygen consumption rate under 3 ‰ because it expends less energy on maintaining metabolic equilibrium *(*Wang *et al.* 1997). Salinity levels above 10 ‰ have been shown to result in clear unfavourable effects with lower growth rates and increased mortality (De Boeck *et al*. 2000). Wang *et al*. (1997) showed that the growth rate of *C. carpio* was highest in freshwater and decreased with increasing salinity. At salinities of 10.5 ‰, fingerlings had poor growth and became emaciated. At 12.5 ‰ and 14.5 ‰ it could survive for only eight and five days, respectively (Wang *et al*. 1997). The overall salinity tolerance of fingerlings was found to be 10.5 ‰ with gradual increases in daily salinity (Wang *et al*. 1997). Salinities above 6 ‰ were found to be have a negative impact on egg survival and hatch rates (Lam & Sharma 1985).

The pH tolerance of this species is between 5.0 and 10.5 (Koehn 2004) (**Table 5.2**). Oyen *et al*. (1991) noted that pH values below 5.2 were lethal to *C. carpio* embryos. At pH 5.5, embryonic development decreases, higher mortality percentage, delayed hatching time of eggs and increased deformation of juvenile fish. *Cyprinus carpio* can tolerate levels of dissolved oxygen as low as 7% saturation (Koehn 2004) (**Table 5.2**). When dissolved oxygen levels become too low, *C. carpio* have been reported to gulp air at the water surface (de Moor 1996).

Gambusia affinis

The physiological tolerances of *G. affinis* are summarised in **Table 5.3**. *Gambusia affinis* has a wide temperature tolerance, being able to withstand temperatures as low as 0.5°C and as high as 43.7°C (Pyke 2005; Nordlie 2006). The metabolic rate for stationary *G. affinis* has been found to increase with increasing temperature (from 10 to 35°C) which would indicate its thermal preference (Cech *et al*. 1985) (**Table 5.3**).

Table 5.3. Physiological tolerances of *Gambusia affinis* for various physico-chemical parameters.

Parameter	Tolerance	Source
Temperature	Lower limit: 0.5°C Optimal range: 10 - 35°C Upper limit: 43.7°C	Jobling 1981; Cech *et al.* 1985; Nordlie 2006
Salinity	Lower limit: 0 ‰ Optimal range: 0 – 35‰ Upper limit: 41 ‰	Chervinski 1983; Pyke 2005, 2008; Uliano *et al* 2010
Oxygen	Lower limit: 0.2 mg/l Optimal range: 1 – 11 mg/l	Odum & Caldwell 1955; Pyke 2008
pH	Optimal range: 4.9 – 9.0	Pyke 2008

Gambusia affinis tolerate salinities of 35 ‰ (Uliano *et al.* 2010) however, populations have been found at salinities as high as 41 ‰ (de Moor & Bruton 1988). *Gambusia affinis* are known to live at low salinities (< 15 ‰) and it has been demonstrated that they tolerate acute transfer from fresh water to 19.5 ‰ (Chervinski 1983) and a chronic exposure (7 days) to salinity levels of 39 ‰ and 58.5 ‰ (with survival rate of 65% and 50%, respectively) (Chervinski 1983; Uliano *et al.* 2010).

Gambusia affinis are able to withstand exceptionally low dissolved oxygen levels. In a Florida spring, the fish were found in the most hypoxic portion of the spring where oxygen levels were between 0.2 and 0.3 mg/l (Odum & Caldwell 1955). While *G. affinis* can withstand very low oxygen levels, this is only the case if fish have access to the surface of the water where they can gulp air (Cech *et al.* 1985). The preferred dissolved oxygen range is between 1 and 11 mg/l (Pyke 2008).

Micropterus salmoides

The environmental tolerances of *M. salmoides* are summarised in **Table 5.4**. *Micropterus salmoides* can tolerate temperatures between 5 and 36.5°C with an optimum temperature of 28°C (Jobling 1981; de Moor & Bruton 1988). This species becomes inactive at temperatures below 10°C and temperatures below 14.4°C are lethal to incubating eggs (Hey 1971; de Moor & Bruton 1988). Currie *et al*. (1998) determined the upper lethal limit of *M. salmoides* to be 38.5°C (**Table 5.4**).

Table 5.4. Physiological tolerance range for *Micropterus salmoides* for various physico-chemical parameters.

Parameter	Tolerance	Source
Temperature	Lower limit: 5°C Optimal range: 15 – 36.5°C Upper limit: 38.5°C	Jobling 1981; de Moor & Bruton 1988; Currie *et al.* 1998
Salinity	Optimal range: 0 – 8 ‰ Upper limit: 12 ‰	De Moor & Bruton 1988; Meador & Kelso 1990; Norris 2007
Oxygen	Lower limit- 0.92mg/l at 25°C Optimal range 4 – 8 mg/l	Moss & Scott 1961; Stuber *et al.* (1982)
pH	Optimal range: 6.5 – 8.5	Stuber *et al.* (1982)

Juvenile *M. salmoides* in coastal populations are able to tolerate salinities up to about 10 ‰ by increasing metabolic rates while maintaining osmotic pressure within the range of freshwater conditions (Susanto & Peterson 1996). Juvenile fish appear to tolerate salinities better than adult fish, especially when salinities increase towards 12 ‰ (Norris 2007) (**Table 5.4**). There is an apparent ontogenetic pattern in salinity tolerance with age/size which indicates that as fish get older they become less tolerant of higher salinity (approaching 12 ‰) (Susanto & Peterson 1996). Adult and juvenile *M. salmoides* from both coastal and freshwater populations appear to prefer less than 3‰ salinity (Meador & Kelso 1989; Norris 2007). At salinities above 3 - 4.5 ‰, reproduction in *M. salmoides* ceases (Meador & Kelso 1989). *Micropterus salmoides* are thought to osmoregulate more efficiently at salinities lower than 4 ‰ due to reduced energy costs, which may contribute to preferences for lower salinity habitats (Meador & Kelso 1990). Adult *M. salmioides* held at varying salinities have shown a limited tolerance for salinities over 8 ‰, with mortality occurring within 120 days of salinity held constant at 12 ‰ (Meador & Kelso 1990).

The critical dissolved oxygen levels for *M. salmoides* were determined by Moss & Scott (1961). They found the critical oxygen levels at 25°C, 30°C and 35°C to be 0.92, 1.19 and 1.40 mg/l respectively. Optimal growth occurs in areas when dissolved oxygen levels are higher than 8 mg/l, with reductions in growth below 4 mg/l (Stuber *et al.* 1982). Low oxygen concentrations are known to cause high mortality among *M. salmoides* embryos and larvae (Spoor 1977). *Micropterus salmoides* can only reproduce when pH is between 5 and 10, but the optimal pH is between 6.8 and 8.5 (Stuber *et al.* 1982) (**Table 5.4**).

RESULTS

Biotic factors affecting abundance

The relationship between fyke net biomass CPUE and the Shannon-Wiener diversity index sampled in each of the sampling areas was negatively correlated with the biomass CPUE in the fyke net (r^2 = 0.8, d.f. 1, 4, P = 0.02). The CPUE was higher in areas with a lower diversity index (**Table 5.5**). There was no relationship between the fyke net biomass CPUE of *O. mossambicus* and the number of species sampled in each sampling area (r^2 = 0.2, d.f. 1, 4, P = 0.43) (**Table 5.5**). There was no relationship (r^2 = 0.2, d.f. 1, 3, P = 0.5) between the gill net biomass CPUE and the number of species sampled (**Table 5.5**). There was no relationship (r^2 = 0.2, d.f. 1, 3, P = 0.52) between the Shannon-Wiener diversity index and the gill net CPUE for *O. mossambicus*, however, where diversity increased so did CPUE.

Table 5.5. Relationship between the number of species sampled and the Shannon-Wiener diversity index and the fyke net and gill net CPUE of *Oreochromis mossambicus* sampled in the Wilderness Lakes system.

		r^2	F	d.f.	P - value
Fyke net CPUE (kg.net.night^{-1})	Shannon-Wiener diversity index	0.8	12.7	1, 4	0.02
	Species sampled	0.2	0.8	1, 4	0.43
Gill net CPUE (kg.net.night^{-1})	Shannon-Wiener diversity index	0.2	0.5	1, 3	0.52
	Species sampled	0.2	0.6	1, 3	0.50

The relationship between CPUE of *G. affinis* and the fish biota in the Wilderness Lakes system is shown in **Table 5.6**. There was no relationship between the diversity in the Wilderness Lakes system and the CPUE of *G. affinis*. Neither the number of species sampled (r^2 = 0.3, d.f. 1, 4, P = 0.24) nor the Shannon-Wiener Diversity index (r^2 = 0.001, d.f. 1, 4, P = 0.96) was affected by increasing *G. affinis* abundance.

Table 5.6. Relationship between the number of species sampled and the Shannon-Wiener diversity index and scoop net data of *Gambusia affinis* sampled in the Wilderness Lakes system.

		r^2	F	d.f.	P - value
Scoop net CPUE (g/site^{-1})	Shannon-Wiener diversity index	0.001	0.002	1, 4	0.96
	Species sampled	0.3	1.9	1, 4	0.24

The relationship between the total number of *C. carpio* sampled and the native fish biota in the Wilderness Lakes system is shown in **Table 5.7**. There was no relationship between the number of species in the Wilderness Lakes system and the abundance of *C. carpio* (r^2 = 0.001, d.f. 1, 3, P = 0.36). The Shannon-Wiener Diversity index (r^2 = 0.3, d.f. 1, 3, P =0.97) decreased with an increase in the number of *C. carpio* sampled, but the sample size was between two and five fish.

Table 5.7. Relationship between the number of species sampled and the Shannon-Weiner diversity index and the total number of *Cyprinus carpio* sampled in the Wilderness Lakes system.

		r2	F	d.f.	P - value
Total fish sampled	Shannon-Weiner diversity index	0.3	1.1	1, 3	0.36
	Species sampled	0.001	0.002	1, 3	0.97

Physico-chemical parameters

A summary of the physico-chemical parameters measured in the Wilderness Lakes system from April 2010 to May 2011 is given in **Table 5.8**. The temperature ranged from a minimum of 12.1°C in Rondevlei to 30°C in Rondevlei (**Table 5.8**). The salinity in the system ranged from a minimum of 0.1 ‰ in the Touw Estuary to 12.0 ‰ in Island Lake. Langvlei (10.2 ‰) had the highest mean salinity followed by Island Lake (9.78 ‰) (**Table 5.8**). The Touw Estuary had the lowest mean salinity at 2.76 ‰ (**Table 5.8**). The mean pH was lowest in the Touw Estuary (pH 7.07) and highest in Rondevlei (pH 8.45) (**Table 5.8**). Conductivity, which is auto-correlated to salinity, ranged from 0.01 ms in the Touw Estuary to 19.83 ms in Island Lake (**Table 5.8**). Dissolved oxygen levels were lowest in the Rondevlei-Langvlei channel (0.78 mg/l) and highest in Rondevlei (15.57 mg/l).

Table 5.8. Summary statistics of physico-chemical parameters for the Wilderness Lakes system sampled through April 2010 to May 2011.

		Touw Estuary	Serpentine channel	Island Lake	Langvlei	R-L channel	Rondevlei
		N = 17	N = 9	N = 33	N = 21	N = 15	N = 28
	Min	14.0	13.6	13.0	13.7	13.9	12.1
Temperature (°C)	Mean	19.9	20.3	19.9	20.8	21.0	19.7
	Max	25.5	25.8	25.9	28.4	26.5	30.0
	Min	0.1	0.4	7.4	9.9	4.1	8.8
Salinity (‰)	Mean	2.8	5.4	9.8	10.2	7.9	9.1
	Max	6.2	8.9	12.0	10.6	10.7	10.1
	Min	4.1	5.7	7.6	7.0	6.3	7.9
pH	Mean	7.1	7.4	8.1	8.0	7.7	8.5
	Max	8.4	7.9	9.9	8.4	8.8	8.9
	Min	0.0	0.1	12.9	14.2	7.4	13.5
Conductivity (ms)	Mean	4.4	10.6	15.9	17.0	13.5	15.5
	Max	10.0	17.8	19.8	18.0	18.2	17.2
	Min	4.4	5.1	6.8	7.4	0.8	6.6
Oxygen (mg/l)	Mean	8.0	7.0	9.4	9.0	5.2	10.4
	Max	10.5	10.3	13.6	10.6	12.3	15.6

Cluster analysis (**Fig. 5.1)** demonstrated that Island Lake and Rondevlei had the most similar water quality with 96.5% similarity. The three lakes were 95.6% similar, with the Rondevlei-Langvlei channel being 91.0% similar to the three lakes. The two channels (Rondevlei-Langvlei channel and the Serpentine channel) were 88.5% similar. The Touw Estuary was the least similar to all the sampling areas with only 81.5% similarity (**Fig. 5.1**).

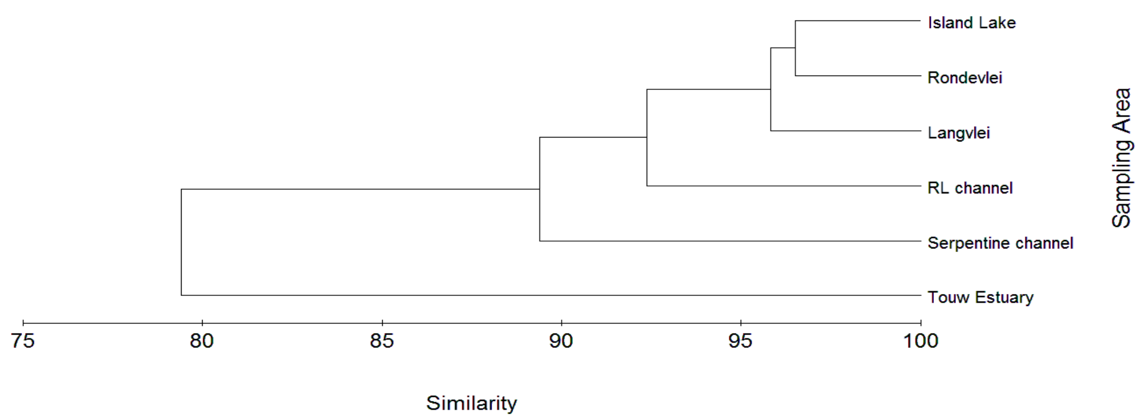


Figure 5.1. Hierarchical dendrogram showing group average clustering of Bray-Curtis indices of similarity based on the non-transformed data (water temperature, pH, salinity, conductivity and dissolved oxygen) for the six sampling areas found in the Wilderness Lakes system. RL channel = Rondevlei-Langvlei Channel.

There were significant differences in the recorded seasonal temperature in Rondevlei, Rondevlei-Langvlei channel, Langvlei, Island Lake, the Serpentine channel and the Touw Estuary (**Fig. 5. 2, 5.3, 5.4**). In all sampling areas in the system, water temperatures were significantly higher in summer than winter.

In Rondevlei, winter temperatures (14.7 ± 0.4°C) were lower than the autumn and summer temperatures (20.4 ± 0.6°C and 26.7 ± 0.8°C respectively) and summer was significantly higher than winter and spring (19.1 ± 0.7°C). In Langvlei, summer and spring (27.0 ± 0.6°C and 23.0 ± 0.0°C respectively) were significantly warmer than winter (15.0 ± 0.3°C). In Island Lake and the Touw Estuary, summer temperatures were significantly higher than in autumn and winter (Kruskal-Wallis ANOVA, $H = 28.7$ and 13.3 respectively; $p < 0.05$).

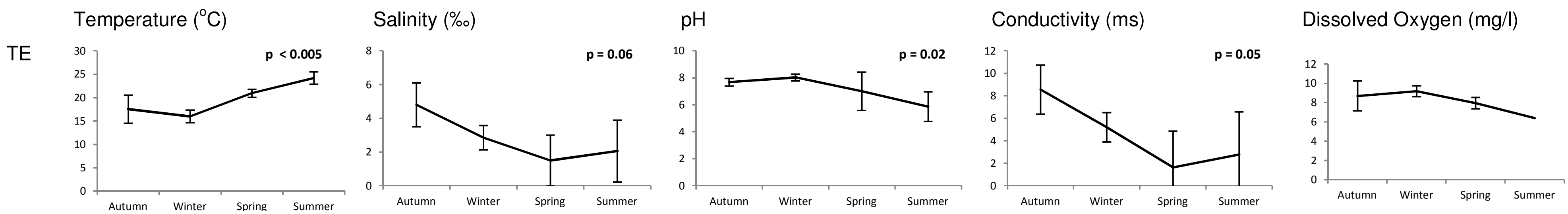


Figure 5. 2. The seasonal averages for temperature, salinity, pH, conductivity and dissolved oxygen in the Touw Estuary (TE), Wilderness Lakes system measured between April 2010 and May 2011. Error bars denote standard deviation.

