

**TRENDS IN THE FISH ASSEMBLAGE STRUCTURE OF TWO SOUTH AFRICAN
TRANSITION-ZONE ESTUARIES. CAN THESE TRENDS BE LINKED TO
CLIMATE CHANGE?**

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ABSTRACT

Changes in the fish communities of the Breede and Mbashe estuaries were investigated in relation to the environmental variables that influence the fish assemblages in these systems. The Breede Estuary (34° 24' 21.6"S, 20° 51' 08.2"E) occurs within the warm-temperate/cool-temperate transition-zone, while the Mbashe Estuary (32° 14' 55.4"S, 28° 54' 03.7"E) falls within the subtropical/warm-temperate transition-zone along the South African coastline. The Breede Estuary was sampled (seine and gill nets) during summer (January-March) over a period of 10 years (2002-2012), while the Mbashe Estuary was sampled (seine net) during spring (October) over a period of three years (2010-2012).

The proportion of tropical fish species was higher in the Mbashe Estuary compared to the Breede Estuary, while the Breede Estuary contained a higher proportion of temperate species than the Mbashe Estuary. Although the abundance of individual species in both estuaries varied, the ranking of species in the Breede Estuary was stable over the 10 year period. Multivariate analysis showed that the fish communities within each reach (upper, middle, lower) of the Breede Estuary remained similar, regardless of year. In contrast, species composition in the Mbashe Estuary differed significantly between years mainly due to differences in the river flow regime during the study period.

Environmental variables responsible for structuring the fish assemblage in each estuary differed. Salinity significantly impacted the spatial fish assemblage structure of the Breede Estuary, with most of the species recorded being associated with the more saline lower reaches. The abundance of tropical species near their distributional limit in the Breede Estuary appeared to be impacted by sea surface temperature (SST) as peak total abundance of

these species coincided with the warmest SST, while the lowest total abundance coincided with the coldest annual SST during the study period. The fish assemblage structure in the Mbashe Estuary was impacted by differing freshwater input prior to sampling, with flooding negatively impacting the mean total abundance, species composition and diversity in this system.

The variables that influenced the fish assemblage structure of both transition-zone estuaries will be affected by climate change in the future. These variables will, in turn, determine the composition, abundance and diversity of species within these important estuarine systems. In order to measure these impacts with any degree of understanding, long-term studies on the abiotic and biotic (including the ichthyofauna) features of these estuaries are required to interpret climate change trends.

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CHAPTER ONE

GENERAL INTRODUCTION

Short and long-term variability in environmental variables has been shown to exert considerable influence on the abundance (Mackenzie *et al.* 2007, Jennings and Brander 2010) and distribution (Mingelbier *et al.* 2001) of various aquatic species (Klyashtorin 1997, Dippner and Ikauniece 2001, Astthorsson *et al.* 2007, Pinchuk *et al.* 2008). Changes in key climate and subsequently hydrological factors (sea surface temperature, turbidity, dissolved oxygen, rainfall, freshwater flow, ocean circulation and salinity) affect the abundance and distribution of fish assemblages (Lehodey *et al.* 2006).

An increase in greenhouse gases, aerosols and burning of fossil fuels has resulted in changes to the world's climate (IPCC 2007, Brierley and Kingsford 2009). Climate change incorporates changes in temperature, rainfall and evaporation rates, sea level rise, storm frequency, ocean circulation, winds and carbon dioxide concentrations (Roessig *et al.* 2004). It has been suggested that climate change will have a variety of consequences for marine, and particularly estuarine, ecosystems (Scavia *et al.* 2002, Roessig *et al.* 2004, Pittock *et al.* 2008, Gillanders *et al.* 2011).

The effect of anthropogenic climate change is difficult to quantify, particularly in aquatic ecosystems, as it is superimposed on natural climate variability (Hulme *et al.* 1999, Rouault *et al.* 2010), and long-term time series of environmental variables are limited (Richardson and Poloczanska 2008). In some regions anthropogenic induced climate change could have less of an impact on resources than natural multi-decadal climate variability (Hulme *et al.* 1999). According to the IPCC (Intergovernmental Panel on Climate Change), human activities have caused or contributed significantly to climate change (IPCC 2007). Anthropogenic stressors such as pollution, habitat loss and disturbance are additional pressures on estuarine, freshwater and marine ecosystems that exacerbate the impacts of climate change (Brander 2010) by reducing the resilience of populations and communities in these systems (Koehn *et al.* 2011). The impact of climate change on fish stocks needs to be assessed along with these issues (Brander 2010).

The influence of climate change on marine fish is widely recognised (Frank *et al.* 1990, Roessig *et al.* 2004, Koehn *et al.* 2011). However, the effect of this change on estuarine fish assemblages has been poorly documented (Power *et al.* 2002, Selleslagh and Amara 2008, Jennings and Brander 2010). In the marine environment, climate change results in changes in biodiversity (Hughes *et al.* 2003), community composition (Holbrook *et al.* 1997, Genner *et al.* 2004), population productivity (Rose 2004), geographic range shifts, distribution and abundance of species (e.g. Berge *et al.* 2005, Daufresne *et al.* 2009, Pitt *et al.* 2010). Loss of shallow water habitats (Kennedy 1990), recruitment failure (Munday *et al.* 2008), increased frequency of marine intrusion and eutrophication (Kennedy 1990) are some additional consequences of climate change in the estuarine environment.

Biogeography

Variations in climate have been found to affect the biogeography of both terrestrial and aquatic organisms including fish (e.g. Beaugrand *et al.* 2002, Walther *et al.* 2002, Thiel *et al.* 2003, Perry *et al.* 2005, Harley *et al.* 2006, Booth *et al.* 2011). This will have consequences for estuarine associated fish taxa in South Africa, as the majority of species are restricted to certain biogeographical regions (Whitfield 1994a, Maree *et al.* 2000) with few species, such as *Mugil cephalus* (flathead mullet) and *Gilchristella aestuaria* (estuarine round herring), occurring along the entire coastline (Harrison and Whitfield 2006). This results in an uneven distribution of estuary-associated fish (Maree *et al.* 2000).

There are three recognised biogeographic regions along South Africa's coastline, a subtropical East Coast, a warm-temperate South coast and a cool-temperate West Coast (Emanuel *et al.* 1992, Turpie *et al.* 2000, Teske *et al.* 2006). The exact location of the boundaries separating these biogeographic regions are the source of some debate (Emanuel *et al.* 1992, Maree *et al.* 2000, Turpie *et al.* 2000, Harrison 2002, Lamberth and Turpie 2003), with the boundaries between these regions defined by changes in species composition and oceanographic conditions (Teske *et al.* 2006). The size and location of the three biogeographic regions in South Africa are principally influenced by the ocean currents (Sink *et al.* 2005, Whitfield 2005a). Consequently, seasonal changes in oceanographic conditions result in a seasonal shift in the biogeographic boundaries (Maree *et al.* 2000, Whitfield 2005a). Given this information, it can be assumed that with the changes that come with climate change, a similar effect on these boundaries can be expected. Possible changes to

species composition and ocean currents from climate change could result in changes to the extent of biogeographic regions and their boundaries.

Based on estuarine ichthyofauna, Harrison (2002) determined that the subtropical region extends from Kosi Bay to the Mdumbi Estuary, the warm-temperate region from the Mdumbi Estuary to Cape Agulhas and the cool-temperate region from Cape Agulhas to the Orange Estuary (Figure 1.1).

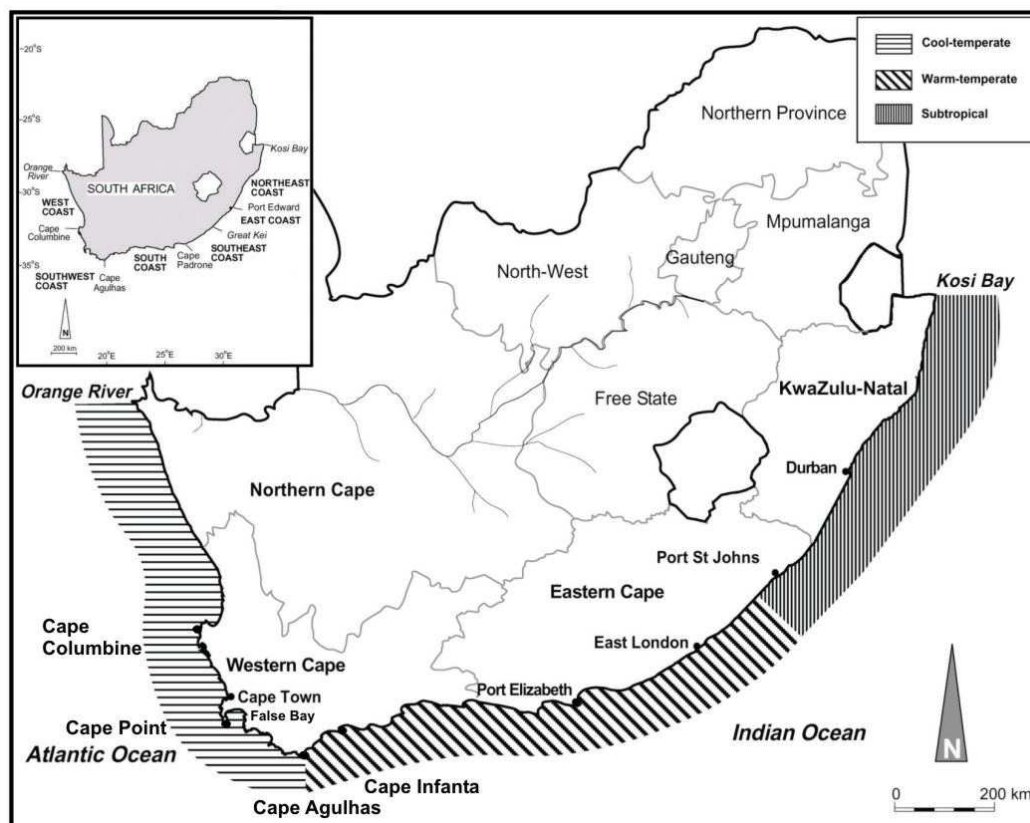


Figure 1.1. Map of South Africa showing the three estuarine biogeographical regions (after Harrison 2002).

Temperature and salinity have been determined to be the primary factors that affect South African estuarine fish biogeography (Harrison and Whitfield 2006). As such, estuarine ichthyofauna can be placed into six categories based on their biogeographical affinities (Table 1.1) (Harrison and Whitfield 2006).

Table 1.1. Biogeographical affinities of estuary-associated fish fauna in South Africa (after Harrison and Whitfield 2006).

Group	Definition
1	Tropical species that prefer warm, brackish turbid systems and are limited to subtropical estuaries.
2	Tropical species that are found in subtropical and warm-temperate estuaries.
3	Endemic species that are found in subtropical and warm temperate estuaries.
4	Endemic species that are found in warm-temperate and cool-temperate estuaries.
5	Temperate species that occur in cool-temperate estuaries and cool clear warm-temperate estuaries.
6	Widespread species that occur in all estuaries throughout South Africa

Boundaries between biogeographical regions are commonly known as transition zones (Morrone 2010) and can either be a sharply defined region or a broad zone (Williams 1996). Transition zones are areas of rapid environmental variability (Attrill and Rundle 2002) and species turnover, resulting in increased levels of species richness (Spector 2002). Transition zones are therefore considered to be biodiversity hotspots (Araújo 2002) as they are biogeographic locations with an abnormally high number of species (Konar *et al.* 2010). These locations have been investigated as key areas for conserving biodiversity (Araújo 2002, Konar *et al.* 2010).