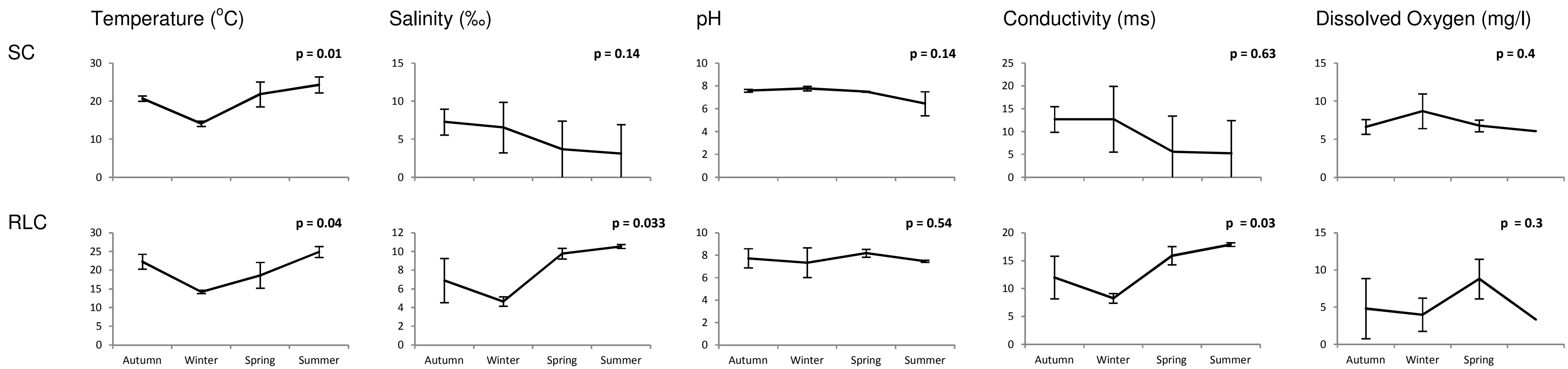


Figure 5. 3. The seasonal averages for temperature, salinity, pH, conductivity and dissolved oxygen in the Serpentine channel (SC) and the Rondevlei-Langvlei channel (RLC), Wilderness Lakes system measured between April 2010 and May 2011. Error bars denote standard deviation.

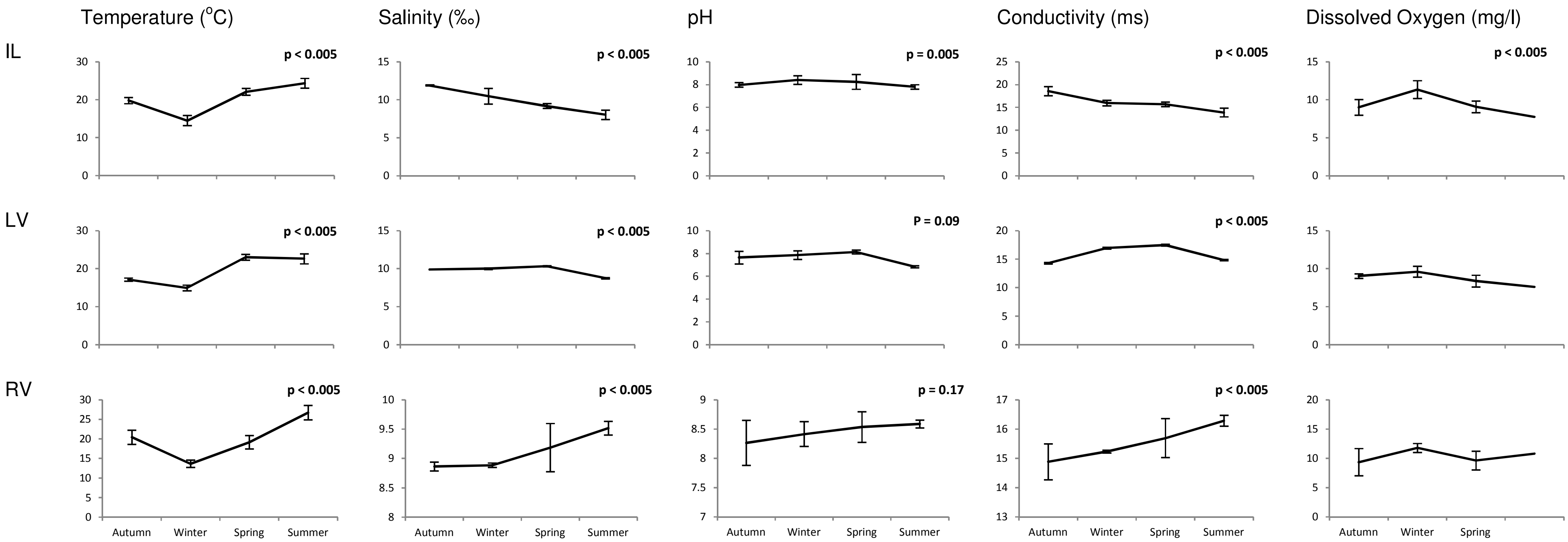


Figure 5. 4. The seasonal averages for temperature, salinity, pH, conductivity and dissolved oxygen in Island Lake (IL), Langvlei (LV) and Rondevlei (RV) Wilderness Lakes system measured between April 2010 and May 2011. Error bars denote standard deviation.

In the Touw Estuary, Serpentine channel and the Rondevlei-Langvlei channel there were no significant differences between seasonal salinities (Kruskal-Wallis ANOVA, $p < 0.05$) (**Fig. 5.2, 5.3**). In Rondevlei, the salinity in summer (9.5 ± 0.04 ‰) was significantly higher than in autumn and winter (8.9 ± 0.03 ‰ and 8.9 ± 0.01 ‰) and spring was significantly more saline than autumn (Kruskal-Wallis ANOVA, $H = 21.5$; $p < 0.05$) (**Fig. 5.4**). In Langvlei, summer was significantly more saline (10.5 ± 0.03 ‰) than autumn and winter (9.0 ± 0.0 ‰ and 10.0 ± 0.04 ‰).

There were no seasonal differences in pH in the Serpentine channel, Langvlei, Rondevlei-Langvlei channel and Rondevlei (Kruskal-Wallis ANOVA, $p > 0.05$) (**Fig. 5.3, 5.4**). In the Touw Estuary, winter had a higher pH than summer (Kruskal-Wallis ANOVA, $H = 9.58$; $p < 0.05$).

There were no seasonal differences in conductivity in the Serpentine channel and the Rondevlei-Langvlei channel (**Fig 5.3**). In Rondevlei, conductivity was higher in summer than in autumn and winter (Kruskal-Wallis ANOVA, $H = 22.29$; $p < 0.05$). In Langvlei, conductivity was significantly lower in winter and autumn (17.0 ± 0.07 ms and 14.3 ± 0.08 ms) than in summer (17.8 ± 0.06 ms). As with Rondevlei, conductivity in Island Lake was higher in summer than autumn and winter (Kruskal-Wallis ANOVA, $H = 26.03$; $p < 0.05$). In the Touw Estuary, conductivity in autumn (8.56 ± 1.1 ms) was significantly higher than in spring (1.64 ± 1.6 ms) (**Fig. 5.4**).

There were significant seasonal variations in dissolved oxygen only in the Touw Estuary and Island Lake (**Fig. 5.3**). In the Touw Estuary, the dissolved oxygen in winter (9.2 ± 0.3 mg/l) was significantly higher than in summer (6.4 ± 0.6 mg/l). In Island Lake, the dissolved oxygen levels were significantly higher than in autumn, spring and summer (Kruskal-Wallis ANOVA, $H = 23.1$; $p < 0.05$) (**Fig. 5.3**).

There were no significant differences between temperatures in the Wilderness Lakes system (**Table 5.9**). Langvlei had the highest salinity (10.2 ± 0.05 ‰) and the Touw Estuary the lowest salinity (2.8 ± 0.05 ‰). There were significant differences in salinity. The lowest mean salinity was recorded in the Touw Estuary (2.8 ± 1.8 ‰) and highest in Langvlei (10. 10.2 ± 0.2 ‰) (**Table 5.9**). Conductivity was highest in Langvlei (16.96 ± 0.25 ms) and lowest in the Touw Estuary (4.4 ± 0.9 ms), with no significant differences between Langvlei, Island Lake, the Rondevlei-Langvlei channel and Rondevlei. Dissolved oxygen was significantly higher in Rondevlei, Langvlei and Island Lake, with values between 10.33 ± 0.4 mg/l in Rondevlei and 8.9 ± 1.9 mg/l in Langvlei. Dissolved oxygen was the lowest in the Rondevlei-Langvlei channel (5.2 ± 0.9 mg/l) (**Table 5.9**).

Table 5.9. Characteristics of the non-parametric Kruskal-Wallis ANOVA of the differences between physico-chemical parameters in the Touw Estuary, Serpentine channel, Island Lake, Langvlei, Rondevlei-Langvlei channel and Rondevlei, Wilderness Lakes system, Western Cape.

	H-value	d.f.	p - value
Temperature (°C)	2.1	5	$p = 0.83$
Salinity (‰)	71.5	5	$p < 0.005$
pH	48	5	$p < 0.005$
Conductivity (ms)	55.8	5	$p < 0.005$
Dissolved oxygen (mg/l)	38.5	5	$p < 0.005$

Effects of physico-chemical parameters on abundance and distribution

Oreochromis mossambicus

The relationship between the abundance and distribution of *O. mossambicus* from gill net sampling and temperature, salinity, pH, conductivity and dissolved oxygen are shown in **Table 5.10**. Temperature ($r^2 = 0.001$, d.f. 1, 22, $P = 0.9$), salinity ($r^2 = 0.1$, d.f. 1, 22, $P = 0.6$), pH ($r^2 = 0.1$, d.f. 1, 22, $P = 0.1$) and conductivity ($r^2 = 0.1$, d.f. 1, 22, $P = 0.1$) showed no relationship to the gill net CPUE of *O. mossambicus*. Dissolved oxygen exhibited a positive relationship ($r^2 = 0.02$, d.f. 1, 22, $P = 0.09$) with CPUE increasing as oxygen concentrations increased.

There was no relationship between salinity ($r^2 = 0.01$, d.f. 1, 22, $P = 0.62$), temperature ($r^2 = 0.01$, d.f. 1, 22, $P = 0.7$) pH ($r^2 = 0.02$, d.f. 1, 22, $P = 0.5$) and conductivity ($r^2 = 0.02$, d.f. 1, 22, $P = 0.6$) and the abundance of *O. mossambicus* sampled in the fyke nets (**Table 5.10**). Again the only physico-chemical parameter which may have affected distribution was dissolved oxygen ($r^2 = 0.1$, d.f. 1, 22, $P = 0.09$) where as dissolved oxygen increased so did the abundance of *O. mossambicus*.

Table 5.10. Characteristics of the linear regression of the differences between physico-chemical parameters and the fyke net and gill net CPUE (log adjusted) of *Oreochromis mossambicus* sampled from April 2010 to May 2011 from the Touw Estuary, Serpentine channel, Island Lake, Langvlei, Rondevlei-Langvlei channel and Rondevlei, Wilderness Lakes system.

Physico-chemical parameters	Sampling gear	r^2	F	d.f.	P - value
	Gill net	0.001	0.2	1, 22	0.9
Temperature (°C)	Fyke net	0.01	0.15	1, 22	0.7
	Gill net	0.12	3.1	1, 22	0.1
Salinity (‰)	Fyke net	0.01	0.26	1, 22	0.62
	Gill net	0.1	2.4	1, 22	0.1
pH	Fyke net	0.02	0.5	1, 22	0.5
	Gill net	0.1	3.1	1, 22	0.1
Conductivity (ms)	Fyke net	0.02	0.3	1, 22	0.6
	Gill net	0.2	5.4	1, 22	0.03
Dissolved oxygen (mg/l)	Fyke net	0.13	3.2	1, 22	0.09

Gambusia affinis

The relationship between the CPUE of *G. affinis* in each sampling area and temperature, salinity, pH and conductivity are shown in **Table 5.11**. The distribution of *G. affinis* was not affected by temperature (r^2 = 0.001, d.f. 1, 5, P = 0.94), salinity (r^2 = 0.3, d.f. 1, 5, P = 0.17) or conductivity (r^2 = 0.43, d.f. 1, 5, P = 0.11). The factor which affects the abundance of *G. affinis* is pH (r^2 = 0.64, d.f. 1, 5, P = 0.03) where as pH increased the abundance of *G. affinis* decreased. A multiple regression showed that environmental factors were important in determining the abundance of *G. affinis* in the Wilderness Lakes system, with pH being the most important determining factor.

Table 5.11. Characteristics of the linear regression of the differences between physico-chemical parameters and the scoop net CPUE of *Gambusia affinis* sampled in May 2011 from the Touw Estuary, Serpentine channel, Island Lake, Island Lake-Langvlei channel, Langvlei, Rondevlei-Langvlei channel and Rondevlei, Wilderness Lakes system.

	r^2	F	d.f.	P - value
Temperature (°C)	0.001	0.01	1, 5	0.94
Salinity (‰)	0.3	2.5	1, 5	0.17
pH	0.64	8.7	1, 5	0.03
Conductivity (ms)	0.43	3.8	1, 5	0.11

Cyprinus carpio

The very low abundance of *C. carpio* and wide range of sizes sampled in the Wilderness Lakes system was not conducive to an analysis of biomass distribution in relation to physico-chemical parameters. The number of fish sampled during a one year period using the standard sampling outlined in Chapter 3 ranged from two fish sampled in Rondevlei to five fish sampled in Island Lake and Langvlei. There was no relationship between the pH (r^2 = 0.2, d.f. 1, 3, P = 0.5), dissolved oxygen (r^2 = 0.1, d.f. 1, 3, P = 0.72), conductivity (r^2 = 0.1, d.f. 1, 3, P = 0.6) and salinity (r^2 = 0.2, d.f. 1, 3, P = 0.5) and the abundance of *C. carpio* (**Table 5.12**). Temperature showed no relationship (r^2 = 0.4, d.f. 1, 3, P = 0.2) with the number of fish sampled in each sampling area, however, as temperature increased there was an increase in CPUE (**Table 5.12**). Due to the very low number of fish sampled in each sampling area, these results give a very poor indication of the relationship between physico-chemical parameters and the abundance and distribution of *C. carpio* in the Wilderness Lakes system. However, temperature does appear to play a role in the abundance and distribution of *C. carpio* in the Wilderness Lakes system.

Table 5.12. Characteristics of the linear regression of the differences between physico-chemical parameters and the Total number of *Cyprinus carpio* sampled from April 2010 to May 2011 from the Touw Estuary, Serpentine channel, Island Lake, Langvlei and Rondevlei, Wilderness Lakes system.

	r^2	F	d.f.	P - value
Temperature (°C)	0.4	2.4	1, 3	0.22
Salinity (‰)	0.2	0.5	1, 3	0.5
pH	0.2	0.62	1, 3	0.5
Conductivity (ms)	0.1	0.4	1, 3	0.6
Dissolved oxygen (mg/l)	0.1	0.2	1, 3	0.72

Micropterus salmoides

The lack of *Micropterus salmoides* sampled from the system did not allow for investigation into the relationship between their abundance, distribution and the water quality in the system. Due to their environmental tolerances, it was assumed that this species would be limited by temperature and salinity in the Wilderness Lakes system.

Long term assessment of temperature

Long term recorded water temperature, which was measured quarterly, between January 1991 and January 2011 in the Wilderness Lakes system is shown in **Figure 5.5**. Temperatures showed an annual trend, being highest in summer and lowest in winter with very small fluctuations in temperatures between sampling areas. The maximum and minimum temperature recorded during the period was 27.3 and 8.3ºC respectively: both recorded in the Touw Estuary. During 1991 and 2011 the recorded temperature was always within the thermal preferences of *G. affinis*, *M. salmoides* and *C. carpio*. The temperature in each of the lakes was always within the thermal preference of *O. mossambicus*, however in the Touw Estuary the temperature was out of the thermal preference for 1.3% for the sampling period.

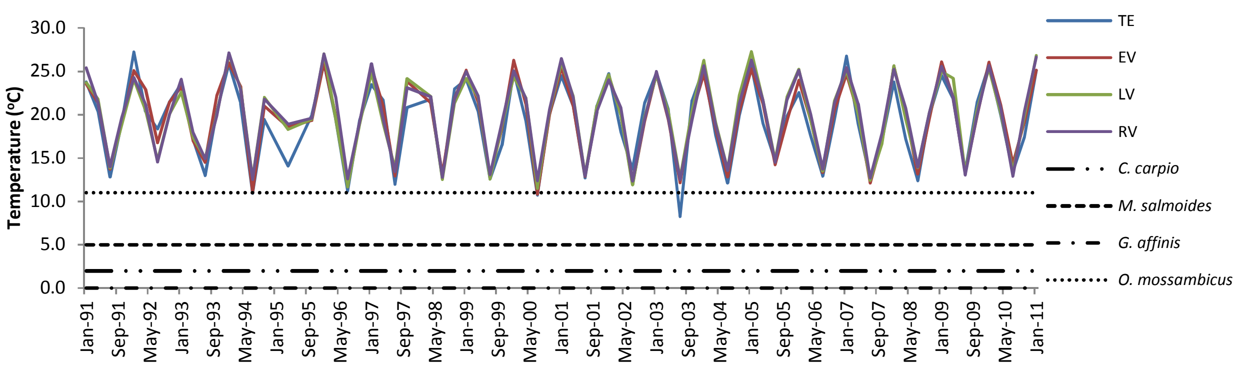


Figure 5.5. Long term temperature recordings for the Touw Estuary, Island Lake, Langvlei and Rondevlei, Wilderness Lakes system from January 1991 to January 2011 (SANParks unpublished data) with the lower temperature tolerance for *Oreochromis mossambicus*, *Gambusia affinis*, *Cyprinus carpio* and *Micropterus salmoides*.

Daily recordings of water temperatures for a one year period are shown in **Figure 5.6**. The maximum and minimum temperature was 27. 9 and 8.8ºC respectively. The temperatures in each of the sampling areas were within the thermal tolerances of *G. affinis*, *C. carpio* and *M. salmoides* for 100% of the sampling period. In the upper Touw Estuary the temperature dropped below the thermal preference for *O. mossambicus* in July 2010 for two days. Temperature was within *O. mossambicus* tolerance range for 99.5% of the sampling period (**Fig. 5.6**).