Transition zones are areas characterised by species overlap between two biogeographical regions (Konar *et al.* 2010), and due to the environmental variability in such zones, these zones are characterised by a high variability in species composition and abundance (Brodeur *et al.* 2003). Fish assemblages occurring in these zones comprise species from both biogeographical regions as well as other, more widespread species (Morrone 2010). Species occurring in these transition-zones are often near their range limit and typically show more variability in abundance between years (Kling *et al.* 2003). Additionally, the population of such species becomes smaller and more fragmented at their range edge (Bridle and Vines 2006).

Alterations to the physico-chemical characteristics of the hydrology of marine or estuarine transition zones are likely to favour one region's assemblages more than another which can result in a community shift. This can be seen in the community structure of Solitary Island Marine Park (SIMP), which falls within a temperate-tropical transition zone associated with the warm East Australian Current (EAC) flowing offshore (and southward), while the cooler

inshore waters flow northward. The EAC is predicted to intensify with changing temperature, which will cause it to move inshore and will decrease the number of temperate species, resulting in a community shift towards tropical species and a decrease in biodiversity in the SIMP (Malcolm *et al.* 2010).

From the above it is apparent that transition zone estuaries are ideally placed to monitor changes in the abundance and distribution of fish assemblages arising from both natural and anthropogenic driven changes in climatic and hydrological variables. Unfortunately, published studies focusing on the possible effects of climate change on fish assemblages in estuaries that occur in biogeographical transition zones in South Africa are limited (Mbande *et al.* 2005). The effects of abiotic variables (salinity, water temperature, freshwater flow, dissolved oxygen, turbidity, mouth phase, biogeography and habitat variability) on estuarine fish assemblages in estuaries have been documented (Blaber and Blaber 1980, Marais 1988, Whitfield 1996, Grange *et al.* 2000, Whitfield 2005b) and this should allow for reasonable predictions of future climate change on fish assemblages in South African transition zone estuaries. This chapter will discuss the potential effects that climate change will have on four key factors that can affect the fish community structure in South African estuaries. These are temperature, freshwater input, storm frequency and sea level rise.

Potential impacts of global climate change on estuaries and their ichthyofauna.

Water temperature

Temperature is the primary factor controlling physiological and life-history processes in fish (e.g. Pauly 1994, Thiel *et al.* 1995, Harrison and Whitfield 2006) as it effects growth, recruitment, reproduction, feeding, swimming ability, developmental rate, mortality and distribution (e.g. Brander 2010, Drinkwater *et al.* 2010, Fodrie *et al.* 2010). Water temperature also indirectly affects food quality and availability (Drinkwater and Frank 1994). As a result, changes in fish assemblages across seasons and years vary with natural sea temperature variation (Kennish 1990). Fish are obligate poikilotherms that behaviourally thermoregulate by remaining in areas within their thermal preference (Kennish 1990, Selleslagh and Amara 2008). Water temperature is a factor that determines temporal changes in estuarine fish assemblages (Pombo *et al.* 2005, Selleslagh and Amara 2008). It has been recorded as the key environmental factor to affect the temporal abundance and species composition of the fish assemblages of the Elbe Estuary, Germany (Thiel *et al.* 1995).

Similarly, Marshall and Elliot (1998) found temperature to be the best predictor of seasonal total abundance in the Humber Estuary (England). It has also been noted that the highest ichthyofaunal densities in KwaZulu-Natal estuaries are generally associated with warm waters in these systems during summer (Whitfield 1994b).

Increasing air temperatures, resulting from increasing greenhouse gases and aerosols (IPCC 2007) have caused a general trend of increasing water temperatures (Schumann *et al.* 1995, Scavia *et al.* 2002, IPCC 2007, Brierly and Kingsford 2009). This has led to warming of the world's rivers, estuaries and oceans (Rijnsdorp *et al.* 2009) which in turn has affected the physico-chemical properties of these water bodies (Kennedy 1990). On a local scale, the Agulhas Current has been warming up by 0.7 °C per decade since the 1980s (Rouault *et al.* 2009), which combined with the predicted strengthening of atmospheric circulation and wind stress from increasing air temperatures, will result in warmer water flowing south (Lutjeharms *et al.* 2001). However, sea surface temperature (SST) data from 1982 to 2009 shows the west coast SST is cooling by -0.55 degrees per decade between January and February. Cooling was also recorded from Port Elizabeth to Port Alfred by up to -0.4 degrees per decade from May to August (Rouault *et al.* 2010).

Increasing water temperature will affect species differently (Koehn *et al.* 2011) as different species have different thermal preferences (Pankhurst and Munday 2011). Increasing sea surface temperatures will shift the distributional range of many mobile species (Sagarin *et al.* 1999, Perry *et al.* 2005, Clark 2006, Sorte *et al.* 2010), while other fish species could face mass mortalities if favourable conditions and habitats disappear (Roessig *et al.* 2004).

Changes in the distributional range of species will result in changes in the composition of species assemblages within estuaries (Elliott 2002, McGinn 2002, Clark 2006). It is predicted that species will show a poleward shift in their distribution (Barry *et al.* 1995). The rate at which species expand or contract their range, varies among and within species (Perry *et al.* 2005). These shifts in fish community structure will be more apparent in fish populations found near their range limit (Perry *et al.* 2005, Pörtner and Peck 2010, Gillanders *et al.* 2011) as these populations are more likely to be influenced by abiotic and density-independent factors such as water temperature and this makes the impact, if any, easier to understand (Myers 1998). Predicting the relative abundance of species that occur with range shifts is very difficult (Daufresne *et al.* 2009). Most species that frequent estuaries in Southern Africa

are marine migrants and fairly widespread and as a result few extinctions are expected as a result of increasing water temperatures (Clark *et al.* 2008).

Increasing water temperatures may also affect timing of spawning, reduce egg size, affect the number and quality of offspring, as well as affect their developmental rates (Pörtner and Farrell 2008). Earlier spawning of fish species may result in a mismatch between hatching larvae and food resources, which would impact their survival (Beaugrand *et al.* 2003, Pankhurst and Munday 2011), and govern the sustainability of populations (Pankhurst and Munday 2011).

Freshwater input

Freshwater flow has a profound effect on the physical, chemical and biological properties of estuarine and coastal systems (Skreslet 1986, Drinkwater and Frank 1994, Ter Morshuizen *et al.* 1996, Whitfield 2005b, Wolanski 2007), which affects the community structure and functioning of these systems (Sklar and Browder 1998). Freshwater flow has direct and indirect effects on fish in estuaries through changes in salinity, turbidity, temperature, dissolved oxygen concentrations, nutrient concentrations (Sklar and Bowder 1998, Whitfield 1999, Alber 2002, Clark 2006), sediment delivery (Eyre 1998) and olfactory cues (Whitfield 1999). Climate variability influences the quality and quantity of freshwater entering estuarine systems by altering precipitation and evaporation rates (Barnston and Livezey 1987). This variability in freshwater flow, which alternates between wet and dry cycles, influences the distribution and abundance of fish communities through its effect on habitat availability and productivity (Kimmerer 2002, Lamberth *et al.* 2009). Natural variation in freshwater flow has been shown to affect the early life history stages of many fish species (Gillson 2009). Similarly, the abundance, growth and mortality of age-0 *Sciaenops ocellatus* in the Suwanee River estuary, North America, was linked to changes in freshwater flow (Purtlebaugh and Allen 2010). Freshwater flow also influences growth through food availability, salinity regimes and water temperature in estuarine systems (Robins *et al.* 2006). However, estuaries require sufficient freshwater to maintain biogeochemical processes (Skreslet 1986).

Variation in freshwater flow affects nutrients and organic matter within estuaries (e.g. Allanson and Read 1995, Kimmerer 2002) which in turn impacts the extent of estuarine primary and secondary productivity (Mallin *et al.* 1993). It also influences the estuary's

suspended sediment load along with detritus. Detritus is an important food source in estuaries (Schlacher and Wooldridge 1996a).

Recruitment of certain marine post-larvae and juvenile fish into estuaries is thought to be due to olfactory cues (Strydom 2002, James *et al.* 2008a), from either freshwater or estuarine sources (Stabell 1992) being transported to the marine environment. A positive relationship between river flow and recruitment has been found for many species (Drinkwater and Frank 1994). It was found that river plumes leaving the Kabeljous Estuary to the adjacent surf zone in the Eastern Cape Province resulted in high densities of estuary-associated fish larvae associated with these plumes (Strydom 2003). Similarly, Whitfield (1994b) found that Eastern Cape estuaries with sufficient freshwater input had the highest postlarval densities of newly recruited marine species. Furthermore, Martinho *et al.* (2007) recorded a higher abundance of estuarine resident and estuarine associated species in years with higher freshwater flow in the Mondego Estuary (Portugal).

Freshwater input affects community composition by altering the distribution and abundance of estuarine and freshwater species due to changes in salinity (e.g. Keup and Bayliss 1964, Rogers *et al.* 1984, Garcia *et al.* 2003a, Whitfield and Harrison 2003). Differences in salinity tolerances between stenohaline and euryhaline species have been attributed to the responses of these species to freshwater flow (Able 2005). Estuarine resident species are generally able to tolerate a wide range of salinity regimes including the freshwater environments in the headwater regions of estuaries (e.g. Araujo *et al.* 1999, Hoeksema and Potter 2006, Gillanders *et al.* 2011). It has been suggested that river pulses entering the estuarine environment are responsible for the abundance of estuarine spawners (Whitfield 1999). Frequent fresh water pulses maintain the biological productivity in estuarine and coastal systems (Flint 1985, Martin *et al.* 1992). Freshwater input also affects the duration and frequency that an estuary is open or closed by adjusting the rates of sediment delivery (Reddering and Rust 1990). The mouth status of an estuary affects the distribution, abundance and species composition of marine fish in estuaries (Whitfield and Kok 1992, Vorwerk *et al.* 2003, James *et al.* 2008b).

Climate change is expected to alter precipitation patterns globally and this will affect freshwater input into estuaries (e.g. Arnell 1999, Dore 2005, Gillanders *et al.* 2011). Understanding the effects of altered spatial and temporal precipitation patterns resulting from

climate change are of the utmost importance to understanding the future status of South African estuaries (Clark *et al.* 2008). Seasonal changes to precipitation may alter estuarine mouth dynamics, which will have a negative impact on recruitment in these systems. The effects of altered freshwater flow on estuaries will also depend on the type of estuary (Gillanders and Kingsford 2002, Clark *et al.* 2008).

Changes to the quality and rate of timing of freshwater delivery to estuaries could exacerbate the existing human modifications to these flows (Alber 2002). For Southern Africa, downscaled high-resolution numerical regional climate models (RCM), which are derived from global circulations models, have predicted a significant decrease in rainfall in the south-western Cape, and an increase in rainfall over the eastern part of South Africa (Engelbrecht *et al.* 2009). Heavy rainfall events during summer have been predicted to increase for KwaZulu–Natal as part of the climate change scenario (Hewitson *et al.* 2005). Long-term climate predictions have sources of uncertainty that must be considered when interpreting results. The unpredictability which results from the different GCMs produces various scenarios on the effects of climate change (Ambrosinho 2011). Other factors that affect climate, such as solar and volcanic forcing as well as greenhouse gases, are unpredictable. It is believed that predictions of climate change cannot deal with the uncertainty that comes from the unpredictability of such factors (Ambrosinho 2011).

Decreased precipitation and the resultant reduction in freshwater flow in the south-western Cape will have key consequences for estuaries and associated fish species (Clark *et al.* 2008). The effects of reduced freshwater input will vary according to estuary type and locality (Harrison 2004). Indirect effects of decreased precipitation and reduction in flow is the effect on the physico-chemical properties of the system which in turn, will affect the distribution and abundance of estuarine species (Drinkwater and Frank 1994). These include changes in sedimentary processes, depth profiles, mouth configuration and tidal prism within an estuary. Changes in nutrient levels, suspended particulate matter, temperature, dissolved oxygen and turbidity could occur with a reduction in flow (Drinkwater and Frank 1994).