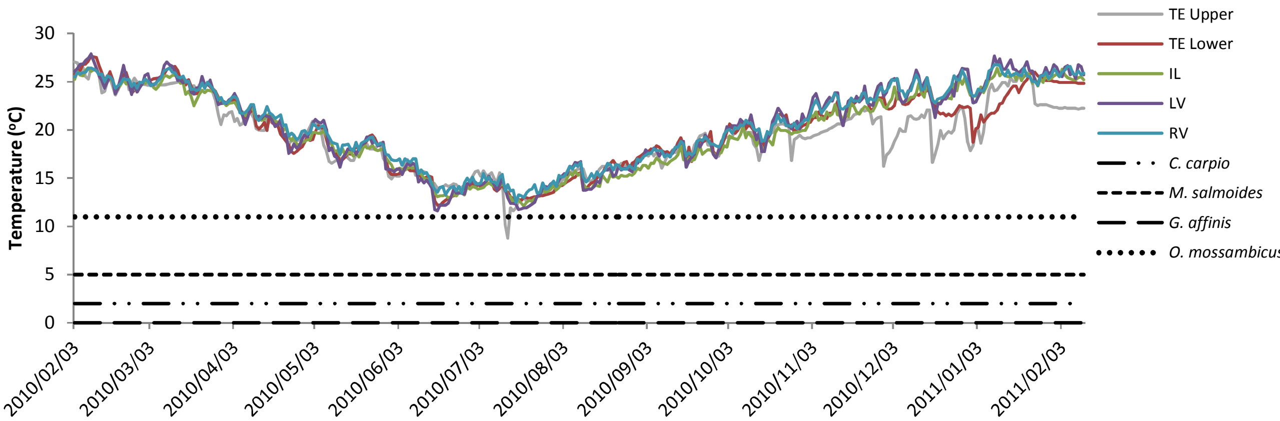


Figure 5.6. Daily recorded temperature for the upper and lower Touw Estuary, Island Lake, Langvlei,and Rondevlei, Wilderness Lakes system from 3 February 2010 to 11 February 2011 with the lower temperature tolerance for *Oreochromis mossambicus*, *Gambusia affinis*, *Cyprinus carpio* and *Micropterus salmoides*.

Long term assessment of salinity

For the long term trend in salinity, the Wilderness Lakes system did not exceed 36 ‰ from January 1991 to January 2011 in each of the lakes. The upper limit of *O. mossambicus* was therefore set at 45‰ because *O. mossambicus* can tolerate salinities well above that of sea water (approximately 35 ‰). For the duration of the sampling period, the tolerances of *G. affinis* and *O. mossambicus* were not exceeded and they could thus tolerate the salinity in the Wilderness Lakes system 100% of the sampling period in all the sampling areas (**Fig 5.7**). In Island Lake and Langvlei, *C. carpio* could tolerate the salinity for 100% analysis period. In Rondevlei, *C. carpio* could tolerate the salinity for 91.4% of the analysis period. The period that was above the tolerance range for *C. carpio* was between August 1991 and May 1993. *Micropterus salmoides* could tolerate the salinity in Island Lake for the majority of the period, but salinities were above the tolerance level between October 2000 and January 2003 and between April 2009 and January 2011 (**Fig. 5.7**). In Langvlei, the salinity was above the tolerance of *M. salmoides* from January 1991 to July 1994, July 2002 to January 2003 and January 2010 to January 2011. In Rondevlei, the salinity exceeded the tolerance of *M. salmoides* for 81% of the sampling period, with the salinities being tolerable only between October 2006 and October 2009 (**Fig. 5.7**).

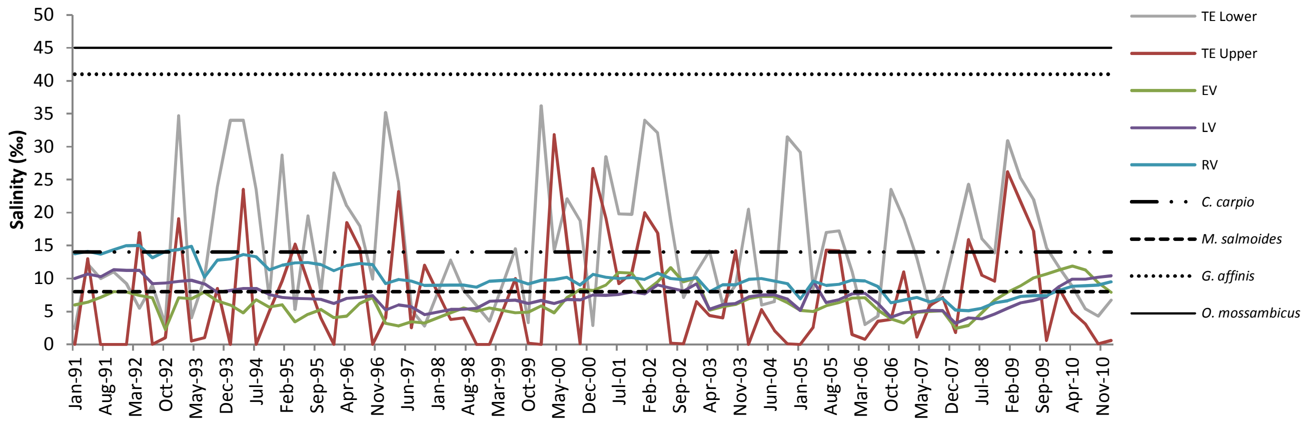


Figure 5.7. Long term salinity recordings for the upper and lower Touw Estuary, Island Lake, Langvlei and Rondevlei, Wilderness Lakes system, Western Cape measured from January 1991 and January 2011 and the upper salinity tolerance of *Oreochromis mossambicus*, *Gambusia affinis*, *Cyprinus carpio* and *Micropterus salmoides*.

The Touw Estuary was highly variable in terms of salinity fluctuations, due to mouth opening and closure. As within the lakes, *G. affinis* and *O. mossambicus* were able to tolerate the salinity in the Touw Estuary for 100% of the sampling period even during times of very high salinity. The salinity fluctuations in the Touw Estuary where *C. carpio* and *M. salmoides* could tolerate the salinity were relatively short due to mouth opening events. In the lower Touw Estuary, *C. carpio* could tolerate the salinity 68% of the analysis period (**Fig. 5.8**). As with the lower Touw Estuary, there are large fluctuations in salinity. In the upper Touw Estuary, *C. carpio* could not tolerate the salinity for 21% of the analysis period. *M. salmoides* could tolerate the salinity in the lower Touw Estuary for only 29% of the period January 1991 – January 2011. In the upper Touw Estuary, where salinities are lower, *M. salmoides* was able to tolerate the salinity for 64% of the sampling period (**Fig. 5.18**).

DISCUSSION

Effect of biotic factors on alien species

The native fish diversity did not appear to have an impact on the abundance of alien fishes in the Wilderness Lakes system. The abundance of *O. mossambicus* was highest in areas with the lowest diversity and *vice versa*, however the native species present in these areas were abundant (Chapter 3). While their abundance was highest in Langvlei, the number of invasive species in Langvlei was the same as in each of the lakes. The abundance of *G. affinis* was not impacted by the native diversity in the Wilderness Lakes system as they were highly abundant in the littoral regions of each of the sampling areas. The number of *C. carpio* sampled was not high enough to permit an assessment of whether they impact native biota.

There is no evidence to support the theory of biotic resistance in the Wilderness Lakes system. This is consistent with the analysis performed by Leprieur *et al*. (2008), who found no evidence to support biotic resistance in freshwater systems. Biotic resistance could not explain the pattern of fish invasions as there were no negative relationships between native and non-native species richness when environmental conditions were accounted for. This would indicate that species-rich environments might not act as a barrier against the establishment of non-native species (Leprieur *et al*. 2008). Thus, the native biota had little impact on the distribution and abundance of the alien fishes in the Wilderness Lakes system.

Biotic acceptance, which states that areas with higher diversity will have higher rates of invasion, is due to the abundance of natural resources. This allows invasion without negative impacts on the native biota thus allowing co-habitation between native and alien species (Leprieur *et al*. 2008; Richardson *et al*. 2000). Optimal environmental conditions will thus allow for successful invasion regardless of the native biota present. One can then deduce that the successful establishment of alien species did not rely on the native biota but on environmental conditions that would allow the alien species to persist and reproduce in the invaded system (Leprieur *et al*. 2008).

Effect of abiotic factors on alien species

The abiotic factors most often associated with the distribution and abundance of species in estuarine systems are temperature and salinity (Jaureguizar *et al*. 2004; Harrison & Whitfield 2006; Richardson *et al*. 2006). The four alien fishes present in the Wilderness Lakes system are generally considered to be freshwater species (de Moor 1996; Pyke 2005) however *O. mossambicus* does occur in estuaries in KwaZulu-Natal and the Eastern Cape, South Africa (de Moor & Bruton 1988; Skelton 2001; Ellender *et al*. 2008). While *O. mossambicus* does occur in estuarine systems, *C. carpio*, *M. salmoides* and *G. affinis* are less often associated with estuarine systems. Their ability to tolerate variable physico-chemical environments in estuarine systems may be the main factor determining the establishment success of these invasive species.

Oreochromis mossambicus

The abundance of *O. mossambicus* in the Wilderness Lakes system appeared to be negatively correlated to dissolved oxygen. The strong relationship seen was due to the high abundance of juvenile *O. mossambicus* in the Serpentine channel and the Rondevlei-Langvlei channel where dissolved oxygen was lowest. It is likely that the abundance of *O. mossambicus* in the channels was influenced more by shelter and predator avoidance (Grenouillet *et al*. 2002) than by oxygen levels in these areas, as this species can tolerate very low levels of dissolved oxygen (de Moor 1996).

Oreochromis mossambicus is well known for being euryhaline and able to tolerate salinities over three times that of sea water (Jobling 1981; Whitfield & Blaber 1979; Jamil *et al*. 2004; Fiess *et al*. 2007). Salinity would therefore not be a limiting factor in their establishment in a system where salinity reaches a maximum on 35 ‰, such as the Wilderness Lakes system.

After excluding dissolved oxygen levels, the environmental factor which would most probably affect *O. mossambicus* in the Wilderness Lakes system would be temperature, as its intolerance for low temperatures is well documented (Allanson *et al*. 1971; Chmilevskii 1998; Cnaani *et al*. 2000; Costa-Pierce & Rackocy 2000; Sardella *et al*. 2004; Fiess *et al.* 2007). The lower lethal temperature varies due to environmental interactions, the exposure history of individual fish and genetic effects (Cnaani *et al*. 2000). *Oreochromis mossambicus* occur naturally from the warmer lower Zambezi river (ranges from 18 – 30^{o}C) (Hall *et al*. 1977) to the Bushman's River (James *et al*. 2007) in the Eastern Cape where temperatures rarely reach to the lower temperature tolerances of this species. Allanson *et al*. (1971) described a chill coma in *O. mossambicus* in freshwater at temperatures of 11^{o}C.

The almost negligible relationship between CPUE and temperature would indicate that temperature does not impact *O. mossambicus* in the Wilderness Lakes system. However, there was very high variation in the data where there were no fish sampled during winter and spring and very high CPUEs in autumn and summer. This resulted in non-normal distributions and inaccurate representations of the relationship between CPUE and the physical environment. There were however, differences in seasonal CPUE as was shown in Chapter 3, where CPUE decreased significantly in winter and spring. This may give an indication that temperature does influence the abundance and distribution of fish in the lakes.

Fish may have either succumbed to the lower temperatures or moved into deeper water in winter and spring. *Oreochromis mossambicus* can tolerate lower temperatures coupled with higher salinities, and the ability to tolerate lower temperatures could be aided by higher salinities in the system (Allanson *et al*. 1971). This shows that the factor that most affects *O. mossambicus* is temperature, which is closely linked to salinity.

Gambusia affinis

Gambusia affinis can tolerate a very wide range of environmental parameters which makes it a highly efficient invasive species (Pyke 2005; Pyke 2008). It is classified as eurythermal, euryhaline and can tolerate extreme environmental conditions (Cech *et al*. 1985). The relationship between the physico-chemical parameters and the abundance of *G. affinis* in the Wilderness Lakes system showed almost no relationship between temperature, salinity, dissolved oxygen and pH and the abundance of fish. The long term salinity and temperature recordings showed that these factors did not go above or below the tolerances of this species. Temperature and salinity would therefore not limit its establishment (Chervinski 1983; Pyke 2005; Uliano *et al.* 2010). Due to its hardy nature and wide environmental tolerances, *G. affinis* can tolerate and even thrive in environments lethal to many other species, it is highly adaptable in its diet and there are very broad habitat requirements for reproduction (Daniels & Felley 1992). The plasticity exhibited by this species has made it widely distributed in the Wilderness Lakes system, even in areas that experience high environmental fluctuations, primarily in salinity in the lower Touw Estuary (**Fig 5.7**). It appears that environmental factors do not limit the establishment of *G. affinis*.

Cyprinus carpio

The relationship between *C. carpio* abundance and temperature, salinity, pH, conductivity and dissolved oxygen was inconclusive due to the low number of fish sampled in each area. However, from the recorded salinity and temperature, it could be that salinity may be limiting the abundance and distribution and therefore establishment in the Wilderness Lakes system as they will not be able to breed successfully. The temperatures and dissolved oxygen in the Wilderness Lakes system were all within the tolerance range of *C. carpio*. The lower tolerance to pH of *C. carpio* larvae is 5.2 and embryonic development is delayed with increased mortality at pH 5.5 (Oyen *et al*. 1991). The pH in the Touw Estuary dropped to 4.14 in December 2010 (**Table 5.5**), which would reduce the chance of successful spawning for *C.*

carpio in the Touw Estuary. In the rest of the Wilderness Lakes system, pH was within *C. carpio* tolerance of between 5.5 and 10.5.

Cyprinus carpio is considered a stenohaline freshwater fish, but it displays a certain tolerance to increased salinity (De Boeck *et al.* 2000). Dilute seawater (0.3‰ – 3.0‰) has been reported to enhance survival, growth, and development of the larvae. However, exposure of larvae to saltwater (10‰) has been shown to result in lower growth rates and increased mortality (De Boeck *et al*. 2000). In the Wilderness Lakes system, salinity fluctuated during periods of mouth opening and closure. During the periods of higher salinity, concentrations were above the salinity tolerance of *C. carpio*. In Island Lake and Langvlei, the salinity levels were always within the tolerance of *C. carpio*, which could explain the higher abundances at these sites. With the long term reverse salinity gradient, the abundance of *C. carpio* may decrease towards Rondevlei, as the salinity was more frequently above the tolerance range (**Fig. 5.7**).

De Boeck *et al*. (2000) showed that exposure to saltwater in *C. carpio* results in decreased food intake, which in turn affects the energy metabolism of fish. The ability to spawn may be limited by the salinity in the Wilderness Lakes system. The optimal temperature for *C. carpio* egg development, determined by Lam & Sharma (1985) was between 0.3 ‰ and 3.0 ‰. While salinities between 6 ‰ and 9 ‰ were optimal for egg development these salinities were detrimental to egg hatching (Lam & Sharma 1985). With salinity levels being consistently higher than 6 ‰ in each of the lakes, *C. carpio* may be limited to spawning only in the Touw Estuary and Serpentine channel during periods of mouth closure. Juvenile fish caught in the Touw Estuary and Serpentine channel during 2010 and early 2010 may be an indication of spawning in these areas during mouth closure (**Chapter 2; Table 2.1**). While adult fish may be able to tolerate the salinity in the Wilderness Lakes system, successful spawning may rely on salinities low enough for larvae to develop. The dispersal of fish throughout the system does not appear to be limited by salinity as *C. carpio* were sampled in each of the lakes. However, juveniles were only sampled in the Serpentine channel and the Touw Estuary, and while spawning may occur throughout the system, the higher salinities may limit the chance of successful development of eggs.

Micropterus salmoides

The lack of samples of *M. salmoides* in the Wilderness Lakes system did not allow for investigation into the relationship between water chemistry and abundance of fish. From literature reviews, the ranges for pH, dissolved oxygen and temperature in the Wilderness Lakes system were all within the tolerance range for *M. salmoides*. While temperatures may be within the tolerance range of adult fish, growth and survival of juveniles may be limited by lower temperatures

According to literature, salinity might be the only physico-chemical parameter inhibiting establishment in the Wilderness Lakes system. This species is stenohaline with very limited tolerance for salinity. Temporal and spatial variation in salinity may be stressful to fishes (Lowe *et al.* 2009). Fish found in regions of higher salinity have been reported to be slower growing and smaller than fish occupying fresher waters (Meador & Kelso 1990). The salinity tolerance range of *M. salmoides* is relatively narrow, with the upper lethal level being 12 ‰ (Norris 2007). The salinity recordings for the Wilderness Lakes system showed that the ideal area for this species would be in Island Lake. While there were periods of salinity tolerance in the upper Touw Estuary, the wide fluctuations in salinity would not be ideal for *M. salmoides*. As with *C. carpio*, the reverse salinity gradient exhibits less desirable areas of occupation as fish move higher up the system. In Rondevlei the salinity was in the tolerance range of *M. salmoides* for a period of only two out of twenty years (**Fig. 5.7**). The ability of *M. salmoides* to reproduce ceases between 3 and 4.5 ‰ (Meador & Kelso 1989) which would indicate that this species cannot reproduce in the Wilderness Lakes system even during periods of low salinity, explaining the low abundance. Reproduction in the Duiwe River or from dams upstream is therefore the most likely source of *M. salmoides* for the Wilderness Lakes system.

Conclusion

The establishment success of the alien fish in the Wilderness Lakes system appears to be mostly affected by at least one physico-chemical parameter. The species diversity in each of the water bodies did not appear to have an effect on the alien species which indicated biotic acceptance. *Oreochromis mossambicus* and *C. carpio* are associated with estuaries and they have the ability to expand or contract their ranges during favourable and unfavourable periods (Bennett 1989). *Gambusia affinis* and *M. salmoides* are not often associated with estuaries, but reports of invasion do occur. The invasibility of estuaries by alien fish relies on the physical environment and alien species' ability to tolerate the dynamic nature of the system. If appears that factors, primarily salinity and temperature, will limit the abundance of these aliens and not the presence of a diverse fish fauna. The physical environment and the ability to tolerate this over long periods will determine the establishment success of alien species. In the Wilderness Lakes, *G. affinis* and *O. mossambicus* are able to withstand the physical environment and will persist in the system for extended periods. *Micropterus salmoides* and *C. carpio* are less tolerant of the physical environment, primarily salinity and their persistence/establishment success in the system will be limited by salinity, with repeated introductions being important in maintaining the population. With two established, one establishing and one casual alien species in the system, the effects these species could have on native species diversity is assessed in Chapter 6.

CHAPTER 6

LONG TERM CHANGES IN ICHTHYOFAUNA IN THE WILDERNESS LAKES SYSTEM

INTRODUCTION

The introduction of invasive species is widely considered to be a leading cause of species threat and extinction (Sakai *et al*. 2001; Koehn 2004; Canonico *et al*. 2005). In fact, invasive species are regarded as the second leading cause of species extinction and threat worldwide, following habitat destruction (Canonico *et al*. 2005; Casal *et al*. 2007; Pejchar & Mooney 2009). The ecological impacts of invasive species on inland water ecosystems vary depending on the invading species, the extent of the invasion, and the vulnerability of the ecosystem being invaded (Canonico *et al*. 2005). Loss and degradation of biodiversity caused by invasive species can occur throughout all levels of biological organization from the genetic and population levels to the species, community, and habitat/ecosystem levels. Impacts can vary in terms of their severity, interaction with other threats, and ability to cascade throughout an entire ecosystem (Khan *et al*. 2003; Canonico *et al*. 2005; Moyle & Marchetti 2006). In South African freshwater ecosystems alien invasive fishes are considered the primary threat to native fishes (Tweddle *et al*. 2009).

Communities of fish inhabiting estuaries represent a combination of freshwater and marine species both living at the edge of their distribution, estuarine resident and migrating species passing the estuary on their way to the spawning grounds (Potter *et al*. 1986; Maes *et al*. 1998; Koutrakis *et al*. 2000). The temporal structure in estuaries is often the result of seasonal migrations of young fish moving between coast and adjacent estuaries (Sheaves 1993; Maes *et al*. 1998). For temperate estuaries, this pattern of movement results in consecutive migration waves of juveniles of marine fish (Maes *et al*. 1998). Estuarine fish communities are highly variable, and in some cases they experience large temporal changes in species composition and abundance (Potter *et al*. 1986; Koutrakis *et al*. 2000). The timing and duration of estuary mouth opening phases are major determinants in fish diversity and abundance (Russell 1996). Mouth opening events need to be of sufficient duration to enable reproductively active adults to move to the sea and spawn and recruitment of fry back into the estuary (Russell 1996; Lukey *et al*. 2006). Mouth opening events which do not coincide with the optimal breeding period could impede the process thereby affecting the composition of the fish community (Russell 1996; Vorwerk 2000; Bell *et al*. 2001; James 2006; James *et al*. 2008). Fish species diversity and abundance declines during extended mouth closure phases due to limited recruitment from the sea.

The ichthyofauna of the Wilderness Lakes system was investigated between 1983 and 1985 (Hall 1985). The fish fauna was made up predominantly of native estuarine species and euryhaline marine species with two alien freshwater species (*O. mossambicus* and *M. salmoides*). Post-1996 saw an additional two alien species, *G. affinis* and *C. carpio* being recorded in the system in 1999 and 2009 (Russell 1999b; Olds *et al*. 2011). In most temporary open/closed estuaries, changes in estuarine fish communities and primarily the marine fish community structure, are driven by mouth state and the opportunity for fish to recruit (James *et al*. 2008). Estuaries are complex systems in which the impacts of alien fishes are difficult to assess. Long term monitoring of systems is therefore crucial in attempting to determine the impact of alien fishes.

The aim of this chapter was to determine if there were changes in the species composition in the Wilderness Lakes system between the 1985 study and the current study and whether those changes could be attributed to alien fish. As a result of this, two hypotheses were addressed in this chapter:

1) Species diversity has decreased between 1985 and 2010
2) Alien fishes have increased in relative abundance

METHODS AND MATERIALS

Field sampling and analysis

Fish were sampled using fyke nets, gill nets, 30 m and 10 m seine nets as described in Chapter 2. Where possible, the sampling procedure and gears used were designed to be comparable with that of Hall (1985). For these analyses, the data collected by Russell (1996, 1999b) and James & Harrison (2008) could not be compared as the sampling techniques were not standardised and would therefore not be comparable.

In the 1985 study by Hall, one set of five gill nets each measuring 25 m × 2.75 m and with stretch meshes of 35, 45, 73 and 93 mm were set for three consecutive nights in different positions in each of the lakes. In the Wilderness Lagoon, Touw Estuary and lower and upper Serpentine channel they were only set once per season. In the present study, a total of six nets (25 m × 2.75 m) with 5 m panels of mesh sizes 35, 45, 73, 93, 118 and 150 mm were set in each of the lakes, three nets were set in the Touw and one in the Serpentine channel. All fish caught in the 118 mm and 150 mm mesh sizes were excluded from the analysis. For comparison, gill net data from 1985 needed to be adjusted to the effort of the 2010 study to enable direct comparison. These adjustments are shown in **Table 6.1**.

Table 6.1. Gill net sampling adjustments to make the study by Hall (1985) and the current study directly comparable. TE = Touw Estuary, SC = Serpentine channel IL = Island Lake, LV = Langvlei and RV = Rondevlei.

Sampling area	Hall (1985)	current study	Conversion
TE	25 m × 10 nets × 4 seasons = 1000 m	25 m × 3 nets × 4 seasons = 300 m	3.3
SC	25 m × 10 nets × 4 seasons = 1000 m	25 m × 1 net × 4 seasons = 100 m	10
IL, LV, RV	25 m × 15 nets × 4 seasons = 1500 m	25 m × 6 nets × 4 seasons = 600 m	2.5

In the current study, the total gill net length in each sampling area was 600 m in each of the lakes, 300 m in the Touw Estuary and 100m in the Serpentine channel. In order to compare the gill net results from Hall (1985) and the current study, the number of fish sampled in 1985 was divided by 2.5 for Island Lake, Langvlei and Rondevlei, divided by 3.3 in the Touw Estuary and by ten for the Serpentine channel and rounded to the nearest whole number. For the gill net CPUE comparison, a CPUE was calculated per 2010 net length (25 m of gill net per night) of fish sampled.

Due to the difficulties in completing four seasons of 30 m and 10 m sampling in 2010, all seine net comparisons were calculated on a per haul basis. Fyke net data were excluded from diversity indices as fyke netting was not included in the study by Hall (1985)

Statistical analysis

To define the fish community mathematically, diversity indices were used. These express the ratio between species and individuals in a biotic community (Magnussen & Boyle 1995). The

Shannon-Wiener diversity index (Shannon & Weaver 1949) was calculated and compared between 1985 and 2010 and between water bodies using PRIMER v6 software, where:

Shannon-Wiener Index:

$$H' = -\sum_{i=1}^{S} Pi \log Pi$$

where H' is the Shannon-Wiener Index, P*i* is the relative abundance of S (each species) and the summation is over the number of species in the sample. The Shannon-Wiener index considers that the individuals were randomly sampled and that all the species were represented in the same sample.

A chi square contingency table was used to assess whether the number of species sampled in each estuarine association category (Whitfield 1998) was dependant on the different study periods.

RESULTS

Species composition

A total of 32 species was sampled from the Wilderness Lakes system during the 1985 and 2010 studies. Of these 21 species were present in both surveys. Two Gobiidae species (*Redigobius dewaali* and *G. callidus*) sampled in 2010 were not sampled in 1985, both of which were sampled in the Touw Estuary and Island Lake and a Clinidae species was sampled in 1985 and not in 2010 (**Table 6.2**). The largest variation in species present was euryhaline marine species. Four species (*Sarpa salpa*, *Rhabdosargus sarba*, *Trachurus capensis* and *Heteromycteris capensis*) sampled in 1985 were not sampled in 2010 and one species (*E. machnata*) was sampled in 2010 not sampled in 1985 (**Table 6.2**). Two alien fish species (*G. affinis* and *C. carpio*) were sampled in the Wilderness Lakes system that were not sampled in

1985. The inclusion of fyke netting into the 2010 sampling regime resulted in the sampling of eel (*A. mossambica)* which were not sampled in 1985 (**Table 6.2**).

Table 6.2. Summary table showing the overall distribution of fish species sampled from the Wilderness Lakes system, Western Cape, South Africa in 1985 (Hall 1985) and 2011. EA = Estuarine Association category (after Whitfield 1998). Grey shading indicates the species was sampled in that study.

Species	EA	1985	2010	Species	EA	1985	2010
Atherina breviceps	Ib	■	■	*Lichia amia*	IIa	■	■
Gilchristella aestuaria	Ib	■	■	*Elops machnata*	IIa		■
Caffrogobius gilchristi	Ib	■	■	*Argyrosomus japonicus*	IIa	■	■
Glossogobius callidus	Ib		■	*Solea bleekeri*	IIa	■	■
Psammogobius knysnaensis	Ib	■	■	*Heteromycteris capensis*	IIb	■	
Redigobius dewaali	Ib		■	*Lithognathus lithognathus*	IIa	■	■
Clinus superciliosus	Ib	■		*Rhabdosargus holubi*	IIa	■	■
Hyporhamphus capensis	Ia	■	■	*Rhabdosargus sarba*	IIb	■	
Syngnathus acus	Ib	■	■	*Sarpa salpa*	IIc	■	
Galeichthys feliceps	IIb	■	■	*Trachurus capensis*	III	■	
Pomadasys commersonnii	IIa	■	■	*Oreochromis mossambicus*	IV	■	■
Monodactylus falciformis	IIa	■	■	*Cyprinus carpio*	IV		■
Mugil cephalus	IIa	■	■	*Gambusia affinis*	IV		■
Liza dumerili	IIb	■	■	*Micropterus salmoides*	IV	■	■
Liza richardsonii	IIc	■	■	*Anguilla mossambica*	Va		■
Liza tricuspidens	IIb	■	■	*Myxus capensis*	Vb	■	■
juvenile Mugilidae	II	■	■				

Species diversity

In both studies, the Touw Estuary had the highest number of species sampled (1985 = 20 species, 2010 = 21 species) and Langvlei had the lowest number of species sampled, with 11 species sampled in 1985 and 13 in 2010. More species were sampled in Island Lake in 2010 (n = 21) and Rondevlei (n = 15) than in 1985 (**Fig. 6.1**). In terms of diversity, the overall trend of geographic variation has remained relatively constant between studies, with the greatest number of species sampled from the Touw Estuary and the least from Langvlei (**Fig. 6.1**). The increase in the number of species sampled is the result of two additional alien species in the system.

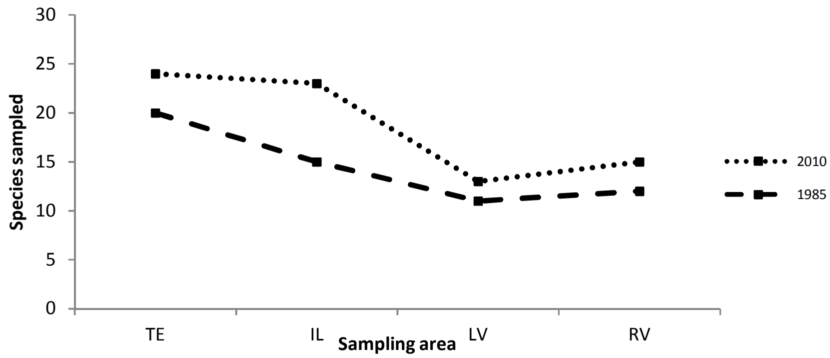


Figure 6.1. Number of fish species sampled in the Wilderness Lakes system in 1985 (Hall 1985) and 2010 in 10m and 30m seine nets and gill nets, Locality: TE = Touw Estuary; IL = Island Lake; LV = Langvlei and RV = Rondevlei.

In 1985 the total H’ was highest in Touw Estuary (H’ = 2.12) and lowest in Langvlei and Rondevlei (H’ = 0.33 and 0.47 respectively). In 2010 the total H’ was lowest in the Touw Estuary (H’ = 0.78) and highest in Rondevlei (H’ = 1.31) (**Fig. 6.2**).

When the diversity is separated into individual gear types, the 10 m seine net data shows a similar trend to the overall diversity, with the highest H’ in Rondevlei followed by Island Lake. In 1985 10 m seine Island Lake (H’ = 1.11) was the most diverse, followed by Rondevlei (H’ = 0.33) and Langvlei (H’ = 0.21) (**Fig. 4.2**). The 30 m seine net and gill net diversity show very similar trends between 1985 and 2010. In 1985, 30 m seine net H’ was highest in the Touw Estuary whereas in 2010 Rondevlei had the highest H’. The gill net diversity has remained the most constant between 1985 and 2010 with the diversity indices being very similar for Rondevlei, Langvlei and Island Lake and with the biggest differences between the Serpentine channel (H’ 1983 = 2.01, H’ 2010 = 1.54) and the Touw Estuary (H’ 1983 = 1.97, H’ 2010 = 1.42) (**Fig 6.2**).

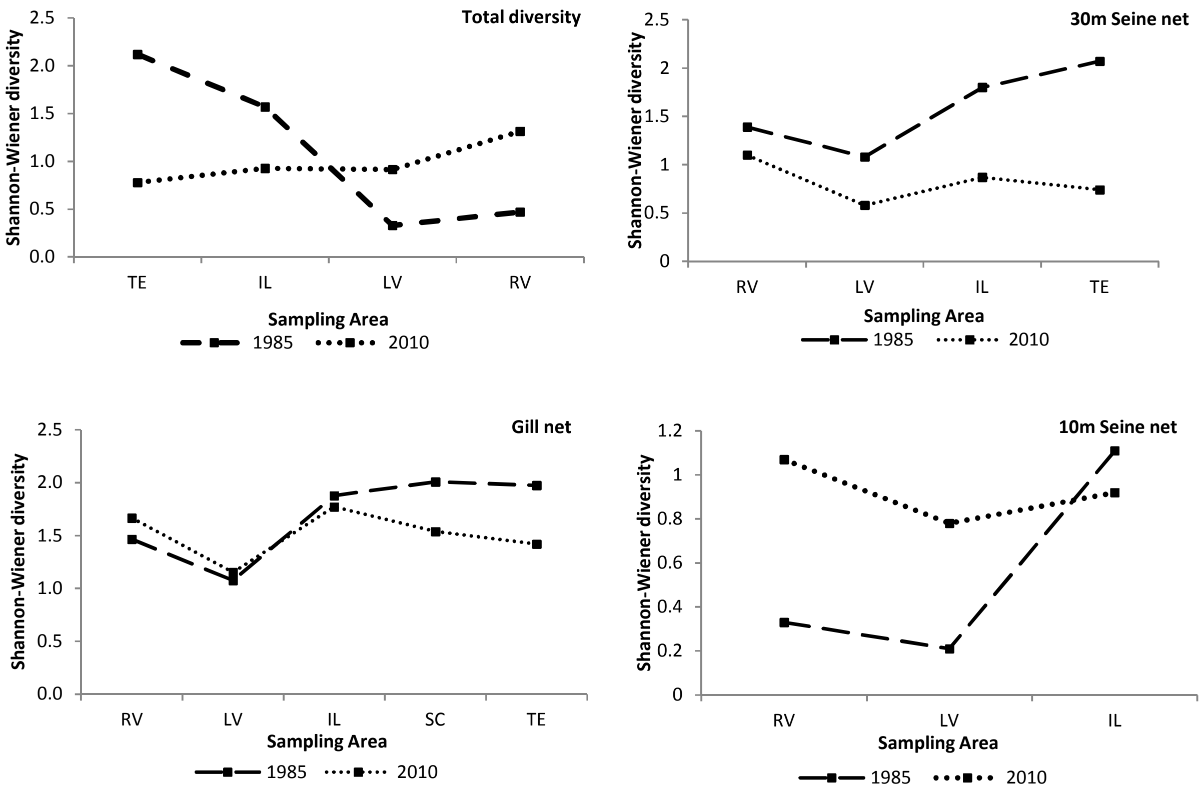


Figure 6.2. Shannon-Wiener diversity index for the Wilderness Lakes system, Western Cape. The total diversity calculated from 10 m and 30 m seine nets and gill nets in 1985 (Hall 1985) and the current study, Locality: TE = Touw Estuary; IL = Island Lake; LV = Langvlei and RV = Rondevlei.

Changes in fish abundance

The fish species sampled in the 10 m seine nets differed between studies (**Table 6.3**). In Rondevlei, the species contributing to the large differences in CPUE were *A. breviceps* (1985 CPUE = 1051.8 $fish.haul^{-1}$, 2010 CPUE = 32.2 $fish.haul^{-1}$) and *G. aestuaria* (1985 CPUE = 45.9 $fish.haul^{-1}$, 2010 CPUE = 1.0 $fish.haul^{-1}$). Juvenile Mugilidae species were sampled in the 10m seine net in 1985 but not sampled in 2010. Differences were seen between numbers of *L. richardsonii*, *Heteromycteris capensis* and *H. capensis* (**Table 6.3**).

Table 6.3. CPUE measured in number of fish. net haul^{-1} from the 10 m seine net sampling in 1985 and 2010 in Island Lake, Langvlei and Rondevlei.