A decrease in freshwater flow will reduce olfactory cues entering the adjacent coastal surf-zone which may affect recruitment (Gillanders and Kingsford 2002, James *et al.* 2008b). Furthermore, reduced nutrient and detrital input (Drinkwater and Frank 1994) from reduced freshwater flow (Gillanders and Kingsford 2002) affects primary productivity through

decreased phytoplankton stocks (Drinkwater and Frank 1994). This will have an effect on the rest of the food web. Zooplankton is an important food source of larval and post-larval fishes in South African estuaries (Whitfield 1985) and these stocks generally reflect the status of phytoplankton stocks, which would have serious consequences for many estuarine dependent marine fish should primary and secondary production be affected by reduced freshwater flow (Grange 1992).

A reduction in freshwater flow coupled with intrusion of marine waters could result in an estuary becoming an extension of the sea (Allanson and Read 1995). In addition, decreased flow will lead to an influx of marine sediment, which will increase the risk of sand bars forming in the lower reaches and mouth region of the estuary (Whitfield 1999). This will restrict access to marine migrant species that use these systems (Clark 2006). In severe cases, mouth closure could occur and this would alter the functioning of the estuary as well as change the fish communities as recruitment will cease and species richness and abundance will decrease (Sinclair *et al.* 1986, Vorwerk *et al.* 2003).

Species composition during prolonged mouth closure would be negatively impacted and the system is likely to become dominated by estuarine species (Kok and Whitfield 1986). Increased salinity due to reduced freshwater input will result in temporal changes in the fish communities with reduced species richness and diversity in estuarine fish assemblages, recorded as salinity is an important factor in fish utilization of estuaries (Marais 1988). Minimal freshwater and marine inputs combined with high evaporation rates can result in hypersaline conditions (Potter *et al.* 2010). This can occur within certain temporarily/open closed systems in South Africa where reduced freshwater flow, particularly during summer, may increase the duration of mouth closure. However, hypersaline conditions can also occur in the upper reaches of freshwater deprived systems such as the Kariëga Estuary (South Africa) (e.g. Allanson and Read 1995, Whitfield and Paterson 2003, Wasserman 2010). This may even lead to mass mortalities as occurred in the Seekoei Estuary (Whitfield 1998). A combination of increased air temperatures and reduced freshwater input resulting from climate change could result in increased hypoxia (low dissolved oxygen concentrations) frequency (Justić *et al.* 2007). Severe cases of hypoxia can result in mortality of benthic fishes and invertebrates (Gillanders and Kingsford 2002).

Increased precipitation and increased freshwater input into estuaries will also lead to changes in fish species assemblages. Whitfield and Harrison (2003) recorded higher freshwater input in the Thukela Estuary, which resulted in a decline in fish abundance. Since mouth phase in temporarily open/closed estuaries is linked to the degree of freshwater flow entering these systems (Whitfield 2005b), high freshwater flow rates could increase the frequency and duration that these estuaries are connected to the marine environment. Alternatively, an increase in rainfall could result in increased frequency and duration of flood events, which could increase the suspended sediment, which can be lethal to fish in estuaries (Whitfield 1999). The decrease in the saline influences within the system can lead to a decline in the marine species component of the estuarine fish fauna (Whitfield and Harrison 2003).

Frequent and unseasonal flooding of an estuary could also have a negative impact on the available nursery habitats (Marais 1988). Increased river flow would increase nutrients entering a system but the residency time could decrease. This influences the amount of phytoplankton that could use these nutrients. This would decrease the productivity of the system (Drinkwater and Frank 1994). In addition, climate-change induced changes in rainfall can disrupt the availability of food during particular seasons and this could lead to mortalities of estuarine species that rely on freshwater pulses at certain times of the year (Drinkwater and Frank 1994, U.S. EPA 2009).

Coastal Storms

Increased storm frequency has the ability to alter the hydrological regimes and sediment transport in estuaries, cause coastal erosion, and lead to the loss of established littoral vegetation (U.S EPA 2009). Mann and Pradervand (2007) recorded a relationship between the abundance of a number of estuarine-associated fish species and suitable nursery habitats. Similarly, increasing frequency and intensity of coastal storms may result in the closure of certain estuary systems, and as such, would negatively impact species diversity and fish abundance (James *et al.* 2013). This will affect the estuarine fish communities and ultimately have consequences for fisheries. Storms also alter the bottom sediment dynamics, terrestrial borne olfactory cues, nutrient availability, salinity and oxygen levels of an estuary (Brierly and Kingsford 2009). Increased storm frequency would tend to increase wind stress, possibly leading to increased upwelling in certain coastal areas (Rouault *et al.* 2009), which could act as a barrier to the range expansion of warm water species (Harrison 2003).

Rationale for this study

There has been comparatively little research on the impacts of climate change on fish species and much of the knowledge of the potential impacts on coastal and estuarine fish species has been inferred from research on the role of climate or environmental variability in structuring biological communities and the physiological responses of fish to predicted climatic stressors under controlled laboratory conditions (Hobday *et al.* 2008, Rijnsdorp *et al.* 2009). The shortfall with this type of research is that climate variability operates on smaller time-scales (annual or decadal) than climate change (many decades or longer). This approach requires time-series of biological and physical data covering more than one cycle of the climate variability pattern (Hobday *et al.* 2008). However, adequate medium to long-term estuarine studies including both biological and physical data sets are scarce (Haedrich 1983, Flint 1985, Wolfe *et al.* 1987).

The estuaries that were sampled were the Mbashe and Breede estuaries. The Mbashe Estuary occurs in the subtropical/warm-temperate transition-zone, while the Breede Estuary falls within the warm-temperate/cool-temperate transition-zone. These estuaries were selected as they are both the focus of long-term monitoring programmes. The Breede Estuary has been monitored since 2002, and will form the main focus of this thesis. The Mbashe Estuary has been sampled since 2010 and results from this chapter are preliminary.

Aim and hypotheses

The primary aim of this study was to document temporal and spatial changes in the fish assemblage structure of the Mbashe and Breede estuaries and relate them to climatic or environmental variability.

Specifically it was hypothesised that:

The fish assemblage in the Breede and subtropical Mbashe estuaries undergo temporal changes in abundance, composition and diversity driven by changes in key variables, particularly temperature, salinity and rainfall (Chapter 2 and Chapter 3).

Chapter 4 consists of a general discussion of the study. The key findings are summarised and potential climate change effects on the two estuaries are discussed.

CHAPTER 2

FISH ASSEMBLAGE TRENDS IN THE BREEDE ESTUARY

INTRODUCTION

The interaction of fresh and marine waters in estuaries gives rise to highly dynamic systems. Consequently, estuaries are characterised by fluctuations in environmental conditions (Flint 1985) with fish abundance and composition exhibiting spatial and temporal variability (Garcia *et al.* 2004, Akin *et al.* 2005). Estuaries provide a variety of benefits to fish including an abundance of food for juvenile fish, critical habitats and reduced predation risk (Claridge *et al.* 1986, Whitfield and Kok 1992, Maes *et al.* 1998). The benefits of living in estuaries are only available to those species that are broadly tolerant of the changing environment and as such, fish assemblages within estuaries are typically characterised by low diversity but a high abundance of individual species (Whitfield 1999). Fish species found in estuaries include resident, migratory and marine species (Franco *et al.* 2008). Estuaries also act as a nursery area for many larval and juvenile estuary-associated fish species (e.g. Beckley 1984, Cyrus and Blaber 1987, Whitfield 1999, Strydom *et al.* 2003), and these nursery areas play a crucial role in the survival of many species, some of which are recreationally and commercially important (e.g. Griffiths 1996, Lamberth and Turpie 2003, Kruger and Strydom 2010).

Numerous factors affect the utilisation of estuaries by fish, including water temperature (e.g. Thiel *et al.* 1995, Marshall and Elliot 1998, Harrison and Whitfield 2006), turbidity (e.g. Cyrus and Blaber 1987, Marais 1988), dissolved oxygen (e.g. Maes *et al.* 1998, Whitfield 1999), salinity (e.g. Loneragan *et al.* 1987, Marais 1988, Martinho and Able 2003, Harrison and Whitfield 2006) and river flow (e.g. Rogers *et al.* 1984, Ter Morshuizen *et al.* 1996, Whitfield and Harrison 2003, Lamberth *et al.* 2008). Other important factors such as mouth phase (e.g. Whitfield 1999, Vorwerk *et al.* 2003, James *et al.* 2008b), zoogeography (e.g. Whitfield 1999, Harrison 2005), estuary type and size (Vorwerk *et al.* 2003); as well as habitat variability (Whitfield 1999) also play a role in estuary utilisation. However, climate change is expected to alter many of these factors (Kennedy 1990), and may have a range of implications for estuary-associated fishes.

The Breede Estuary is an important nursery area for many fish species and contains a high proportion of endemic taxa (Harrison 1999, Lamberth *et al.* 2008). It is also the largest warm-temperate estuary along the South African coastline (Turpie *et al.* 2002a) and is in close proximity to the biogeographic boundary between the warm-temperate and cool-temperate regions which occurs at Cape Agulhas (Harrison 2002). It is therefore likely to fall within the warm-temperate and cool-temperate transition-zone. The ichthyofauna of the Breede Estuary has been well studied over time scales that have varied from once-off (Carter 1983, Harrison 1999 to seasonal studies over a period of less than a year (Montoya-Maya and Strydom 2009) and short-term studies (<5 years) (Lamberth *et al.* 2008, Coetzee and Pool 1991). However, the relationship between changes in the fish community and climatic or environmental variables in the Breede Estuary is largely unknown. A study by Lamberth *et al.* (2008) determined the effects of different future flow scenarios on the Breede Estuary ichthyofauna. Montoya-Maya and Strydom (2009) determined the correlation between temperature, salinity and water transparency and the densities of larval fish species in nine south and west coast estuaries, which included the Breede Estuary.

The primary aim of this study was to assess changes in the fish assemblage structure over a 10 year period relating to climate or environmental variability. It was hypothesized that the Breede Estuary fish assemblage undergoes short-term temporal changes in composition, diversity and abundance as a result of inter-annual variability in key climatic variables, particularly temperature, salinity and freshwater flow. In addition to an assessment of the overall estuarine fish communities from different years, all species were placed into functional groups based on estuary association and biogeographic origin in order to determine whether any group specific patterns were evident in relation to physico-chemical conditions.

MATERIALS AND METHODS

Study site description

The coastal region from Cape Agulhas to Cape Infanta is a Mediterranean climate region which is characterised by winter rainfall and dry summers (Raal and Burns 1996). The rainfall patterns become bimodal east of Cape Infanta (Heydorn and Tinley 1980). The Breede River is 322km long (DWAF 2004, Lamberth *et al.* 2008) and falls in the transition zone between the bimodal and winter rainfall regions, but most of the 12600 km² catchment

falls in the winter rainfall region (Carter 1983, Lamberth *et al.* 2008). The Breede Estuary thus experiences greater flow variability than adjacent systems. Approximately 80% of the rainfall falls between April and September for the Breede River (Steynor *et al.* 2009) and the estuary falls within an area that receives 400-500 mm per annum (DWAF 2004). The mean annual run-off (MAR) is $1785 \times 10^6 \text{ m}^3$ per annum (Van Niekerk *et al.* 2012a) and has been reduced from reference levels due to impoundments on the tributaries and catchment area (Taljaard *et al.* 2001, Lamberth *et al.* 2008). The majority of the catchment area consists of natural vegetation (64%), while agriculture comprises 34% and only 1% is degraded shrubland. Less than 1% of the catchment area consists of urban development (Harrison *et al.* 2001).

The Breede Estuary (34° 24' 21.6"S, 20° 51' 08.2"E) is a permanently open estuary located $\pm$ 7 km north of Cape Infanta (Figure 2.1). The estuary has a total surface area of 455 ha and a mean depth of 4.6 m, with 17 m deep scour holes occurring in the middle and upper reaches (Lamberth *et al.* 2008). The estuarine bed consists of three distinct zones: an estuarine sand zone that extends from the mouth to 5.5 km upstream, an estuarine mudzone that ranges from 5.5 km to 18.5 km upstream and a fluvial sand zone that extends from 18.5 km to 35 km upstream (Carter 1983). The gentle gradient of the coastal plain allows the tidal effects to penetrate 50 km upstream as far as Malgas (Day 1981, Carter 1983, Lamberth *et al.* 2008).

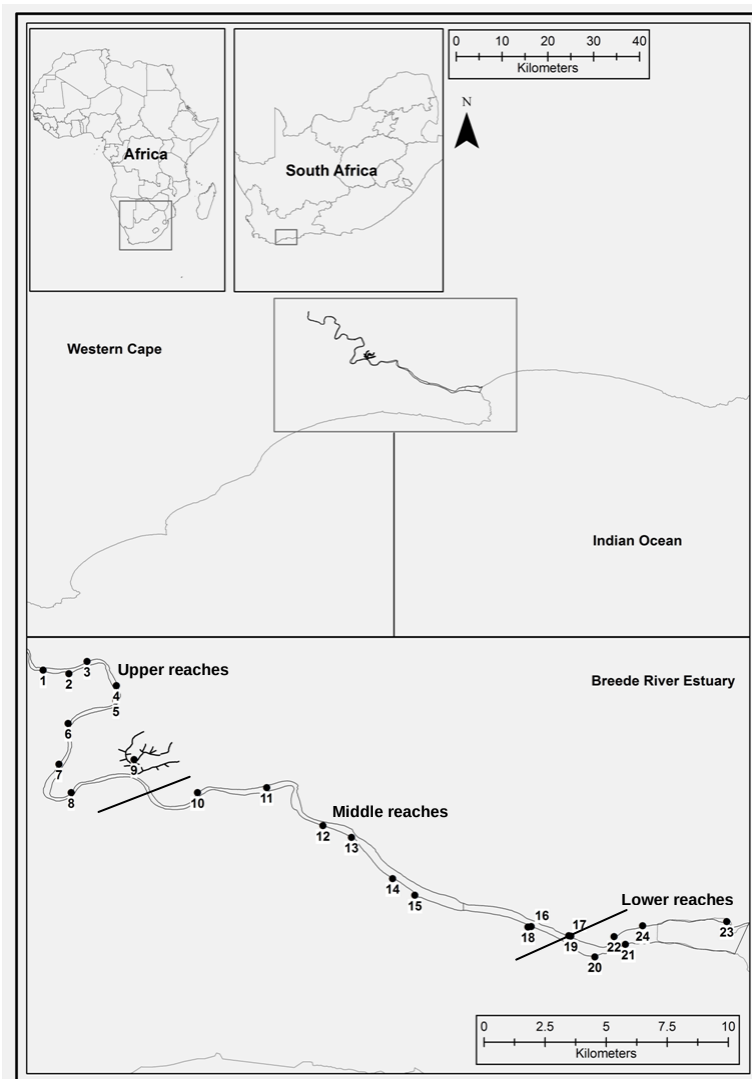


Figure 2.1. Map showing the location of the Breede Estuary along the South African coastline and the location of the 24 sampling sites in the three estuarine reaches.

Catchment and coastal-scale environmental variables

Hydrological data (freshwater flow rates) for the Breede River were obtained from the South African Department of Water Affairs (DWA) (www.dwaf.gov.za/Hydrology/). The verified mean monthly flow rate data (January 1968-March 2012) were obtained from two stations that contribute to the overall Breede River flow rates: namely the Swellendam H7H006 (34°04'03" S, 20°24'20.16" E) and Eenzaamheid Buffelsjas H7H013 (34°01'9.1" S, 20°34'58.08" E) stations. Mean monthly flow rates were compared to historical mean monthly values (1970-2010) to determine higher than average flow rates (above historical average) and lower than average flow rates (below historical average). The catch per unit

effort (CPUE) of the most abundant species and monthly flow rates were compared to determine if different flow rates had any impact on the CPUE of species.

Sea surface temperature (SST) values were obtained from the Pathfinder version 5.0 sea surface temperature (SST) dataset. The ocean pathfinder SST data set is a reprocessing of global SST data from the advanced very high resolution radiometer (AVHRR) on board NOAA (National Oceanic and Atmospheric Administration) satellites. Daily sea surface temperature data (SST) of Pathfinder data are available at a 4 km resolution and are accurate within 0.03 °C (Kilpatrick *et al.* 2001). Data were downloaded from the NOAA (National Oceanic and Atmospheric Administration) CoastWatch program (http://las.pfeg.noaa.gov/oceanWatch/oceanwatch_safari.php). Monthly summer means (January- March) were calculated for the 4 km² block (34° 23' 24" S 20° 53' 60" E, 34 ° 28' 30"S, 21° 06' 00" E) incorporating the Breede Estuary mouth for the period January 1982-March 2012.

Field sampling

The Breede Estuary has been sampled up to four times a year since 1997 as part of a long-term monitoring programme (S.J. Lamberth unpublished data). The summer data (January– March) collected between February 2002 and February 2012 from this monitoring programme were utilised in this chapter. Autumn, winter and spring data were excluded from the analysis as these seasons were not consistently sampled each year. The Breede Estuary was sampled at 24 sites, unless impractical due to factors such as site flooding (Figure 2.1). At each site water temperature (°C), salinity (expressed as practical salinity units), pH, turbidity (NTU), secchi depth (cm) and dissolved oxygen (mg l⁻¹ and percentage) were recorded with a YSI multi-parameter meter.

A seine net (30 m x 1.7 m x 15 mm bar mesh seine net with a 5 mm bar mesh purse) was used to sample littoral habitats at the 24 sites, while gill nets that were 30 m long and 2 m deep with stretched mesh sizes of 44 mm, 48 mm, 51 mm, 54 mm, 75 mm, 100 mm and 144 mm were used to sample the ichthyofauna at fixed sites (Site 1, 7, 11, 13, 17, 20 and 21). Seine-netting was carried out during daylight hours. The seine net was taken out in a row boat and placed in a semi-circle formation and subsequently hauled to shore by four people. Gill netting occurred during daylight hours in the channel at each site.

Fish collected in the seine nets were placed into buckets of estuarine water and then as far as possible sorted to the lowest taxonomic level (usually species). Once the fish were identified they were measured (mm TL) and returned to the system. Specimens caught in the gill nets were identified to the lowest taxonomic level (usually species), measured (mm TL) and where possible returned to the system.

Data analysis

Environmental variability

Temperature ($^{\circ}\text{C}$), salinity and dissolved oxygen (mg l^{-1}) were used to determine the lower, middle and upper estuary reaches within the Breede Estuary using Principal Component Analysis (PCA) in the PRIMER v 6.1.6 (Plymouth Routines in Multivariate Ecological Research) package (Clarke and Warwick 2001). All three variables had to be measured at sites to be included in the analysis (2002, 2004, 2005, 2009 and 2011) and as a result some years were omitted. A draftsman plot was generated to visually assess normality and determine transformation choice for each environmental variable (Clarke and Gorley 2006). Salinity was log transformed ($\text{Salinity} + 0.5$). Once the transformation of salinity values had corrected the distribution of the data, all three variables values were then normalised to transform variables to equal scales (Clarke and Warwick 2001) and a PCA plot generated. The sites were then placed into an estuary reach, based on the ordination diagram.

Temperature ($^{\circ}\text{C}$), salinity and dissolved oxygen (mg l^{-1}) data collected from each site were allocated to the lower, middle and upper reaches as determined by the PCA plot. The data were tested for normality using the Shapiro-Wilk test and equality of variance using Levene's test. Once these two assumptions were fulfilled, a one-way analysis of variance (ANOVA) was conducted on the data to determine if the mean values in each reach were significantly different from each other. If either of the assumptions were not met then the non-parametric Kruskal-Wallis test was utilised. If significant statistical differences were recorded using the ANOVA analysis, then a post-hoc Tukey's test was used to determine which estuary reaches were significantly different from each other. The post-hoc Mann-Whitney U test was used on data analysed by the non-parametric Kruskal-Wallis test.

Richness and diversity

Data collected from both the seine and gill nets were combined to investigate inter-annual trends in species richness and diversity over the period 2002-2012. Thereafter, only seine net data were analysed. All alien species and those only identified to family were removed from the analysis. The translocated species recorded during this study i.e. *Oreochromis mossambicus* was not considered an alien species and as such was included in the analysis. Species sampled in seine nets were allocated an estuary-association category, according to the classification system of Whitfield (1994a) (Table 2.1). Species were also grouped according to their biogeographical affinities, based on the six categories outlined by Harrison and Whitfield (2006) (Table 1.1).

Table 2.1. Estuary-association categories of fishes utilising southern African estuaries (after Whitfield 1994a).

Categories	Description of categories
I	Estuarine species which breed in southern African estuaries. Further divided into: Ia. Resident species which have not been recorded spawning in the marine or freshwater environment Ib. Resident species which also have marine or freshwater breeding populations.
II	Euryhaline marine species which usually breed at sea with the juveniles showing varying degrees of dependence on southern African estuaries. Further divided into: IIa. Juveniles dependent on estuaries as nursery areas. IIb. Juveniles occur mainly in estuaries but are also found at sea. IIc. Juveniles occur in estuaries but are usually more abundant at sea
III	Marine species which occur in estuaries in small numbers but are not dependent on these systems
IV	Euryhaline freshwater species, whose penetration into estuaries is determined primarily salinity tolerance. Includes some species which may breed in both freshwater and estuarine systems.
V	Catadromous species which use estuaries as transit routes between the marine and freshwater environments. Further divided into: Va. Obligate catadromous species which require a freshwater phase in their development Vb. Facultative catadromous species which do not require a freshwater phase in their development

Spatial and temporal trends

Catch per unit effort (CPUE) was used as the index of relative abundance and was given as fish per haul. Catch per unit effort values per sampling occasion and estuary reach for each species were calculated by dividing the number of individuals by the total number of seine

hauls. The percentage contribution of individual species to the summer fish community was calculated by dividing each species CPUE by the total species CPUE.

To study changes in species diversity multivariate diversity indices based on taxonomic distinctness were calculated (Δ^* and Δ^+). Taxonomic diversity measures have been found to be more sensitive to environmental change and less sensitive to sample size or sampling effort than traditional diversity indices (e.g. Simpsons, Shannon-Weiner and Pielou) (Warwick and Clarke 1998). Taxonomic distinctness is expected to decrease with increasing disturbance (Collie *et al.* 2004). Stability of the estuarine fish assemblage was calculated using the non-parametric Kendall's coefficient of concordance (W), which tested for significant rank differences based on the abundance of species in the nine years of sampling. Values ranged from 0 (no agreement) to 1 (complete agreement) (Kendall 1962). Spearman rank correlation coefficients determined the similarity of the fish assemblages between years. A Bonferroni correction was applied to the level of significance in order to account for multiple comparisons (corrected $P < 0.001$).