	1985			2010		
	Island Lake	Langvlei	Rondevlei	Island Lake	Langvlei	Rondevlei
Psammogobius knysnaensis	0.2	0.0	0.0	5.5 ± 12.1	0.9 ± 1.8	2.6 ± 8.6
Caffrogobius gilchristi	0.0	0.0	0.0	0.0	0.0	0.0
Hyporhamphus capensis	0.7	0.4	10.0	1.4 ± 5.2	0.3 ± 0.7	2.8 ± 10.9
Monodactylus falciformis	0.1	0.0	0.0	0.0	0.0	0.0
Mugil cephalus	1.2	0.1	0.3	0.0	0.0	0.0
Liza dumerili	2.8	0.0	0.4	0.0	0.0	0.0
Liza richardsonii	3.0	0.0	4.1	0.1 ± 0.5	0.0	0.0
Lithognathus lithognathus	0.6	0.0	0.0	0.0	0.0	0.0
Rhabdosargus holubi	4.2	0.0	0.0	0.0	0.0	0.0
Rhabdosargus sarba	0.0	0.0	0.0	0.0	0.0	0.0
Atherina breviceps	186.5	772.3	1051.8	114.7 ± 188.2	397.5 ± 631	32.3 ± 91.1
Gilchristella aestuaria	34.8	24.1	45.9	107.4 ± 142.8	485.4 ± 974.1	0.9 ± 1.8
Micropterus salmoides	0.1	0.1	0.0	0.0	0.0	0.0
Oreochromis mossambicus	95.5	9.8	17.6	3.6 ± 20.3	16.5 ± 75.9	21.9 ± 73.9
Clinus superciliosus	0.0	0.0	0.0	0.0	0.0	0.0
Syngnathus acus	0.0	0.0	0.1	1.1 ± 2.6	0.0	0.3 ± 1.1
Gambusia affinis	0.0	0.0	0.0	0.03 ± 0.2	0.1 ± 0.2	0.0

The same number of species was sampled from Langvlei in both studies and the major differences in CPUE were *A. breviceps* (1985 CPUE = 772.3 fish.haul^{-1}, 2010 CPUE = 397.5 ± 631 fish.haul^{-1}) and *G. aestuaria* (1985 CPUE = 24.1 fish.haul^{-1}, 2010 CPUE = 485.4 ± 974 fish.haul^{-1}) (**Table 6.3**).

Similarly, in Island Lake the largest changes in CPUE were *A. breviceps* and *G. aestuaria*. Higher number of *O. mossambicus* were sampled in 1985 than in 2010 (1985 CPUE = 95.5 fish.haul^{-1}, 2010 CPUE = 3.6 ± 20.3 fish.haul^{-1}). Larger numbers of species were caught in Island Lake in 1985 which comprised primarily of juveniles of four Mugilidae species, *L. lithognathus*, *R. holubi*, *Rhabdosargus sarba, M. falciformis, C. gilchristi* and *M. salmoides*.

CPUE in the 30 m seine net differed mainly between the smaller shoaling pelagic species (**Table 6.4**). These differences could be attributed to the difference in net mesh sizes. The 1985 mesh size was 28 mm whereas in 2010 the mesh size was 12 mm. Fish small enough to fit through a 28 mm mesh such as smaller *A. breviceps*, *G. aestuaria*, juvenile *O. mossambicus, S. acus* and *P. capensis* would have been able to escape from the net and these species were all common and abundant in the 2010 sampling. Due to this, direct comparisons on CPUE of these species were not possible.

For the larger species, higher numbers of *O. mossambicus*, *P. knysnaensis*, *H. capensis* were sampled in Island Lake (**Table 6.4**). *Sarpa salpa*, *M. cephalus*, *M. capensis*, *L. dumerili*, *S. bleekeri* and *H. capensis* were sampled in the Touw Estuary in 1985 but not sampled in 2010. Species sampled in 2010 but not in 1983 include *G. affinis*, *S. acus*, *R. dewaali* and *C. carpio*. In 1985 the CPUE of *R. holubi* and *L. richardsoni*i was higher than in 2010.

The comparisons of gill net CPUE between the two studies are shown in **Table 6.5**. More species were sampled in Island Lake, Langvlei and Rondevlei in 2010 than in 1985. Fewer species were sampled in 2010 in the Serpentine channel and the Touw Estuary than in 1985. Species sampled in the gill nets in 1985 but not in 2019 were *Trachurus capensis* and *C. gilchristi*. The species sampled in 2010 but not 1985 was *E. machnata*.

In the Touw Estuary the CPUE of *G. feliceps* (1985 CPUE = 0.5 fish. 25m net.night^{-1}, 2010 CPUE = 4.4 ± 9.9 fish. 25m net.night^{-1}) was much higher than in 1985 where the CPUE of the other species sampled remained fairly low and constant. In the Serpentine channel the CPUE for *M. falciformis*, *L. richardsonii* (1985 CPUE = 0.7 fish. 25m net.night^{-1}, 2010 CPUE = 3.8 ± 3.9 fish. 25m net.night^{-1}), *M. capensis* and *P. commersonnii* were markedly higher in 2010 than in 1985. In Island Lake, *L. richardsonii* (1985 CPUE = 3.8 fish. 25m net.night^{-1}, 2010 CPUE = 6.9 ± 4.7 fish. 25m net.night^{-1}) and *P. commersonnii* (1985 CPUE = 0 fish. 25m net.night^{-1}, 2010 CPUE = 2.8 ± 4.3 fish. 25m net.night^{-1}) had a much higher CPUE in 2010 than in 1985 whereas the CPUE decreased for *O. mossambicus*, *L. dumerili* and *M. falciformis* between 1985 and 2010.

In Langvlei, the CPUE increased from 1985 to 2010 for *O. mossambicus*, and *M. falciformis* (1985 CPUE = 0 fish. 25m net.night^{-1}, 2010 CPUE = 11.6 ± 10.5 fish. 25m net.night^{-1}) but CPUE decreased for *L. richardsonii*. In Rondevlei the CPUE of *M. falciformis*, *L. dumerili* and *M. capensis* (1985 CPUE = 0.2 fish. 25m net.night^{-1}, 2010 CPUE = 4.6 ± 5.7 fish. 25m net.night^{-1}) were higher in 2010 than 1985.

Table 6.4. CPUE measured in number of fish. net haul^{-1} from the 30 m seine net sampling in 1985 and 2010 in the Touw Estuary, Island Lake, Langvlei and Rondevlei.

	1985				2010			
	Touw Estuary	Island Lake	Langvlei	Rondevlei	Touw Estuary	Island Lake	Langvlei	Rondevlei
Atherina breviceps	0.1	16.3	0.0	36.5	858.3 ± 2156.3	1309.2 ± 4886.2	159.6 ± 464.7	441.9 ± 1217.3
Gilchristella aestuaria	8.1	17.8	0.0	27.1	551.2 ± 1397.4	1006.7 ± 1195.8	763 ± 1739.3	655.7 ± 1346.8
Hyporhamphus capensis	0.0	1.0	0.0	0.0	0.1 ± 0.3	17.1 ± 35.4	116.9 ± 217.3	566.5 ± 1183.1
Psammogobius knysnaensis	1.0	0.0	0.0	0.0	5.1 ± 12.9	4.5 ± 9.2	0.1 ± 0.2	0.5 ± 1.0
Caffrogobius gilchristi	0.8	0.0	0.0	0.0	0.03 ± 0.2	0.0	0.0	0.0
Redigobius dewaali	0.0	0.0	0.0	0.0	0.1 ± 0.3	0.04 ± 0.2	0.0	0.0
Liza richardsonii	2.9	44.0	0.1	1.2	0.5 ± 2.1	0.7 ±2.5	0.6 ± 1.7	1.2 ± 2.5
Liza dumerili	3.5	10.0	0.0	1.1	0.0	0.0	0.0	0.0
Liza tricuspidens	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Mugil cephalus	2.4	10.3	0.3	0.4	0.0	0.04 ± 0.2	0.0	0.0
Juvenile Mugilidae	0.0	0.0	0.0	0.0	2.1 ± 12.5	0.0	0.0	0.0
Rhabdosargus holubi	9.4	11.7	0.0	0.0	2.2 ± 6.8	2.3 ± 6.0	0.0	0.0
Monodactylus falciformis	0.1	0.9	0.0	0.0	0.4 ± 0.9	0.0	0.0	0.0
Syngnathus acus	0.0	0.0	0.0	0.0	1.0 ± 2.0	0.6 ± 1.4	0.0	0.4 ± 1.1
Solea bleekeri	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Heteromycteris capensis	0.0	0.0	0.0	7.1	0.0	0.0	0.0	0.0
Diplodus capensis capensis	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sarpa salpa	3.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Lithognathus lithognathus	3.1	2.8	0.0	0.0	1.4 ± 3.3	1.3 ± 1.8	0.0	0.0
Lichia amia	0.2	0.0	0.0	0.0	0.1 ± 0.5	0.1 ± 0.5	0.0	0.0
Oreochromis mossambicus	0.1	0.8	0.1	21.2	3.0 ± 9.1	81.6 ± 246.0	1.1 ± 1.9	1.6 ± 3.3
Cyprinus carpio	0.0	0.0	0.0	0.0	0.1 ± 0.3	0.1 ± 0.6	0.3 ± 0.7	0.2 ± 0.6
Gambusia affinis	0.0	0.0	0.0	0.0	3 ± 12.9	0.0	0.0	0.0
Micropterus salmoides	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0
Myxus capensis	0.1	0.1	0.0	0.1	0.03 ±0.2	0.0	0.2 ± 0.6	0.0

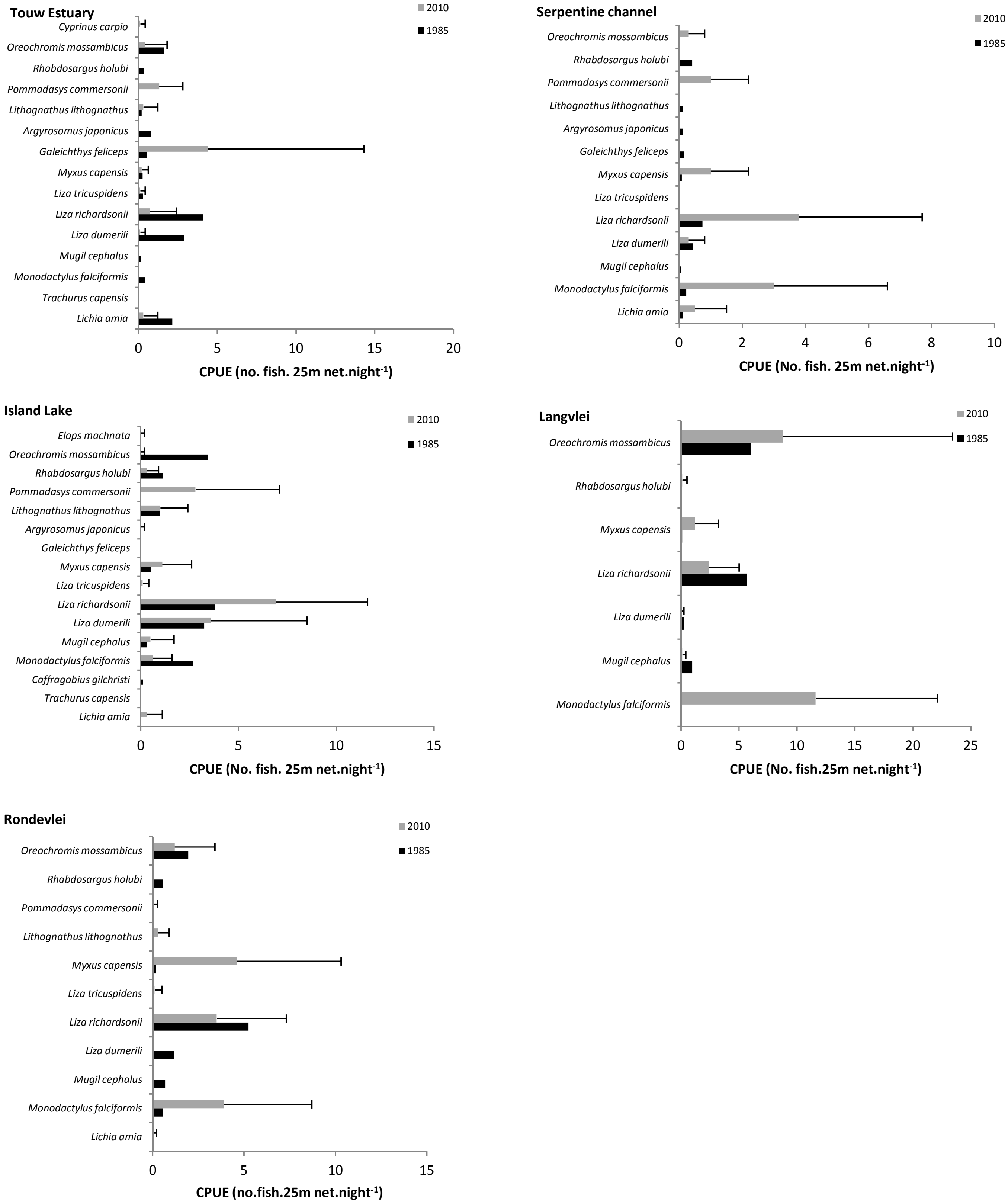


Figure 6.3. The adjusted gill net CPUE (number of fish. 25 m net. night^{-1}) data recorded during 1985 and 2010 for Island Lake, Langvlei, Rondevlei, the Serpentine channel and the Touw Estuary. The error bars shown on the 2010 data represent the standard deviation.

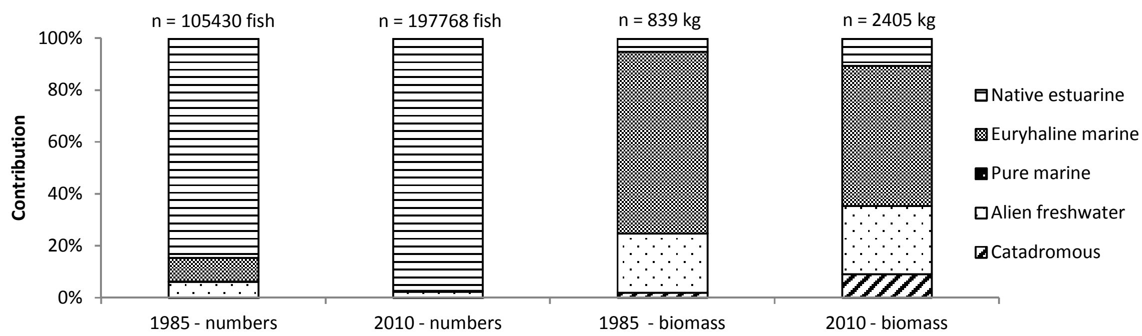


Figure 6.4. The changes in fish abundance and biomass grouped according to Whitfield's estuarine association category in 1994a and 1998.

In both studies the native estuarine fishes dominated the number of fish sampled but the euryhaline marine fishes dominated the biomass sampled (**Fig. 6.4**). The proportion of fish numbers sampled in each category did not differ significantly (five estuarine categories × two years contingency table, χ^2= 1.93; d.f. = 4; P = 0.75) between studies. The biomass proportion of fish sampled in each estuarine category were very similar between 1985 and 2010. The proportion by number of native estuarine fishes dominated numerically whereas gravimetrically the euryhaline marine species were dominant. The biomass of the freshwater fishes increased from 1985 to 2010 (**Fig. 6.4**).

DISCUSSION

According to Preston (1948) three time scales are relevant to the study of species abundance and distributions. These are sampling duration, which is the period over which data are collected, ecological time, during which the composition and possibly the shape of the species abundance distribution will change as a result of succession, immigration and other dynamic processes, and evolutionary time when new forms arise through speciation or are removed by extinction. Much of the work on species abundance distributions relates to the first two time scales (Margurran 2007). The changes in species composition over time are inevitable but changes in composition as a result of alien invasion is of large concern to ecologists and conservationists. Estuarine fish communities are highly variable, and in some cases they experience large temporal changes in species composition and abundance (Koutrakis *et al.* 2000). The introduction of non-native species into estuarine systems may be resulting in a homogenization of fauna with invasive species claimed to cause extinctions of native species by competition and predation (Gurevitch & Padilla 2004; Scott & Helfman 2001). Over the past 28 years, the number of known invasive species in Wilderness Lakes system has increased from two to four and as such the possible changes in species composition and abundances is important in determining the potential impacts of invasive fish species.

Relative abundance of invasive fish species

Habitat destruction and biotic invasions are two of the leading causes in altering natural systems (Baxter *et al*. 2004). In Australia, invasive, non-native fish species are one of the leading causes in the decline of 22 species of native fish that are classified as endangered, vulnerable, or rare (Conanico *et al*. 2005). The number of alien fish species present in the Wilderness Lakes has increased since 1985. No *C. carpio* or *G. affinis* were sampled in 1985 and nine *M. salmoides* were sampled.

Gambusia affinis was first sampled in the Wilderness Lakes system in 1999 (Russell 1999b) in the Duiwe and Touw Rivers. While their abundance was low in the current catch composition, specific sampling targeting this species showed very high abundances of *G. affinis* in the littoral zones of the lakes and the interconnecting channels (Chapter 4). The most recent introduction into the Wilderness Lakes was *C. carpio*, which were first noted by SANParks staff (I. Russell, aquatic scientist, SANParks, pers. comm.) in 2003. Numbers sampled in the seine nets and gill nets have been low (n = 13) and anglers reported catching adult *C. carpio* since 2008 (Chapter 5).