Spatial and temporal trends of the fish assemblages of the Breede Estuary were analysed using the PRIMER v 6.1.6 (Plymouth Routines in Multivariate Ecological Research) package (Clarke and Warwick 2001). The annual summer seine net CPUE data from the three different reaches (lower, middle and upper) were fourth root transformed to reduce the weighting differences between common and rare species. An association matrix was produced using the Bray-Curtis similarity measure. Clusters analysis using group average hierarchical sorting and non-metric multidimensional-scaling (nMDS) plots were used to determine the spatial fish assemblage structure over the study period. The goodness-of-fit for the data points in the nMDS was measured by a stress coefficient. A stress value of <0.2 gives a potentially good 2-dimensional picture, <0.1 stress is a good ordination, <0.05 is an excellent representation and a stress value of 0 means the data is perfectly represented. Two-way crossed analysis of similarity (ANOSIM) was carried out to determine if there were differences between estuary reach and year. ANOSIM determines the Global value R , which indicates the degree of similarity between groups. Values of R range between -1 and 1; the more similar the replicates within a sample are to each other, then the closer to a value of 1 they become. If significant differences in values were found, then a similarity percentages analysis (SIMPER) was used to identify the species primarily responsible for those differences.

nMDS analyses indicated assemblage differences between the lower, middle and upper reaches regardless of time. Canonical correspondence analysis (CCA) from the CANOCO v4.5 (Ter Braak and Smilauer 2002) software package was used to assess the importance of explanatory environmental parameters in determining this pattern. In CCA the ordination axes are restricted to linear combinations of environmental variables which allow a direct relationship between community variation and environmental variation (Ter Braak 1986). The following environmental variables were used in the CCA: mean summer (January-March) estuarine temperature and salinity per reach, mean low (December – June) flow, mean high (July-November) flow and year. Annual summer fish abundance data were used to calculate CPUE in the lower, middle and upper reaches of the estuary. The data were log transformed (CPUE +1) to reduce the influence of abundant species. Rare species (species occurring in less than three samples) were excluded from the analysis. The significance of the relationships between the species and environmental variables were analysed using the stepwise selection method and Monte Carlo permutation test (499 permutations) which is based on the sum of all canonical eigenvalues. The Spearman rank correlation coefficient was used to determine the significance of each environmental variable against the observed abundance of fish species (Zar 1984), as well determining the nature of the effect. The level of significance for these correlations was $P < 0.05$ after applying the Bonferroni correction.

RESULTS

Environmental variability

The results of PCA for the three physico-chemical variables revealed that the first two PC axes accounted for 88% of the variation between sites. The first axis was related to salinity and temperature (°C) and the second axis dissolved oxygen (mg l^{-1}) and temperature (°C). The PCA ordination showed that the environmental variables grouped sites 1-9 (upper reaches) together on the right hand side of the ordination. The sites that were recorded in the middle and lower reaches overlapped slightly, but the middle reaches (sites 10-18) were largely limited to the top left of the ordination. The lower reaches sites (19-24) were clustered mainly at the bottom left of the ordination (Figure 2.2).

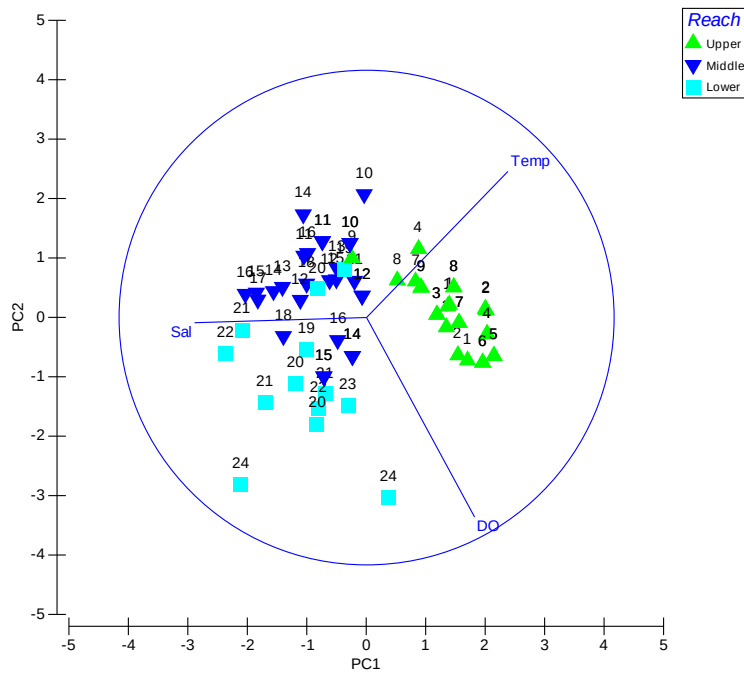


Figure 2.2. The results from the PCA ordination determining the upper, middle and lower reaches based on the temperature (Temp), salinity (Sal) and dissolved oxygen (DO) at each site.

Water temperature in summer ranged from 21.0 to 30.3 °C, with mean values showing a trend of increasing temperature from the lower (23.8 °C) to the upper reaches (25.1 °C). There were significant differences in the temperature values in the upper, middle and lower reaches ($H = 21.18$, $P < 0.01$) with the mean temperature values in each reach significantly different from each other (Mann Whitney U test $P < 0.05$). Salinity values ranged from 0.0 to 35.2, with mean values increasing from the upper (3.0) to the lower reaches (28.9), and significant differences recorded between the upper, middle and lower reaches ($F = 408.59$, $P < 0.001$). The mean salinity values in each reach were significantly different from each other (Tukey's test $P < 0.05$). Dissolved oxygen values ranged from 6.0 mg l⁻¹ to 17.1 mg l⁻¹, with mean values decreasing slightly from the upper reaches (7.4 mg l⁻¹) to the lower reaches (7.0 mg l⁻¹), and these three reaches were not significantly different from each other ($F = 0.24$, $P > 0.05$) (Figure 2.3).

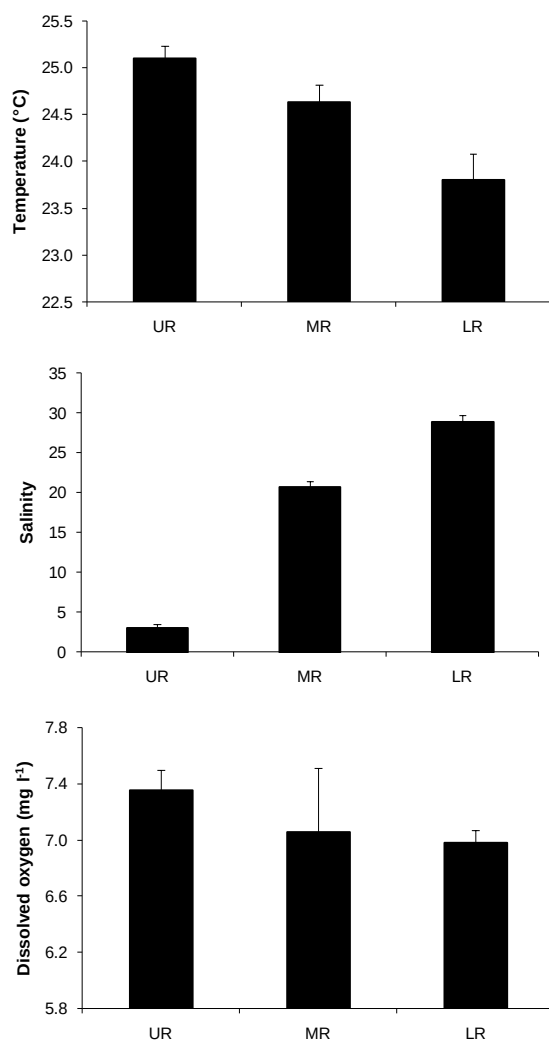


Figure 2.3. Mean ($\pm$ SE) summer environmental values of the three estuary reaches of the Breede Estuary from 2002 to 2012 UR-Upper reach, MR-Middle reach and LR-lower reach.

The combined monthly freshwater flow rate (Swellendam and Buffelsjags) over the period July 2001 to February 2012 showed obvious seasonal patterns, with higher flow generally recorded between July and November and lower flow rates recorded between December and June. Values above the 1970-2010 historical mean were considered high flow events and values below the historical mean were considered low flow events. The highest flow rates were recorded in September 2006 ($281.4 \text{ m}^3 \text{ s}^{-1}$) and October 2008 ($307.4 \text{ m}^3 \text{ s}^{-1}$), while the lowest flow rates were recorded in January 2006 ($0.5 \text{ m}^3 \text{ s}^{-1}$) and February 2012 ($0.6 \text{ m}^3 \text{ s}^{-1}$) (Figure 2.4).

The mean summer AVHRR SST values recorded between 1982 and 2012 showed no significant trend ($r^2 = 0.06$, $P > 0.05$). However, between 1995 and 2004 the highest significant

increase ($P < 0.05$) in SST values were observed. The lowest SST values occurred in 1989 (20.4 °C) and 1994 (20.0 °C), while peak temperature values occurred in 2003 (22.3 °C) and 2007 (22.2 °C) (Figure 2.5).

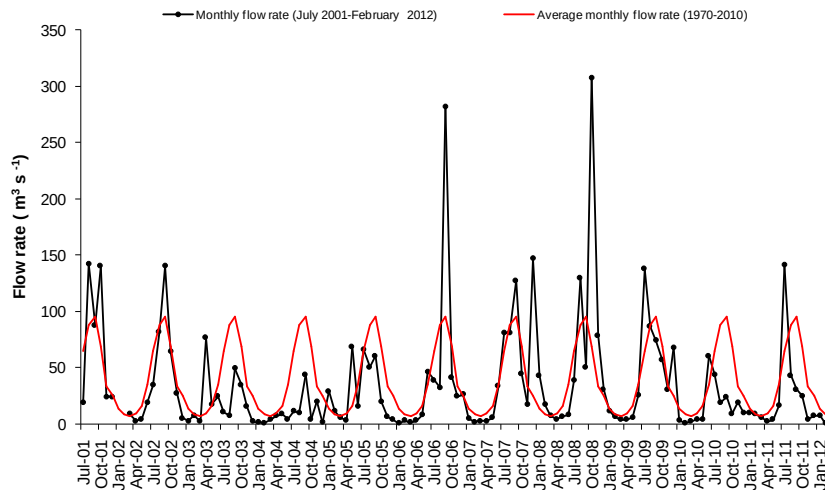


Figure 2.4. Temporal variation of the river flow rate values between July 2001 and February 2012 plotted against the average monthly river flow rate values for the period 1970-2010.

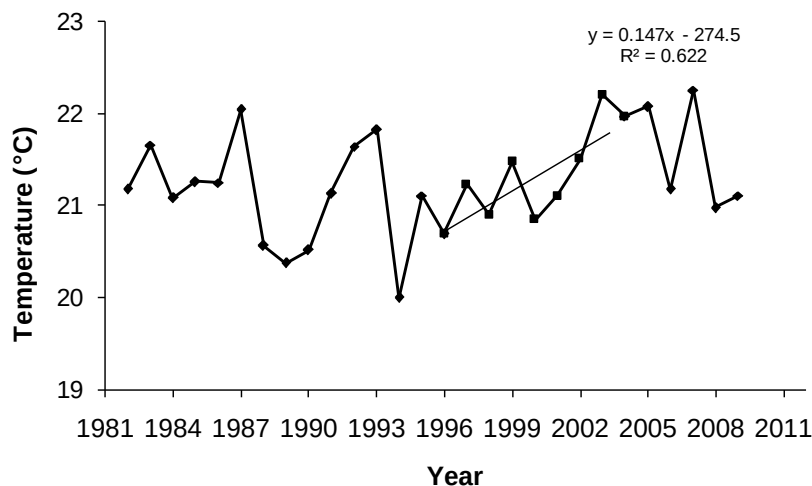


Figure 2.5. Mean summer sea temperature (SST) values over the period 1982-2012 (solid line = trend line).