Species composition and diversity

Estuarine fish assemblages often exhibit large year-to-year variations in abundance, species and size composition (James *et al*. 2008). Changes in one or more environmental variables may influence the pattern of immigration from estuaries (de Ben *et al*. 1990). Estuaries are highly dynamic and their physical and chemical characteristics can change over a scale of hours to years (James *et al*. 2008). Changes were observed in the species diversity between the two studies, which is not unexpected considering the time between the studies. The variation in diversity indices can reflect seasonal and annual changes in abundance of individuals and species (de Ben *et al*. 1990). Invasion has been thought to decrease native diversity in freshwater and estuarine environment causing a homogenous populations (Moyle & Marchetti 2006; Casal *et al*. 2007; Pejchar & Mooney 2009). However, the theory that correlations between native diversity and the success of non-native species are mostly positive is reasonable, because the factors known to promote or limit native diversity are thought to similarly influence invasions (Levine 2000). The species composition in the Wilderness Lakes system was not significantly different between 1985 and 2010. However, a decrease in euryhaline marine species was noted but this is more likely to be a consequence of mouth opening than an impact of the alien fishes.

Factors affecting species abundance in the Wilderness Lakes System

The native estuarine species composition was very similar between studies with only three species not being sampled in both studies. The numerical dominance of the shoaling pelagic species (*A. breviceps* and *G. aestuaria*) was very evident in both studies. The abundance of the shoaling pelagic species is characteristic of temporary open/closed estuaries (Vorwerk *et al*. 2003; James 2006). The primary difference between the two studies was the lower number of species and overall numbers of the euryhaline marine species. One characteristic of estuarine fish assemblages is the temporary nature of abundances (de Ben *et al*. 1990). Immigration and emigration of both young fish and adults can affect population density. De Ben *et al*. (1990) observed a decrease in abundance of fish in Yaquina Bay, Oregon, during winter. This was attributed to the migration of many individuals from six of the more abundant fishes. The decrease in diversity and abundance of euryhaline marine fishes may be due to the differences in recruitment opportunity prior to both studies. The lower numbers but higher biomass of euryhaline marine fish may be due to the occurrence of fewer but larger fish which were not able to migrate to the ocean to spawn because of mouth closure (**Chapter 3, Figure 3.21**).

Studies have shown that mouth opening makes a large contribution to the species composition in temporary open/closed estuaries (Bok 1979; Bennett 1989; Russell 1996; Whitfield 1999; James 2006). In large permanently open estuaries in the northern hemisphere, the strength of recruitment pulses for different marine species vary markedly between years. This has been related to fluctuating environmental conditions in the estuarine and marine environments that influence reproductive success, larval survival and efficiency of larval transport mechanisms (James *et al*. 2008). Recruitment into estuaries occurs predominantly during spring and summer, with the optimum months being between November and January (Russell 1996). The mouth state prior to and during the study in 1985 was between 91.3% and 100% open whereas prior to and during this study the mouth state was between 7.1% and 100% open (Chapter 2). In years following mouth opening events in spring (September to November) a higher number of species are recorded than in years following no mouth openings in spring (James *et al*. 2008). During the 1985 study, the mouth was open throughout the prime recruitment periods prior to the study and in the current study the mouth was also open for the optimum recruitment period between September and November (SANParks, unpublished data). However, a drought occurred during 2010 directly after the optimum recruitment period, when the mouth closed for approximately ten months, which could have effected recruitment into the system. The changes in species composition and abundance may therefore be closely linked to the mouth state of the Touw Estuary.

Sampling programmes attempt to accurately characterize the underlying abundance distribution of communities (Margurran 2007). Two studies determining year-long species abundance and distribution 28 years apart give two snapshots of the system over time. Sampling continuously over a longer period helps to accurately determine species abundance and distribution as it includes taxa that are temporarily absent from the assemblage, or ones that were not detected during the initial survey (Margurran 2007). Species turnover is continuous, even in communities that appear to have reached species equilibrium and this needs to be taken into consideration when sampling systems over relatively short periods of time. With estuaries being so dynamic in behaviour, it is difficult to accurately determine the causes of species decline or change.

Conclusion

According to James *et al*. (2008), short-term studies have shown that most estuarine fish assemblages undergo significant changes in community structure, often related to changes in estuarine mouth phase, flood events and seasons. The decrease in species diversity and the increase of alien species abundance occur simultaneously and often correlation is thought to imply causation (Gurevitch & Padilla 2004). The decrease in native abundance and the increase in alien abundance may co-occur but may be caused by external abiotic factors which affect both communities. Studies can rarely predict conclusively whether it is the increase in invasion that is causing the decrease or extinction of native species. In the Wilderness Lakes system, the number of invasive species has increased but their abundances have remained fairly constant (*O. mossambicus*) or low (*M. salmoides*). *Cyprinus carpio* abundance may be too low to impact the native species yet. The resident native species diversity has remained the same, with an increase in overall abundance and biomass. There was no evidence of impact on the native fish abundance that could be linked to the invasive fishes. The migratory component of the fish fauna appears to be more closely linked to mouth opening events but is still a very strong and abundant component of the Wilderness Lakes system.

CHAPTER 7

GENERAL DISCUSSION

The mission of SANParks is to develop, manage and promote a system of national parks that represents biodiversity and heritage assets by applying best practice, environmental justice, benefit-sharing and sustainable use (SANParks vision and mission). An important aspect of this is managing invasion by alien species. SANParks has designed an alien management programme whose main objective is to anticipate, prevent entry and where feasible and/or necessary control invasive alien species in an effort to minimise the impact on, and maintain the integrity of indigenous biodiversity in national parks. The Wilderness Lakes system is of high conservation priority due to its status as a RAMSAR site and its location in a national park. All aliens present fall under the alien management programme. For this reason investigation into the status and management of alien species in the system is important.

Status of alien fishes in the Wilderness Lakes system

The introduction of alien fishes into the Wilderness Lakes system was not impeded by the high native fish species diversity. In the Wilderness Lakes system the species composition and diversity were higher than surrounding temporary open/closed estuaries which, together with evidence that abundance has not changed drastically (Chapter 6), indicates that there is no evidence of impact on the fish community. While the highest abundance of alien fishes was in the areas of lowest native diversity, the abundance of native fish would indicate that they are not negatively impacted by the alien fishes. The theory of biotic resistance was not exhibited in the system and it would appear that this system tends towards biotic acceptance, where 'the rich get richer' (Richardson *et al*. 2011) . The environmental conditions which allow for high native diversity also allow for high alien abundance.

The establishment of the four alien species in the Wilderness Lakes are in different stages. *Oreochromis mossambicus* is fully established and after the native shoaling pelagic species (*A. breviceps* and *G. aestuaria*), is one of the most abundant and widely distributed species in the system. It has been there for at least twenty five years and is an invasive species. Its ability to tolerate high salinity and temperature and low dissolved oxygen and pH aided this species in establishing itself in systems around the world (Costa-Pierce 2003; Canonico *et al.* 2005; Casal *et al*. 2007). Natural dispersal of *O. mossambicus* to estuaries along the south east coast via the sea has been reported (Harrison 1966; Whitfield & Blaber 1979) however, the most likely source of introduction is through stocking of fish into dams in the catchment.

Gambusia affinis is a relatively new introduction into the Wilderness Lakes system having first been recorded in 1999 (Russell 1999b). This species is fully established and invasive in the system. The date and purpose of introduction is unknown, but its introduction is generally for the purpose of mosquito control and as a fodder fish for non-native predators (de Moor & Bruton 1988). This species was not intentionally stocked into the Wilderness Lakes system but was introduced into the nearby Groenvlei as fodder fish for *M. salmoides*. It can be assumed that *G. affinis* was either introduced illegally or was accidentally introduced into the system. The wide environmental tolerances (Chapter 5) of this species mean the physical environment did not limit its establishment.

Micropterus salmoides have been recorded in the Wilderness Lakes system for over twenty five years, yet their abundances have remained low and their distribution appears to be limited to areas with lower salinity (Chapter 5). Introduction from farm dams in the Wilderness Lakes catchment is the most likely source of introduction. Successful reproduction in the system appears to be limited by the high salinities. Therefore, repeated introductions from the Duiwe River during flooding are the most likely cause of this species persistence in the system and limits its establishment. This type of persistence via repeated introductions makes it a casual species in the system.

Cyprinus carpio were introduced into the Wilderness Lakes system before 2003 (I. Russell, aquatic scientist, SANParks, pers. comm.) with the first record of it in the system in 2009. The purpose and source of introduction is unknown but it is most likely from dams in the catchment that is stocked with *C. carpio*, and fish were washed into the system during floods (Chapter 4). This species is in the process of establishing, being distributed throughout the system. The abundance still appears to be low. Reproduction has been successful but this may only be possible during years of low salinity as higher salinity hinders egg development. If repeated reproduction is successful it is likely that the species will become fully established in the system.

Framework for biological invasions

The barriers experienced by alien fish during the establishment phase were described in Chapter 4. From these barriers, the framework for biological invasions was based (Richardson *et al*. 2000; Blackburn *et al*. 2011). According to the framework, *O. mossambicus* and *G. affinis* are invasive and established species; *C. carpio* is alien and establishing and *M. salmoides* is alien and a casual species. It is possible that *M. salmoides* could still face invasion failure if fish no longer move into the system from the Duiwe River.

In the framework (**Fig. 7.1**) there are categories A to E which relate to the alien species invasion pathway. *Oreochromis mossambicus* and *G. affinis* fall into category E which is a fully invasive species, with individuals dispersing, surviving and reproducing at multiple sites across a greater or lesser spectrum of habitats and extent of occurrence (Blackburn *et al*. 2011). *Micropterus salmoides* is in category C2, where individuals are surviving in the system with reproduction occurring, but the population is not self-sustaining (Blackburn *et al*. 2011). *Cyprinus carpio* are currently between C2 and C3, where individuals are surviving in the system with reproduction occurring and the population may or may not be self sustaining. If the juveniles from the 2010/2011 spawning season survive to spawn, the species will be in category C3, possibly moving towards categories D and E (Blackburn *et al*. 2011).

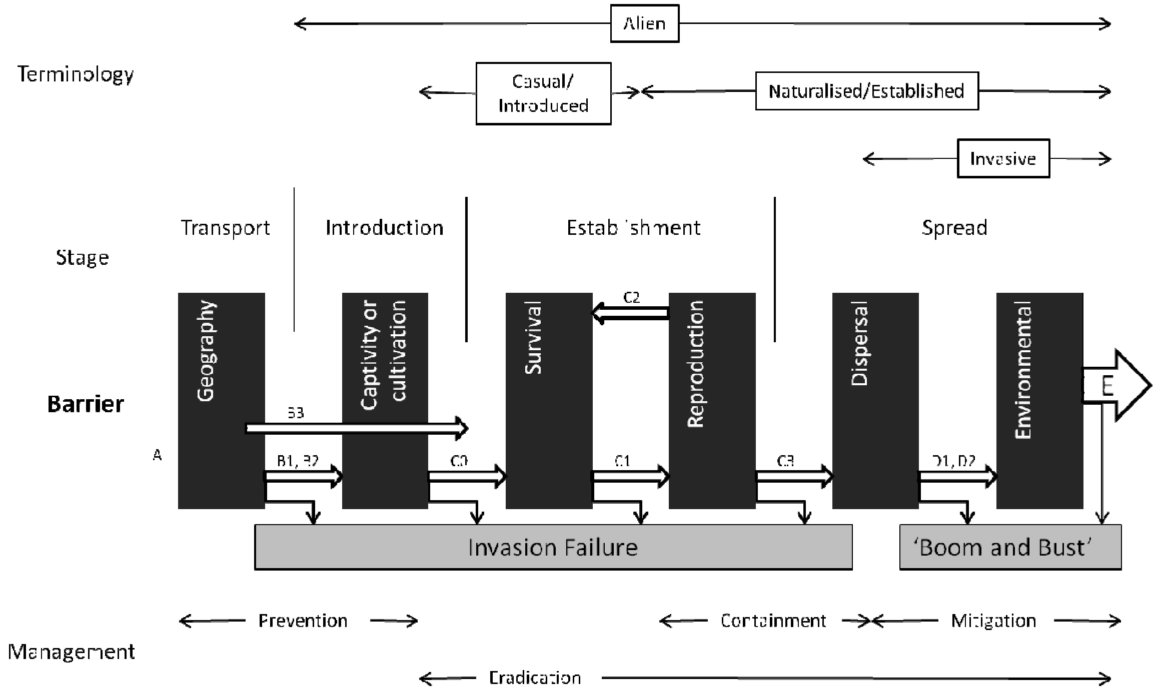


Figure 7.1. A proposed uniformed framework for biological invasions. The proposed framework divides the process of invasion into a series of stages. In each stage there are barriers that need to be overcome for a species or population to pass on to the next stage. These species are referred to by different terms depending on where in the invasion process they have reached. Different management interventions apply at different stages. The unfilled block arrows describe the movement of species along the invasion framework with respect to the barriers, and the alphanumeric codes associated with the arrows relate to the categorisation of species with respect to the invasion (from Blackburn *et al*. 2011).

Potential impacts of alien fishes

Oreochromis mossambicus occurs naturally in estuarine systems that have similar species compositions to the Wilderness Lakes system (Vorwerk *et al*. 2001; James *et al*. 2007; James & Harrison 2008), which suggests that negative impacts on the fish community are unlikely. Their high abundance in the Wilderness Lakes system could result in competition with native species for space and food. The trophic structure of fish biota is determined by the feeding biology of fishes (**Fig. 7.2**). *Oreochromis mossambicus* is a detritivore and is situated relatively low on the food chain. They are in the same feeding guild as the native Mugilidae family (**Fig. 7.2**); however their primary food source and the feeding mechanisms differ (Whitfield & Blaber 1979). This results in the available microbenthos being consumed in differing quantities, reducing the risk of competition. However, it was shown in Chapter 6 that there appears to be little impact from alien fishes on the native diversity. The Mugilids which are in the same feeding guild as *O. mossambicus* (Whitfield & Blaber 1979) are one of the most important and abundant (Chapter 3) families in the Wilderness Lakes system. This would indicate that they are not being out competed by *O. mossambicus* for food as their food source is unlikely to be limiting, a finding also supported by Whitfield & Blaber (1979).

Gambusia affinis reach a maximum size of 70 mm TL, which makes them one of the smallest fish species in the system. For this reason it is unlikely that they cause any direct impact on the larger species in the native fish community. However, they are aggressive opportunistic omnivores (Pyke 2005) that prey on mosquito larvae, algae, crustaceans, insects and their larvae, amphibian larvae and small fishes (Leyse 2004). The native estuarine shoaling species, *G. aestuaria* and *A. breviceps*, feed primarily on copepod and decapod larvae (Talbot & Baird 1985). Competition for food with the shoaling species is a possibility, however these species were numerically dominant throughout the system (Chapter 3) (**Fig 7.2**) so they do not appear to be negatively affected by *G. affinis*. The introduction of *G. affinis* can drastically reduce zooplankton populations and shift size frequencies of remaining populations through selective predation of larger individuals (Leyse 2004). Mosquito larvae form only a small component of the diet of *G. affinis* and they prey on other invertebrate predators. Predation on zooplankton, a common food item for *G. affinis* in many aquatic habitats, may indirectly affect mosquito populations (Blaustein & Karban 1990). By serving as an alternative prey, zooplankton may initially decrease the predation pressure on mosquito larvae. But feeding on zooplankton will increase the population growth of *G. affinis*, which in time would increase the predation pressure on mosquito larvae (Blaustein & Karban 1990).

Gambusia affinis are reported to prey on a wide range of amphibian eggs, including species from the Order Anura (Grubb 1972). Thirteen species from six families from the Anuran family are present in the Wilderness Lakes system (Russell *et al*. 2010) and their juveniles will be susceptible to predation by *G. affinis*. Predation on the endemic California fairy shrimp (*Linderiella occidentalis*) significantly reduced its abundance in pools even when alternative food sources were available (Leyse 2004). Cascade effects on communities caused by *G. affinis* were investigated by Hurlbert *et al*. (1972). This study found that predation on zooplankton, whose diet consisted of phytoplankton, resulted in a large decrease in zooplankton abundance and algal blooms in experimental ponds. The impacts of *G. affinis* can therefore extend further than just predation on invertebrates but the cascade effects that result from its predatory nature and these impacts need to be assessed.

Cyprinus carpio is a benthivorous fish which forages for food in the sediment (De Moor 1996; Zambrano *et al*. 2001). *Cyprinus carpio* are considered omnivorous fish that feed on chironomids, tubificids, larger zooplankton, zooperiphyton and detritus (García-Berthou 2001). While direct competition with the native biota feeding on the benthos is possible (**Fig. 7.2**), their indirect impact on native species may be more closely linked to their feeding behaviour rather than what they are feeding on (García-Berthou 2001; Britton *et al*. 2007). During feeding this species uproots macrophytes, thus disturbing and destroying the food sources for the native species (Parkos III *et al*. 2003). Elsewhere this species has been shown to affect plant and animal communities in shallow environments both directly and indirectly (Miller & Crowl 2006). They directly consume macrophytes and indirectly uproot or even break macrophytes by mechanically damaging plants while foraging. Their feeding behaviour increases turbidity through resuspension of sediments (Khan *et al*. 2003; Parkos III *et al*. 2003; Koehn 2004; Miller & Crowl 2006). The specialist feeding mechanism of sieving through the substrate allows them to take advantage of potentially underutilised resources, including detritus at a base level of the food chain (Koehn 2004). *Cyprinus carpio* should increase the flux of nutrients from the benthic to the pelagic zone of lakes and change the dominant primary producer from submerged macrophytes to phytoplankton (Zambrano *et al*. 2001; Parkos III *et al*. 2003).

Top-down cascades can result from the feeding mechanism of *C. carpio*. They may influence algae in several ways, including alteration of nutrient concentrations through excretion, resuspension of nutrients by sediment disturbance, damaging macrophytes, which compete with phytoplankton for nutrients and light, and by direct feeding on zooplankton and thus decreasing zooplankton grazing on algae (Khan *et al*. 2003). Their introduction into new waters often results in dramatic ecological disruptions at the community and ecosystem levels, which are related to their vigorous benthic foraging. Adverse impacts have been recorded across the world, including North America, India and Australia (Zambrano *et al*. 2006; Britton *et al*. 2007). Long term monitoring of turbidity is therefore required to monitor any turbidity changes as a result of *C. carpio* feeding.

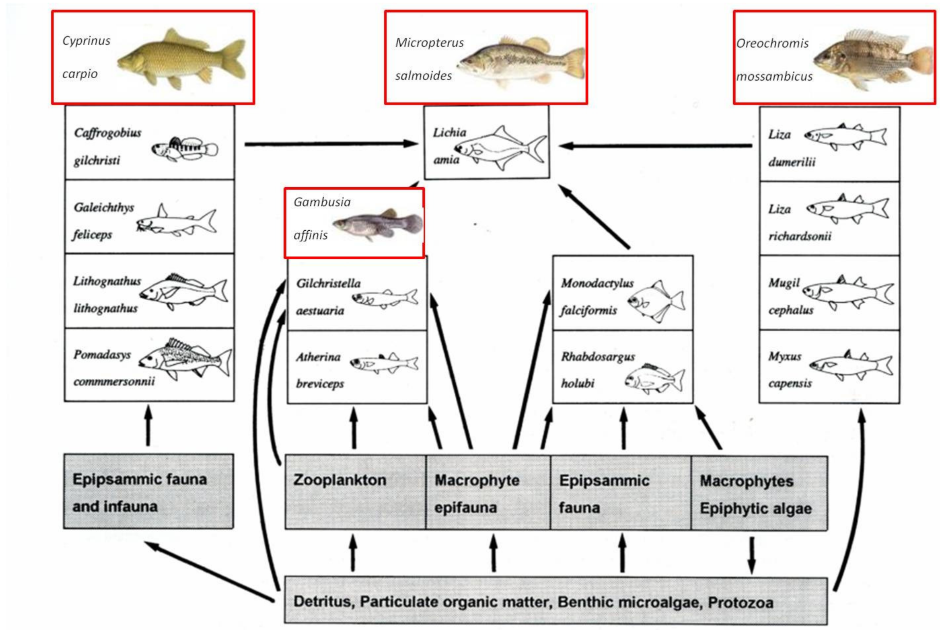


Figure 7.2. Proposed food web diagram of native fish species that utilise South African estuaries and where in the food web the alien fishes would probably fit and compete (adapted from Whitfield 1998).

Micropterus salmoides is an aggressive visual predator (de Moor 1996). Smaller fish species form a large component of their diet with, Jang *et al*. (2006) determining that fish contributed to approximately 71% of the diet of *M. salmoides* in South Korean rivers. Predation by *M. salmoides* can significantly reduce the diversity and abundance of native fish, especially in freshwater systems (de Moor & Bruton 1988; Gratwicke & Marshall 2001). In South Africa, *M. salmoides* is more commonly associated with freshwater systems than estuaries, where they feed on the native shoaling species, *A. breviceps* and *G. aestuaria* and other species that were introduced as food for them (de Moor & Bruton 1988). Weyl & Lewis (2006) determined that in estuarine systems *M. salmoides* would readily prey on smaller estuarine species like *M. falciformis*, *M. cephalus* and *M. capensis*. The impact is not only on fish species as predation on invertebrates does occur. The diet of juvenile *M. salmoides* in the Wit River, South Africa, comprised completely of invertebrates, with the most frequently consumed prey items being Libellulidae, Ephemeroptera and Chironomidae with significant decreases in abundance of larger invertebrate species in the presence of *M. salmoides* (Weyl *et al*. 2010). Wasserman *et al*. (2011) determined that the diet of larger *M. salmoides* in the Kowie River, South Africa, comprised mainly crabs and Odonata nymphs as opposed to fish. The predation pressure exerted by established populations of *M. salmoides* is usually sufficient to incur significant ecological impacts in invaded systems, including major shifts in the species composition and size structure of communities, and major shifts in the habitat utilisation, foraging behaviour and somatic growth of their prey items (Britton *et al*. 2010).

Management of alien fishes

The framework required to manage alien species involves four facets which address different phases of alien establishment. The prevention of alien introduction is the ideal scenario where there are mechanisms in place to minimise the chances of unwanted introductions. Once a species has been introduced into a system it is vital that in the early stages after introduction, the species is eradicated completely. If this is not possible the species needs to be contained. However, once a species is established, mitigation measures need to be put in place to effectively manage it in an attempt to reduce its impact on native systems.

Conventional mechanisms of alien eradication typically used include physical methods of removing fish, such as gill netting, electro fishing and long lining, and chemical methods involving the use of piscicides, most commonly Rotenone (Tweddle 2009). The status of the Wilderness Lakes system in a national park, as a RAMSAR site and the abundance of native fish species make all of these methods unviable as many of them will result in high mortality of native fish.

In the Wilderness Lakes system, prevention is no longer an option as two of the alien species are established and one is establishing. The eradication of *M. salmoides* is still a possibility as the abundances are low and the species appears to be limited mainly to Island Lake and the Duiwe River. The V-notch weir in the lower Duiwe River inhibits the movement of fish upstream. The source of introduction of *M. salmoides* from the movement of fish down the Duiwe River into Island Lake makes it critical that the point sources be located and *M. salmoides* removed from them. Once the catchment and Duiwe River are free from *M. salmoides*, the population in the system may die out naturally over time. An aspect that was not included in this study is the impact *M. salmoides* may have on the native population of the Eastern Cape redfin (*Pseudobarbus afer*) in the Duiwe River and this needs to be assessed.

The presence of *C. carpio* in the Wilderness Lakes system is of concern. It is a highly fecund, generalist species that can have significant negative impacts on native systems and their associated biota (Khan *et al*. 2003; Koehn 2004). The eradication of *C. carpio* seems unlikely unless attempts are made to fish out the adults currently in the system. This may reduce the population, therefore limiting the possibility of a 'boom'. While salinity may limit the successful reproduction of *C. carpio* it will not eradicate it. Eradication measures need to be assessed and implemented and a long term assessment of turbidity needs to be done.

The control of *O. mossambicus* and *G. affinis* is limited to mitigation measures such as assessing their impacts on the system and long term monitoring of abundance and distribution. Both species are widely distributed and highly abundant and are not limited by the physical environment. The impact of *G. affinis* on the invertebrate community needs to be assessed and monitoring of the fish community needs to continue to monitor the changes in alien and native fish abundance.

In conclusion, this study has shown that the Wilderness Lakes system has fish biota typical of other temporary open/closed estuaries in the Southern Cape with high native diversity. There are however four alien fish species in the Wilderness Lakes system. *Oreochromis mossambicus* and *G. affinis* are fully established and their abundance and distribution need to be monitored. The impact of *G. affinis* on invertebrate communities needs to be a priority. *Micropterus salmoides* is a casual species in the system and relies on repeated introductions for population continuity. The direct impact of this species on the Wilderness Lakes system is not of great concern as their numbers remain low. The impact on the population of *P. afer* in the Duiwe River needs to be investigated. Eradicating *M. salmoides* relies on eradicating the fish in the catchment. *Cyprinus carpio* is establishing in the system and with spawning occurring successfully it is likely that it will become fully established. Eradication of this species is possible, however conventional methods of alien eradication are not feasible. Further research on the invertebrate communities in the Wilderness Lakes system, and how they are being impacted by *G. affinis* predation, the predation pressure on *P. afer* from *M. salmoides* and long term changes in turbidity as a result of *C. carpio* feeding behaviour needs to be conducted. An alien management plan needs to be drawn up and implemented.

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APPENDIX 1

Length-weight relationships for fishes sampled in the Wilderness Lakes system.

Family	Species	a	b	Length	Reference
Anguilidae	*Anguilla mossambica*	0.0007	3.3	TL	Mann 2000
Ariidae	*Galeichthys feliceps*	9.55E-06	3.08	FL	Mann 2000
Atherinidae	*Atherina breviceps*	7.27E-06	3.135	SL	Froese and Pauly 1999
Carangidae	*Lichia amia*	3.18E-05	2.894	SL	Harrison 2001
Cichlidae	*Oreochromis mossambicus*	0.0311	3.029	SL	Harrison 2001
Clupidae	*Gilchristella aestuaria*	6.10E-06	3.182	SL	Harrison 2001
Cyprinidae	*Cyprinus carpio*	0.0147	3.108	FL	Froese and Pauly 1999
Elopidae	*Elops machnata*	0.0406	2.47	FL	Froese and Pauly 1999
Gobiidae	*Caffrogobius gilchristi*	0.0115	3.1777	SL	Harrison 2001
	Glossogobius callidus	0.0134	3.045	SL	Harrison 2001
	Psammogobius knysnaensis	0.0122	3.116	SL	Harrison 2001
	Redigobius dewaali	1.22E-05	3.116	SL	Harrison 2001
Haemulidae	*Pomadasys commersonnii*	1.40E-05	2.956	TL	Mann 2000
Hemiramphidae	*Hyporhamphus capensis*	1.33E-07	3.576	SL	Harrison 2001
Monodactylidae	*Monodactylus falciformis*	0.0333	2.921	SL	Harrison 2001
Mugilidae	*Mugil cephalus*	2.45E-05	2.979	FL	Froese and Pauly 1999
	Myxus capensis	1.55E-05	3.039	SL	Harrison 2001
	Liza dumerili	3.73E-05	2.858	SL	Harrison 2001
	Liza richardsonii	0.0172	3.023	SL	Mann 2000
	Liza tricuspidens	2.42E-05	2.943	SL	Harrison 2001
	juvenile Mugilidae	0.0172	3.023	SL	Mann 2000
Poeciliidae	*Gambusia affinis*	0.0082	3.31	TL	Froese and Pauly 1999
Sciaenidae	*Argyrosomus japonicus*	0.0301	2.76	TL	Froese and Pauly 1999
Soleidae	*Solea bleekeri*	1.24E-05	3.086	SL	Harrison 2001
Sparidae	*Lithognathus lithognathus*	2.28E-02	2.8562	FL	Mann 2000
	Rhabdosargus holubi	0.0305	2.92	FL	Mann 2000
Syngnathidae	*Syngnathus acus*	3.24E-07	3.074	SL	Harrison 2001

APPENDIX 2

Catch per unit effort (CPUE) expressed as number of fish sampled per haul (10 m and 30 m seine nets) or number of fish sampled per net per night (fyke net and gill net) over the sampling period from April 2010 to May 2011 in the Wilderness Lakes system, Western Cape.

Gill net

		CPUE (No. fish.net.night^{-1}) Mean ± SD				CPUE (kg. net .night^{-1}) Mean ± SD			
		RV	LV	IL	TE+SC	RV	LV	IL	TE+SC
Species	EA	n = 354	n = 580	n = 414	n = 134	n = 317.3 kg	n = 237.3 kg	n = 278.2 kg	n = 68.4 kg
Lichia amia	IIa	0.0 ± 0.2		0.3 ± 0.8	0.4 ± 0.9	0.4 ± 2.2		1.1 ± 2.9	0.4 ± 1.1
Elops machnata	IIa			0.0 ± 0.2				0.1 ± 0.3	
Pomadasys commersonnii	IIa	0.04 ± 0.2		2.8 ± 4.3	1.3 ± 1.4	0.2 ± 0.9		2.6 ± 2.4	1.1 ± 1.2
Monodactylus falciformis	IIa	3.9 ± 4.8	11.6 ± 10.5	0.6 ± 1.0	0.8 ± 2.1	0.8 ± 0.9	2.2 ±2.0	0.1 ± 0.2	0.0 ± 0.1
Argyrosomus japonicus	IIa			0.0 ±0.2				0.0 ± 0.1	
Lithognathus lithognathus	IIa	0.3 ± 0.6		1 ± 1.4	0.3 ± 0.8	1.2 ± 2.7		1.0 ±1.3	0.1 ± 0.5
Rhabdosargus holubi	IIa	0.8 ± 1.5	0.1 ± 0.4	0.3 ± 0.6		0.7 ± 1.3	0.6 ± 1.2	0.1 ± 0.4	
Mugil cephalus	IIa	0.3 ± 0.6	0.1 ± 0.3	0.5 ± 1.2	0.1 ± 0.3	0.7 ± 1.4	0.2 ± 0.6	0.7 ± 1.5	0.1 ± 0.3
Liza tricuspidens	IIb	0.1 ± 0.4		0.1 ± 0.3		0.1 ± 0.3		0.1 ± 0.3	
Galeichthys feliceps	IIb				3.5 ± 8.9				0.2 ± 0.5
Liza dumerili	IIb		0.04 ± 0.2	3.6 ± 4.9	0.1 ± 0.4		0.0 ± 0.1	1.0 ± 1.3	0.0 ± 0.1
Liza richardsonii	IIc	3.5 ± 3.8	2.4 ± 2.6	6.9 ±4.7	1.5 ±2.7	2.8 ± 2.7	1.6 ± 2.2	4.2 ± 3.4	0.7 ± 1.3
juvenile Mugilidae	II								
Oreochromis mossambicus	IV	1.2 ± 2.2	8.8 ± 14.6	0.0 ± 0.2	0.4 ± 1.3	3.1 ± 8.2	5.3 ± 9.4	0.0 ± 0.1	1.3 ± 5.1
Cyprinus carpio	IV				0.1 ± 0.3				0.01 ± 0.03
Gambusia affinis	IV								
Myxus capensis	Vb	4.6 ±5.7	1.2 ± 2.0	1.1 ± 1.5	0.4 ± 0.7	3.3 ± 4.0	0.5 ±0.9	0.6 ± 0.9	0.3 ± 0.6

30 m seine net

		CPUE (No. fish.haul^{-1}) Mean ± SD					CPUE (kg.haul^{-1}) Mean ± SD				
Species	EA	RV n = 21681	RLC n = 4356	LV n = 18751	IL n = 58184	TE n = 51439	RV n = 80.7 kg	RLC n = 52.9 kg	LV n = 79.8 kg	IL n = 179.5 kg	TE n = 158.1 kg
Hyporhamphus capensis	Ia	566.5 ± 1183.1		116.9 ± 217.3	17.1 ± 35.4	0.1 ± 0.3	2.7 ± 5.5		0.7 ± 1.2	0.3 ± 1.1	< 0.00001
Atherina breviceps	Ib	441.9 ± 1217.3	83.5 ± 84.9	159.6 ± 464.7	1309.2 ± 4886.2	858.4 ± 2156.3	0.9 ± 2.5	0.2 ± 0.2	0.5 ± 1.5	1.3 ± 4.5	0.6 ± 2.2
Gilchristella aestuaria	Ib	655.7 ± 1346.8	974.0 ± 1798.6	763.0 ± 1739.3	1006.7 ± 1195.8	551.2 ± 1397.4	1.0 ± 2.3	1.2 ± 1.2	0.7 ± 1.6	1.2 ± 1.5	0.7 ± 2.9
Caffrogobius gilchristi	Ib					0.03 ± 0.2					0.0003 ± 0.002
Psammogobius knysnaensis	Ib	0.5 ± 1.0		0.1 ± 0.2	4.5 ± 9.2	5.1 ± 12.9	0.0004 ± 0.001		0.0001 ± 0.0003	0.01 ± 0.02	0.01 ± 0.02
Redigobius dewaali	Ib				0.04 ± 0.2	0.1 ± 0.3				< 0.0001	< 0.0001
Syngnathus acus	Ib	0.4 ± 1.1			0.6 ± 1.4	1.0 ± 2.0		0.001 ± 0.002		0.001 ±0.002	0.001 ±0.002
Lichia amia	IIa				0.1 ± 0.5	0.1 ± 0.5				0.6 ± 2.5	0.1 ± 0.3
Monodactylus falciformis	IIa					0.4 ± 0.9					0.03 ± 0.06
Lithognathus lithognathus	IIa				1.3 ± 1.8	1.4 ± 3.3				1.0 ± 1.6	0.7 ± 1.6
Rhabdosargus holubi	IIa				2.3 ± 6.0	2.3 ± 6.8				0.7 ± 2.1	1.0 ± 5.0
Mugil cephalus	IIa				0.004 ± 0.2					0.02 ± 0.1	
Liza richardsonii	IIc	1.2 ± 2.5		0.6 ± 1.7	0.7 ± 2.5	0.5 ± 2.1	0.6 ± 1.1		0.5 ± 1.3	0.5 ± 1.6	0.3 ± 1.6
juvenile Mugilidae	II					2.1 ± 12.5					0.003 ± 0.02
Oreochromis mossambicus	IV	1.6 ± 3.3	18.3 ± 19.0	1.1 ± 1.9	81.5 ± 246.0	3.0 ± 9.1	0.9 ± 2.1	11.9 ± 23.5	1.0 ± 2.5	1.0 ± 4.5	0.9 ± 4.3
Cyprinus carpio	IV			0.3 ± 0.7	0.2 ± 0.6	0.1 ± 0.3			0.9 ± 2.3	0.8 ± 2.3	0.0001 ± 0.0001
Gambusia affinis	IV		13.3 ± 20.4			3.0 ± 12.9		0.002 ± 0.003			0.001 ± 0.003
Myxus capensis	Vb			0.2 ± 0.6		0.03 ± 0.2			0.1 ± 0.3		0.01 ±0.05