Species composition

A total of 44 species were caught in both gear types (seine net and gill net) in summer during the period 2002-2012. The total number of species caught each year varied between 16

(2008) and 36 (2011). The most important families were Sparidae and Mugilidae, which were represented by eight (18.0%) and six (14%) species respectively. Of the 44 species caught, 22 (50%) are endemic to southern Africa (Table 2.2).

The number of estuary-associated marine species (category II) varied from eight (2008) to 16 (2004, 2005, 2011). These species collectively comprised 41% of all species, followed by marine stragglers (category III), which comprised 34% and varied between one (2002, 2009) and 10 (2011) species. They included *Myliobatus aquila* and *Pteromylaeus bovinus*, which were only recorded in gill nets. Estuarine resident species (category I) varied between five (2002, 2008, 2009) and nine (2004, 2005) and collectively comprised 21% of all species recorded. *Oreochromis mossambicus* and *Myxus capensis* were the only respective representatives of category IV and V species, with *M. capensis* a facultative catadromous species (Table 2.2).

Temperate species dominated the number of species recorded (in both the gill and seine nets) in terms of biogeographic origin and comprised between 66.0% (2011) and 88.0% (2008) of all the species recorded. Tropical species contributed between 0% (2002, 2008) and 26.0% (2011) of the species recorded. Widespread species varied from zero (0%) to three (18.0%) species caught and included *Gilchristella aestuaria*, *Mugil cephalus* and *Pomatomus saltatrix* (Table 2.2).

The six most abundant species that were consistently caught in the seine net were *Caffrogobius* spp. (6.8-58.4%), *Galeichthys feliceps* (0.6-62.5%), *Gilchristella aestuaria* (0.5-36.0%), *Liza richardsonii* (7.1-42.7%), *Liza dumerili* (0.7-6.9%) and *Solea turbynei* (1.3-12.9%). These species together comprised more than 76% of the seine net catch in all years (Figure 2.6). *Caffrogobius* spp. could not be identified to species level in the field and were assigned to the family *Caffrogobius*. Lamberth *et al.* (2008) and Harrison (1999) have noted that the two most common *Caffrogobius* species occurring in the Breede Estuary are *Caffrogobius gilchristi* and *Caffrogobius nudiceps*. Other dominant species included *Monodactylus falciformis*, which comprised between 0% and 11.7% of the catch and *Rhabdosargus holubi* which comprised between 0.4% and 10.8% of the catch (Figure 2.6). Category I and II species were the most abundant species recorded and contributed more than 96% of the catch in all years. Category I species contributed between 23.7% (2004) and 72.4% (2003) of the catch and dominated the catch in 2003, 2005 and 2011. Category II

species contributed between 27.1% (2003) and 76.2% (2004) and dominated the catch in the other six years (Figure 2.6).

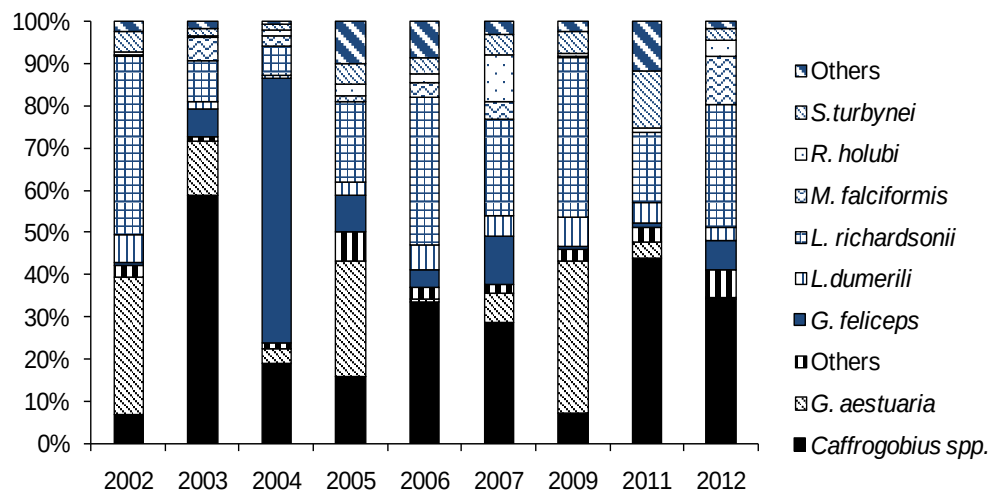


Figure 2.6. Annual seine-net catch composition (%) of the dominant species recorded in the Breede Estuary between 2002 and 2012 (black represents estuary resident species and blue represents marine species).

Table 2.2. Fish species recorded in the seine and gill nets in the Breede Estuary in the summer months between 2002 and 2012 (1 represents species present in the catch, 0 represents species absent from the catch).

Family	Species	Category	Year									
			2002	2003	2004	2005	2006	2007	2008	2009	2011	2012
Ariidae	<i>Galeichthys feliceps</i> *	I Ib	1	1	1	1	1	1	1	1	1	1
Atherinidae	<i>Atherina breviceps</i> *	Ib	0	1	1	1	0	0	1	0	1	0
Aulostomidae	<i>Aulostomus chinensis</i>	III	0	0	0	0	1	0	0	0	1	0
Blenniidae	<i>Omobranchus woodi</i> *	Ia	0	1	1	1	1	0	0	0	0	1
Carangidae	<i>Gnathanodon speciosus</i>	III	0	1	0	0	0	0	0	0	0	0
Carangidae	<i>Lichia amia</i>	IIa	1	1	1	1	1	1	1	1	1	1
Chaetodontidae	<i>Chaetodon marleyi</i> *	III	0	0	0	0	0	0	0	0	1	0
Cichlidae	<i>Oreochromis mossambicus</i>	IV	0	0	0	1	0	1	0	0	1	0
Clinidae	<i>Clinus superciliosus</i> *	Ib	0	1	1	1	1	1	0	0	1	0
Clupeidae	<i>Gilchristella aestuaria</i> *	Ia	1	1	1	1	1	1	0	1	1	1
Diodontidae	<i>Diodon holocanthus</i>	III	0	0	0	0	0	0	0	0	1	1
Gobiidae	<i>Caffrogobius</i> spp.*	Ib	1	1	1	1	1	1	1	1	1	1
Gobiidae	<i>Psammogobius knysnaensis</i> *	Ia/Ib	1	1	1	1	1	1	1	1	1	1
Gobiidae	<i>Redigobius dewaali</i> *	Ib	0	0	1	1	1	1	0	0	1	1
Haemulidae	<i>Pomadasys commersonnii</i>	IIa	0	0	1	1	0	0	0	1	1	0
Haemulidae	<i>Pomadasys olivaceus</i>	III	0	1	1	1	0	1	0	0	0	0
Hemiramphidae	<i>Hyporhamphus capensis</i> *	Ia	1	0	1	1	1	1	1	1	1	1
Monodactylidae	<i>Monodactylus falciformis</i>	IIa	1	1	1	1	1	1	0	1	0	1
Mugilidae	<i>Liza dumerili</i>	IIb	1	1	1	1	1	1	1	1	1	1
Mugilidae	<i>Liza macrolepis</i>	IIa	0	0	1	0	0	0	0	0	0	0
Mugilidae	<i>Liza richardsonii</i> *	IIc	1	1	1	1	1	1	1	1	1	1
Mugilidae	<i>Liza tricuspidens</i> *	IIb	0	1	0	1	1	1	0	1	1	0
Mugilidae	<i>Mugil cephalus</i>	IIa	1	1	1	1	0	1	0	1	1	0
Mugilidae	<i>Myxus capensis</i> *	Vb	1	1	1	1	1	1	1	1	1	1
Mullidae	<i>Parupeneus rubescens</i>	III	0	1	0	0	1	0	0	0	1	0
Myliobatidae	<i>Myliobatus aquila</i>	III	0	0	0	0	0	1	0	0	0	0
Myliobatidae	<i>Pteromylaeus bovinus</i>	III	0	0	0	0	0	0	0	1	1	0

Notocheiridae	<i>Iso natelensis</i>	III	0	0	0	1	0	0	0	0	0	0
Pomatomidae	<i>Pomatomus saltatrix</i>	IIc	1	1	1	1	0	1	0	1	1	1
Sciaenidae	<i>Argyrosomus japonicus</i>	IIa	0	1	1	0	0	0	0	1	1	0
Soleidae	<i>Heteromycteris capensis*</i>	IIb	0	1	1	1	1	1	1	0	1	0
Soleidae	<i>Solea turbynei*</i>	IIb	1	1	1	1	1	1	1	1	1	1
Sparidae	<i>Diplodus hottentotus*</i>	III	0	0	0	0	1	1	0	0	1	0
Sparidae	<i>Diplodus capensis*</i>	IIc	1	1	1	1	1	1	1	1	1	0
Sparidae	<i>Lithognathus lithognathus*</i>	IIa	1	0	1	1	1	1	1	1	1	1
Sparidae	<i>Lithognathus mormyrus</i>	III	0	0	0	1	0	1	0	0	0	0
Sparidae	<i>Rhabdosargus globiceps*</i>	IIc	0	1	1	1	1	1	0	0	1	0
Sparidae	<i>Rhabdosargus holubi*</i>	IIa	1	1	1	1	1	1	0	1	1	1
Sparidae	<i>Sarpa salpa</i>	IIc	0	0	0	1	0	0	0	0	1	0
Sparidae	<i>SpondylIOSoma emarginatum*</i>	III	0	1	0	0	0	0	1	0	1	0
Syngnathidae	<i>Syngnathus temminckii*</i>	Ib	1	1	1	1	1	1	1	1	1	1
Tetraodontidae	<i>Amblyrhynchotes honckenii</i>	III	1	1	1	1	1	1	1	1	1	0
Tetraodontidae	<i>Arothron immaculatus</i>	III	0	1	0	0	0	1	0	0	1	1
Tetraodontidae	<i>Pelagocephalus marki</i>	III	0	0	0	0	0	0	0	0	1	0

*Endemic to southern Africa

Community change

Kendall's coefficient of concordance ($W = 0.74$) indicated that there was little change in the rank order of abundance of individual species within the fish assemblage between years (χ^2 , $r = 265$, $P < 0.05$). The Spearman rank correlations showed a similar result with no significant differences between years, which indicates a similar rank order of abundance for the different species each year.

Total CPUE varied considerably between years and ranged from 89 fish haul⁻¹ in 2009 to 367 fish haul⁻¹ in 2004 (Figure 2.7) which mirrors the considerable annual variation in the CPUE of individual species (Figure 2.8). CPUE for *Gilchristella aestuaria* peaked in 2003 (45 fish haul⁻¹) and 2005 (49 fish haul⁻¹). CPUE for *Caffrogobius* spp. peaked in 2003 and 2011, with 203 fish haul⁻¹ and 81 fish haul⁻¹ recorded respectively, while peak CPUE for *Galeichthys feliceps* occurred in 2004, with 229 fish haul⁻¹. *Liza dumerili* recorded peak CPUE in 2006 (11 fish haul⁻¹) and 2011 (9 fish haul⁻¹), while CPUE for *Liza richardsonii* peaked in 2002 (42 fish haul⁻¹) and 2006 (66 fish haul⁻¹). CPUE for *Solea turbynei* peaked in 2011 with 25 fish haul⁻¹ (Figure 2.8).

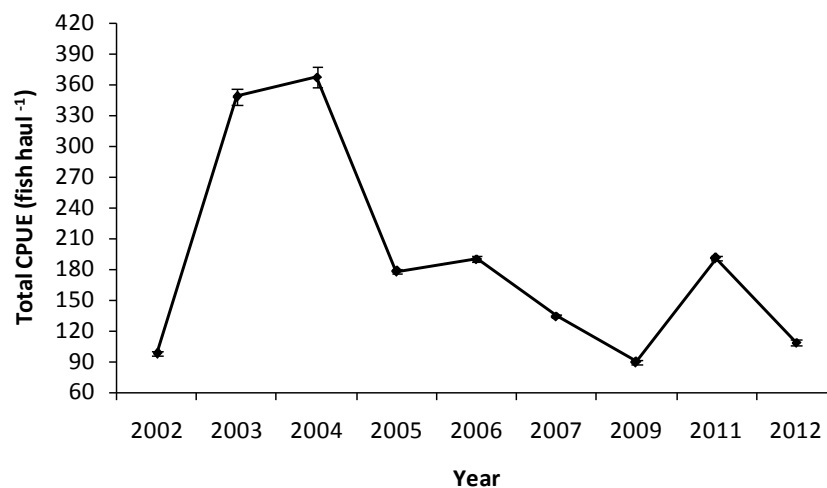


Figure 2.7. The total CPUE (fish haul⁻¹) from the summer seine net samples taken from February 2002-February 2012 in the Breede Estuary.

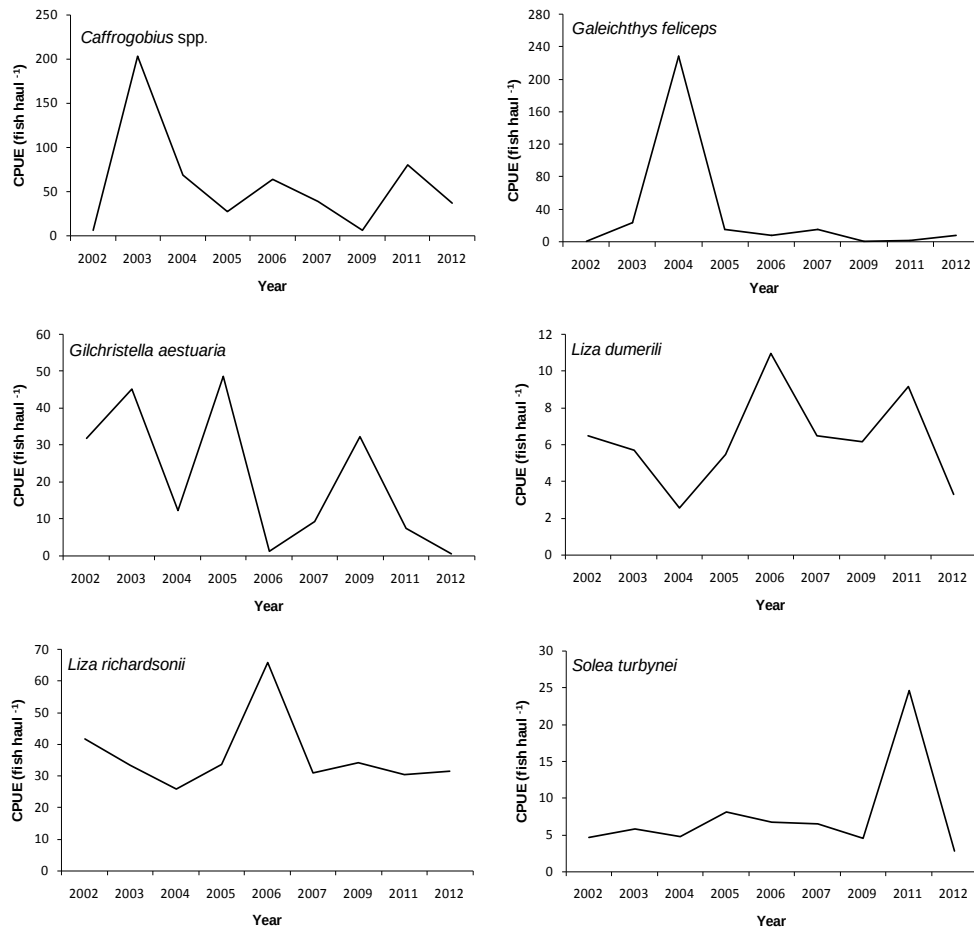


Figure 2.8. Annual CPUE (fish haul⁻¹) of the six most abundant species caught in the Breede Estuary from 2002-2012.

Taxonomic distinctness (Δ^*) ($r^2 = 0.258$, $P = 0.17$) and average taxonomic distinctness based on presence/absence data (Δ^+) ($r^2 = 0.183$, $P = 0.25$) showed no significant ($P < 0.05$) linear trends over time (Figure 2.9).

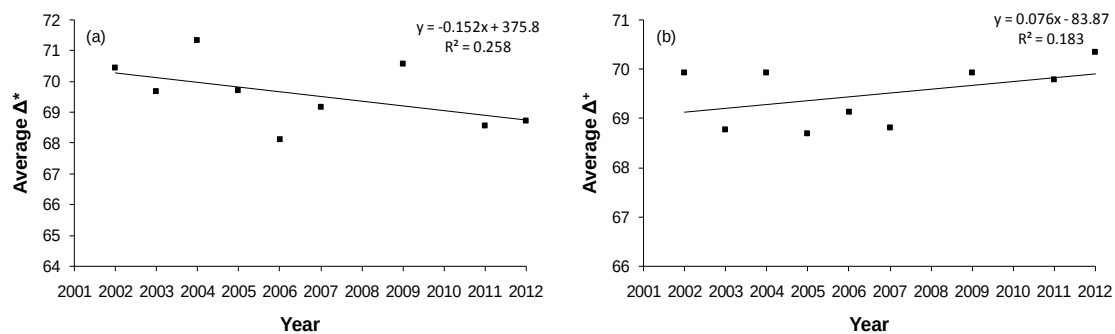


Figure 2.9. Time series of two multivariate diversity measures (a) taxonomic distinctness (Δ^*) and (b) average taxonomic distinctness based on presence/absence data (Δ^+) for the summer Breede Estuary fish assemblage.

Spatial and temporal trends

The cluster dendrogram (Figure 2.10) and MDS ordination plot (Figure 2.11) separated the estuarine fish assemblage into three distinct estuary reaches (upper, middle and lower), regardless of year. The two-way crossed ANOSIM revealed significant differences among reaches (averaged across years) and between years (averaged across reaches). The fish communities within each estuary reach were similar each year (Global $R > 0.7$; $P < 0.01$). Highly significant dissimilarities were evident between the three estuary reaches, with the most dissimilarity occurring between the upper and lower reaches (Table 2.3).

Table 2.3. Results of the Global R tests on the effect of estuary reach and year, and the pairwise tests on the effect of reach, using a two-way crossed ANOSIM.

	Main effect	R-statistic
Global R		
	Estuary reach	0.76**
	Year	0.32*
Pairwise tests		
	Upper vs. Middle	0.61**
	Upper vs. Lower	0.97**
	Middle vs. Lower	0.63**

* $P < 0.05$, ** $P < 0.01$

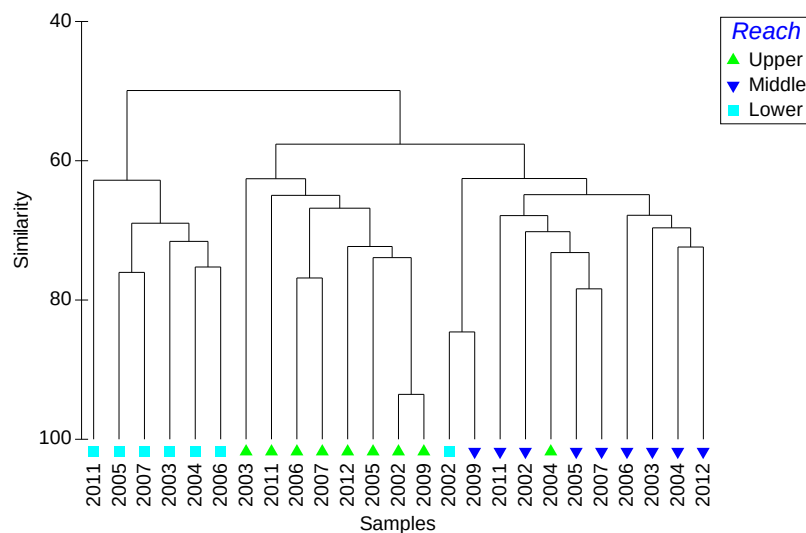


Figure 2.10. Dendrogram from the cluster analysis showing the annual fish community structure in the upper, middle and lower reaches.

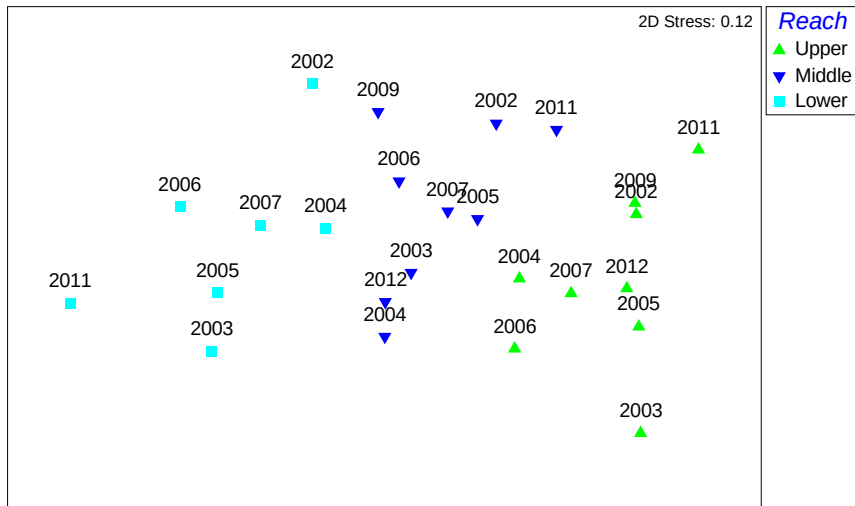
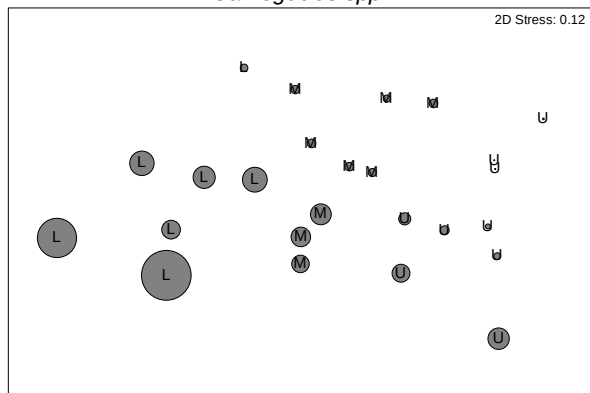


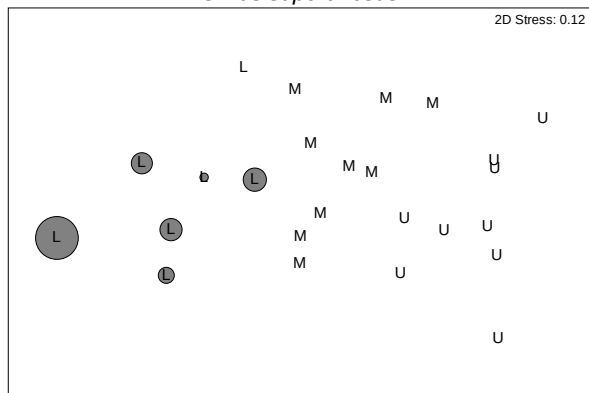
Figure 2.11. Multidimensional scaling (MDS) plot of the summer Breede Estuary fish assemblages in the three estuarine reaches for the period 2002-2012.