10 m seine net

Species	EA	CPUE (No. fish. haul $^{-1}$) Mean ± SD RV n = 2249	RLC n = 961	LV n = 33327	IL n = 9351	CPUE (g. haul $^{-1}$) Mean ± SD RV n = 7.7 kg	RLC n = 4.2 kg	LV n = 28.6 kg	IL n = 11.3 kg
Hyporhamphus capensis	Ia	2.8 ± 10.9		0.3 ± 0.7	1.4 ± 5.2	9.3 ± 39.3		0.02 ± 0.1	3.2 ± 12.8
Atherina breviceps	Ib	32.3 ± 91.1	5.8 ± 9.6	397.5 ± 631	114.7 ± 188.2	47.8 ± 186.4	4.1 ± 6.9	223.3 ± 356.9	130.7 ± 199.4
Gilchristella aestuaria	Ib	0.9 ± 4.2	13.0 ± 32.8	485.4 ± 974.1	107.4 ± 142.8	0.7 ± 3.8	8.9 ± 23.2	444.0 ± 728.2	127.0 ± 159.1
Psammogobius knysnaensis	Ib	2.6 ± 8.6	0.7 ± 1.5	0.9 ± 1.8	5.5 ± 12.1	2.2 ± 6.9	1.6 ± 3.9	1.5 ±3.6	14.3 ± 30.9
Syngnathus acus	Ib	0.3 ± 1.1			1.1 ± 2.6	0.1 ± 0.3			0.2 ± 0.6
juvenile Mugilidae	II				0.1 ± 0.5				0.1 ± 0.9
Oreochromis mossambicus	IV	21.9 ± 73.9	39.7 ± 74.4	16.5 ± 75.9	3.6 ± 20.3	148.9 ± 604.2	334.2 ± 980.1	619.2 ± 1188.4	7.4 ± 36.7
Gambusia affinis	IV		21.0 ± 34.9	0.1 ± 0.5	0.03 ± 0.2		3.0 ± 5.1	0.02 ± 0.05	0.003 ± 0.02

Fyke net

		CPUE (No. fish.net.night^{-1}) Mean ± SD						CPUE (kg.net.night^{-1}) Mean ± SD					
		RV	RLC	LV	IL	SC	TE	RV	RLC	LV	IL	SC	TE
Species	EA	n = 538	n = 827	n = 1917	n = 971	n = 478	n = 552	n = 7.1 kg	n = 11.5 kg	n = 35.1 kg	n = 33.0 kg	n = 15.3 kg	n = 19.8 kg
Caffrogobius gilchristi	Ib				0.4 ± 1.1	9.5 ± 12.1	7.7 ± 8.9				13.5 ± 34.8	0.2 ± 0.2	0.1 ± 0.2
Glossogobius callidus	Ib						0.04 ± 0.2						0.001 ± 0.003
Psammogobius knysnaensis	Ib	0.04 ± 0.2	0.1 ± 0.3					0.001 ± 0.002	0.001 ± 0.004				
Monodactylus falciformis	IIa			0.02 ± 0.1	0.1 ± 0.3	0.5 ± 1.6	1.2 ± 2.8			0.003 ± 0.03	0.009 ± 0.03	0.01 ± 0.04	0.03 ± 0.07
Solea bleekeri	IIa				0.02 ± 0.1	0.1 ± 0.4	0.1 ± 0.3				0.001 ± 0.008	0.001 ± 0.004	0.002 ± 0.01
Rhabdosargus holubi	IIa				0.1 ± 0.7	0.2 ± 0.8	0.2 ± 0.8				0.03 ± 0.1	0.1 ± 0.2	0.02 ± 0.07
Galeichthys feliceps	IIb						0.1 ± 0.3						0.01 ± 0.02
Liza richardsonii	IIc		0.1 ± 0.3		0.02 ± 0.1				0.07 ± 0.3		0.003 ± 0.02		
Oreochromis mossambicus	IV	11.2 ± 26.0	68.8 ± 97.4	39.9 ± 83.2	19.5 ± 87.6	21.5 ± 47.8	13.6 ± 46.5	0.15 ± 0.3	0.88 ± 1.24	0.73 ± 1.72	0.63 ± 2.72	0.76 ± 1.76	0.61 ± 1.90
Anguilla mossambica	Va				0.02 ± 0.1		0.04 ± 0.2				0.003 ± 0.02		0.01 ± 0.07

APPENDIX 3

Seasonal catch per unit effort (CPUE) expressed as number of fish per haul (10 m and 30 m seine nets) or number of fish per net per night (fyke net and gill net) over the sampling period from April 2010 to May 2011 in the Wilderness Lakes system, Western Cape.

Gill net

	Rondevlei				Langvlei				Island Lake				Touw Estuary			
	Mean CPUE ± SD				Mean CPUE ± SD				Mean CPUE ± SD				Mean CPUE ± SD			
Species	Autumn	Winter	Spring	Summer	Autumn	Winter	Spring	Summer	Autumn	Winter	Spring	Summer	Autumn	Winter	Spring	Summer
Galeichthys feliceps														9.3 ± 14.5	8.3 ± 14.4	
Lichia amia		0.2 ± 0.4							0.8 ± 1.3			0.3 ± 0.5		0.3 ± 0.6	1.0 ± 1.7	
Oreochromis mossambicus	1.8 ± 2.4	2.0 ± 3.5	0.3 ± 0.8	0.7 ± 0.5	1.7 ± 2.9		6.7 ± 9.5	26.7 ± 18.4				0.2 ± 0.4	1.7 ± 2.9			
Cyprinus carpio																0.3 ± 0.6
Pomadasys commersonnii		0.2 ± 0.4							2.5 ± 1.5	0.5 ± 0.8	5.7 ± 8.0	2.3 ± 1.5	0.3 ± 0.6	2.3 ± 2.1	2.3 ± 1.2	0.3 ± 0.6
Monodactylus falciformis	0.7 ± 0.8	8.7 ± 6.6	5.5 ± 2.5	0.8 ± 1.2	8.8 ± 3.8	22.8 ± 12.8	5.8 ± 5.3	8.8 ± 9.5	0.2 ± 0.4	0.5 ± 1.2	1.3 ± 1.2	0.3 ± 0.5				
Mugil cephalus	0.3 ± 0.5	0.8 ± 1.0	0.2 ± 0.4		0.2 ± 0.4	0.2 ± 0.4		0.2 ± 0.4	0.2 ± 0.4		0.7 ± 1.2	1.3 ± 2.0		0.3 ± 0.6		
Myxus capensis	2.5 ± 3.7	7.5 ± 6.7	6.2 ± 7.5	2.2 ± 3.1	1.8 ± 3.1		1.3 ± 1.7	1.7 ± 1.9	0.7 ± 0.8	2.2 ± 2.3	1.7 ± 1.0				0.3 ± 0.6	0.3 ± 0.6
Liza dumerili		0.2 ± 0.4			0.2 ± 0.4				4.8 ± 4.2	4.0 ± 8.4	1.0 ± 1.3	4.5 ± 3.3				0.3 ± 0.6
Liza richardsonii	1.0 ± 2.0	8.5 ± 3.9	2.3 ± 1.2	2.2 ± 2.0	1.3 ± 1.5	0.3 ± 0.5	4.5 ± 2.4	3.3 ± 3.0	5.3 ± 3.9	10.0 ± 6.2	4.0 ± 3.6	8.5 ± 3.3			0.3 ± 0.6	2.3 ± 3.2
Liza tricuspidens	0.2 ± 0.4		0.3 ± 0.8						0.2 ± 0.4			0.2 ± 0.4				
Argyrosomus japonicus									0.2 ± 0.4							
Elops machnata											0.2 ± 0.4					
Lithognathus lithognathus		0.3 ± 0.5	0.3 ± 0.8	0.3 ± 0.8					0.5 ± 0.5	1.7 ± 2.4	0.2 ± 0.4	1.7 ± 1.0		0.3 ± 0.6	1.0 ± 1.7	
Rhabdosargus holubi		1.5 ± 2.5	1.3 ± 1.5	0.2 ± 0.4				0.3 ± 0.8			1.0 ± 0.9	0.2 ± 0.4				

30 m seine net

Species	Rondevlei (Mean CPUE ± SD)			Langvlei (Mean CPUE ± SD)		
	Autumn	Winter	Spring	Autumn	Winter	Spring
Atherina breviceps	24.7 ± 60.4	1315 ± 2106.5	112.3 ± 108.7		140.8 ± 151.1	676 ± 1123.6
Oreochromis mossambicus	3.0 ± 4.5	0.3 ± 0.5	1.3 ± 1.5	1.0 ± 2.0	0.2 ± 0.4	1.0 ± 2.6
Gilchristella aestuaria	915.5 ± 1711.9	700.3 ± 1375.9	76.7 ± 83.7	1206.9 ± 2304.0	11.5 ± 14.2	311.4 ± 1361.9
Cyprinus carpio					0.7 ± 1.0	0.1 ± 0.6
Psammogobius knysnaensis	0.3 ± 0.5		1.3 ± 1.5		0.2 ± 0.4	
Hyporhamphus capensis	1124.3 ± 1630.0	154.3 ± 30.3	0.3 ± 0.6	207.1 ± 257.6	13.5 ± 15.8	17.7 ± 36.3
Myxus capensis					0.7 ± 0.8	
Liza richardsonii	2.2 ± 3.5	0.3 ± 0.5	0.7 ± 1.2		1.7 ± 2.7	0.1 ± 0.6
juvenile Mugilidae	0.3 ± 0.8					
Syngnathus acus	0.8 ± 1.6					

	Island Lake (Mean CPUE ± SD)				Touw Estuary (Mean CPUE ± SD)			
	Autumn	Winter	Spring	Summer	Autumn	Winter	Spring	Summer
Atherina breviceps	4694 ± 9552.1	97.0 ± 225.9	284.2 ± 396.6	400.7 ± 319.2	1142.6 ± 1454.6	194.1 ± 3880.8	338.9 ± 800.0	10.2 ± 16.7
Lichia amia	0.5 ± 0.8				0.4 ± 0.7		0.1 ± 0.3	
Oreochromis mossambicus	0.3 ± 0.5			325.8 ± 427.5	9.0 ± 16.5	0.6 ± 1.7	0.1 ± 0.3	2.4 ± 5.7
Gilchristella aestuaria	967.2 ± 923.6	64.3 ± 120.5	1533.3 ± 1610.9	1462 ± 1210.7	885.4 ± 2509.0	101.9 ± 238.0	807.3 ± 1247.5	410.0 ± 431.4
Cyprinus carpio		0.3 ± 0.8	0.2 ± 0.4	0.3 ± 0.8				0.2 ± 0.7
Caffrogobius gilchristi								0.3 ± 0.5
Psammogobius knysnaensis	13.5 ± 13.3	3.0 ± 3.9	1.3 ± 2.8	0.2 ± 0.5	7.9 ± 10.8	9.2 ± 23.2	1.4 ± 2.1	1.8 ± 3.2
Redigobius dewaali				0.2 ± 0.4	0.1 ± 0.3			0.2 ± 0.4
Hyporhamphus capensis	44.5 ± 58.1	15.0 ± 31.2	0.2 ± 6.7	4.3 ± 9.2	0.2 ± 0.7			
Monodactylus falciformis					1.0 ± 1.2	0.1 ± 0.3	0.7 ± 1.1	
Mugil cephalus		0.2 ± 0.4						
Myxus capensis								0.1 ± 0.3
Liza richardsonii		2.7 ± 4.5	0.2 ± 0.4			0.1 ± 0.3		1.9 ± 4.0
juvenile Mugilidae								8.6 ± 24.9
Gambusia affinis					0.3 ± 1.0	0.2 ± 0.7	0.2 ± 0.4	11.2 ± 24.9
Lithognathus lithognathus	1.5 ± 0.8	2.0 ± 3.0	0.2 ± 0.4	1.3 ± 1.5	1.4 ± 4.3	3.0 ± 4.8	1.0 ± 1.5	0.2 ± 0.7
Rhabdosargus holubi			6.7 ± 12.8	2.7 ± 3.9	4.3 ± 7.6	3.8 ± 9.2	0.4 ± 0.7	0.6 ± 1.1
Syngnathus acus	0.2 ± 0.4	0.5 ± 0.8		1.7 ± 2.3	0.9 ± 1.6	0.8 ± 2.3	2.2 ± 2.5	

10 m seine net

	Rondevlei (Mean CPUE ± SD)				Rondevlei-Langvlei channel (Mean CPUE ± SD)			
	Autumn	Winter	Spring	Summer	Autumn	Winter	Spring	Summer
Atherina breviceps	1.3 ± 2.1	5.4 ± 16.7	52.1 ± 145.1	60.7 ± 104.5	3.0 ± 5.2	7.0 ± 9.6		13.0 ± 15.6
Oreochromis mossambicus	0.3 ± 0.7	0.1 ± 0.3		80.5 ± 142.3	128.0 ± 117.7	9.0 ± 4.6	0.3 ± 0.6	21.3 ± 25.7
Gilchristella aestuaria	0.7 ± 2.2	0.1 ± 0.3	0.2 ± 0.6	2.5 ± 9.4	52.0 ± 53.6			
Psammogobius knysnaensis	0.3 ± 0.7		3.5 ± 03.1	5.7 ± 19.4	2.3 ± 2.5		0.3 ± 0.6	
Hyporhamphus capensis				10.3 ± 22.6				
Gambusia affinis					62.0 ± 52.7	3.3 ± 4.9	0.3 ± 0.6	18.3 ± 16.1
Syngnathus acus			0.4 ± 0.7	0.8 ± 2.2				
	Langvlei (Mean CPUE ± SD)				Island Lake (Mean CPUE ± SD)			
	Autumn	Winter	Spring	Summer	Autumn	Winter	Spring	Summer
Atherina breviceps	5.5 ± 9.5	16.5 ± 42.3	1164.4 ± 729.1	284.4 ± 332.2	197.2 ± 268.2	58.9 ± 173.4	126.3 ± 110.4	76.5 ± 163.0
Oreochromis mossambicus	4.6 ± 145.1	0.1 ± 0.3	7.4 ± 22.7	7.7 ± 13.0	0.4 ± 0.5			14.1 ± 40.2
Gilchristella aestuaria	106.2 ± 325.7		867.8 ± 1553.9	821.8 ± 695.3	18.4 ± 20.4	1.8 ± 5.3	218.6 ± 100.5	190.9 ± 187.3
Caffrogobius gilchristi							0.1 ± 0-1	
Psammogobius knysnaensis	0.2 ± 0.6	0.2 ± 0.4	2.2 ± 2.7	0.7 ± 1.8	13.2 ± 21.4	1.4 ± 1.6	2.9 ± 3.7	4.4 ± 8.2
Hyporhamphus capensis	0.1 ± 0.3		0.9 ± 1.0		4.9 ± 9.9		0.5 ± 1.1	
juvenile Mugilidae								0.3 ± 0.9
Gambusia affinis	0.3 ± 0.7	0.2 ± 0.4			0.1 ± 0.3			
Syngnathus acus					1.3 ± 1.6	2.2 ± 3.4	0.1 ± 0.3	0.4 ± 1.6

Fyke net

	Rondevlei (Mean CPUE ± SD)				Rondevlei-Langvlei channel (Mean CPUE ± SD)				Langvlei (Mean CPUE ± SD)			
	Autumn	Winter	Spring	Summer	Autumn	Winter	Spring	Summer	Autumn	Winter	Spring	Summer
Oreochromis mossambicus	16.1 ± 13.5	0.6 ± 1.2	0.9 ± 0.8	27.1 ± 46.6	113.7 ± 44.8	2.3 ± 2.1	4.3 ± 3.5	154.7 ± 152.3	6.6 ± 17.5	0.08 ± 0.5	0.02 ± 0.3	33.2 ± 127.4
Psammogobius knysnaensis			0.2 ± 0.4				0.3 ± 0.6					
Monodactylus falciformis											0.02 ± 0.3	
Liza richardsonii								0.3 ± 0.6				

	Island Lake (Mean CPUE ± SD)				Serpentine channel (Mean CPUE ± SD)				Touw Estuary (Mean CPUE ± SD)			
	Autumn	Winter	Spring	Summer	Autumn	Winter	Spring	Summer	Autumn	Winter	Spring	Summer
Anguilla mossambica			0.02 ± 0.3									0.2 ± 0.4
Galeichthys feliceps									0.2 ± 0.4		0.3 ± 0.5	
Oreochromis mossambicus	19.5 ± 167			0.04 ± 0.6	100.7 ± 64.8	3.5 ± 5.7		1.8 ± 2.4	50.7 ± 88	3 ± 2.8	0.3 ± 0.5	0.3 ± 0.8
Caffrogobius gilchristi		0.04 ± 0.4	0.08 ± 0.5	0.3 ± 2.1	14.3 ± 24.8	4 ± 3.7	9 ± 10.6	11.8 ± 8.6	2.8 ± 3	10.2 ±14	11.5 ± 4.7	6.3 ± 9.3
Glossogobius callidus										0.2 ± 0.4		
Monodactylus falciformis		0.08 ± 0.5			2 ± 3.5	0.5 ± 0.6			3 ± 5.1	1.3 ± 1.9	0.3 ± 0.8	
Liza richardsonii		0.02 ± 0.3										
Rhabdosargus holubi		0.1 ± 1.4				0.8 ± 1.5				0.8 ± 1.6		
Solea bleekeri				0.02 ± 0.3		0.3 ± 0.5	0.3 ± 0.5			0.2 ±0.4	0.3 ± 0.5	

APPENDIX 4

Olds, A.A., Smith, M.K.S., Weyl, O.L.F. & Russell, I.A. 2011. Invasive alien freshwater fishes in the Wilderness Lakes System, a wetland of international importance in the Western Cape Province, South Africa. African Zoology, 46: 179 - 184.