SIMPER analysis revealed that nine species were responsible for the differences in the fish assemblages between reaches: *Caffrogobius* spp., *Diplodus capensis*, *Galeichthys feliceps*, *Gilchristella aestuaria*, *Heteromycteris capensis*, *Monodactylus falciformis*, *Clinus superciliosus*, *Rhabdosargus holubi* and *Syngnathus temminckii*. Of these species *Caffrogobius* spp., *C. superciliosus*, *H. capensis*, *D. capensis* and *S. temminckii* were more abundant in the lower reaches, while *G. feliceps* and *R. holubi* were more abundant in the lower and middle reaches. *Monodactylus falciformis* showed a preference for the middle and upper reaches and were most abundant in the upper reaches. *Gilchristella aestuaria* was abundant in the upper, middle and lower reaches (Figure 2.12).

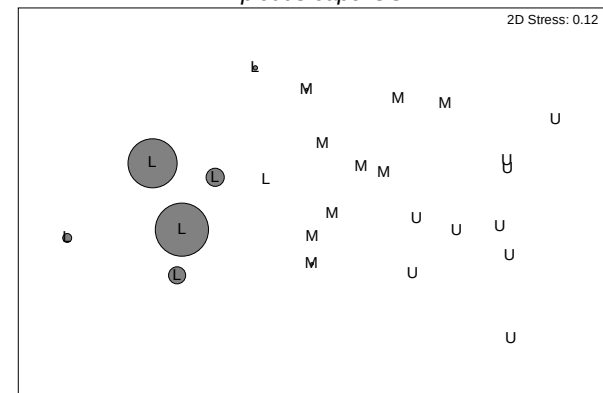
Caffrogobius spp.



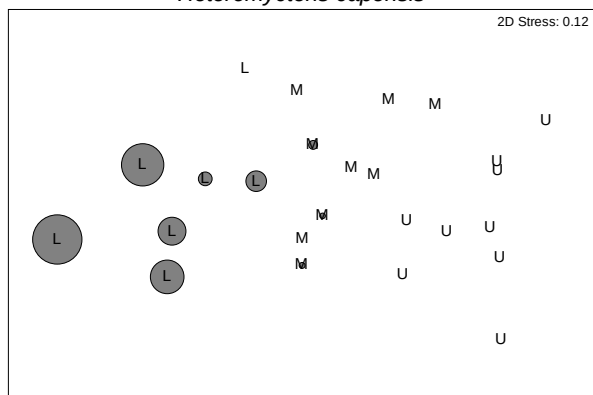
Clinus superciliosus



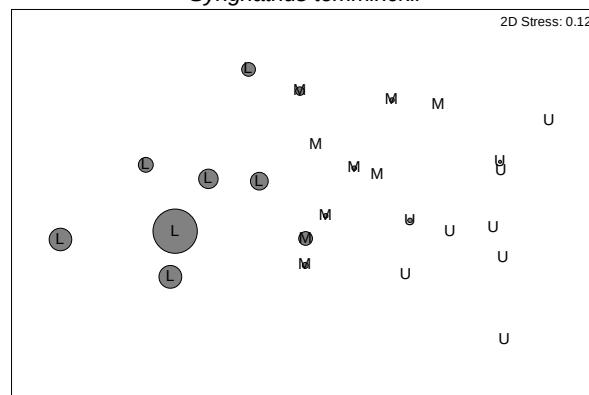
Diplodus capensis



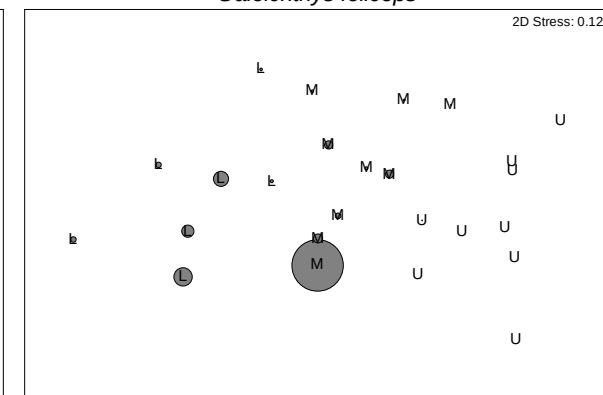
Heteromycteris capensis



Syngnathus temminckii



Galeichthys feliceps



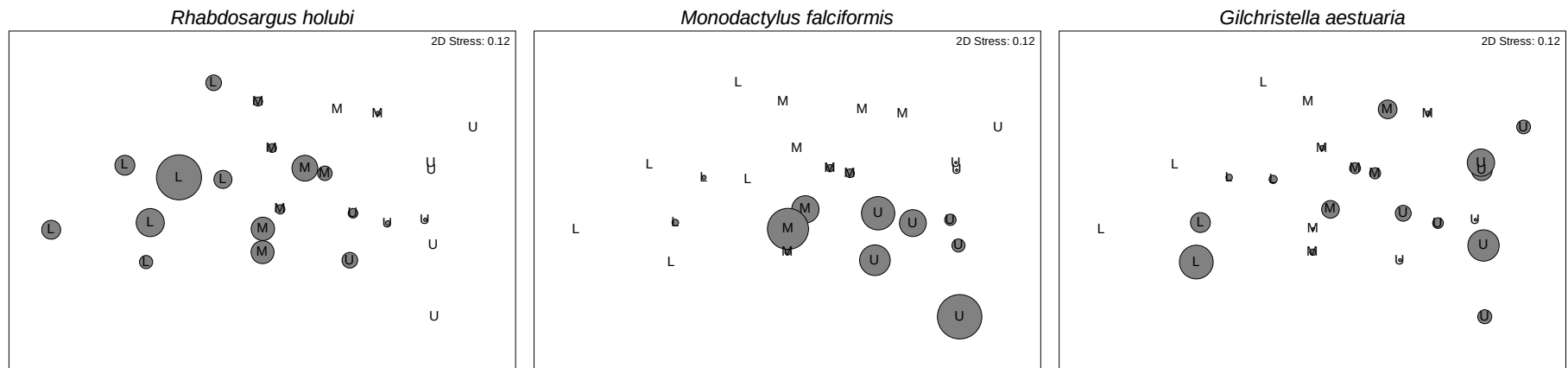


Figure 2.12. MDS ordination plots with superimposed relative abundances of nine discriminating species as determined by SIMPER analysis along the upper, middle and lower estuary reaches. Sizes of bubbles indicate relative abundance for that particular species, but are not comparable between species. U-upper reaches, M-Middle reaches, L-lower reaches.

Assemblage structure and reach-scale environmental variables

The first two axes of the canonical correspondence analysis (CCA) explained 27% of the species variation and 77% of the species-environment correlation (Table 2.4). The Monte-Carlo test indicated that although there was a significant ($P < 0.001$) relationship between environmental factors and species abundance, only salinity was considered significant ($P < 0.05$). The length of the vector for a particular variable indicates its importance (Figure 2.13) and the closer a particular species occurs to a vector the stronger the relationship with that variable. The first axis explained most of the variation (59%) and was correlated to salinity (0.84) and temperature (0.39) with high salinity and low temperature values on the left hand side and low salinity and high temperature values on the right hand side. The second axis was correlated with temperature (0.42) and average high “winter” flow rate (0.46), with high flow rates at the top of the graph and high temperatures on the bottom half of the graph (Figure 2.13).

Table 2.4. Results of the CCA used to determine the relationship between the relative abundance (CPUE) of fish species and environmental variables in the Breede Estuary.

Variable	Axis			
	1	2	3	4
Eigenvalue	0.17	0.05	0.03	0.03
Cumulative percentage of species variation	20.4	26.7	30.6	34.0
Cumulative percentage of species-environment correlation	58.5	76.7	87.8	97.6
Sum of all unconstrained axes				0.83
Sum of all canonical axes				0.30

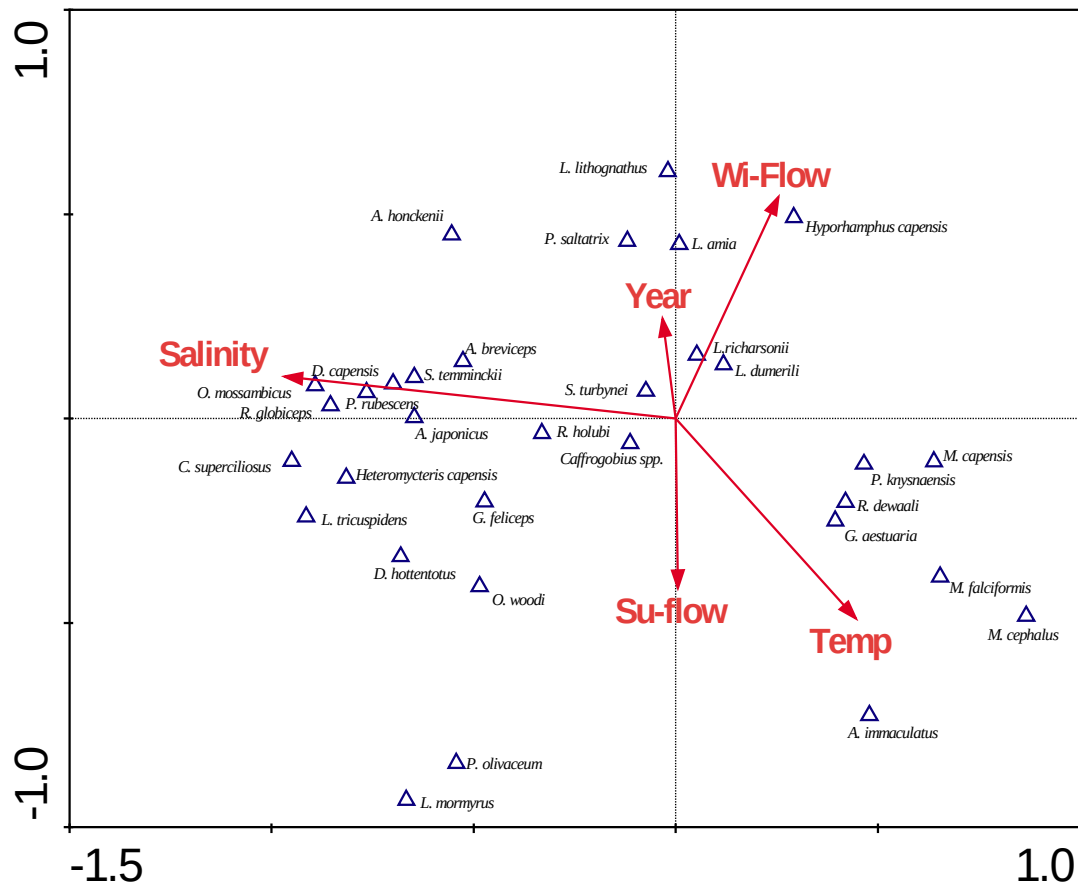


Figure 2.13. Canonical correspondence analysis (CCA) ordination relating estuary-associated fish species composition to environmental variables.

A significant and positive correlation was recorded between abundance and salinity for *Caffrogobius* spp., *Diplodus capensis*, *Galeichthys feliceps*, *Liza richardsonii*, *Rhabdosargus holubi*, *Solea turbynei* and *Syngnathus temminckii*, which are more abundant in the lower reaches (Figure 2.12). Higher abundance occurred at lower salinities for *Psammogobius knysnaensis* (Figure 2.13, Table 2.5). Temperature was an important variable for *Gilchristella aestuaria*, as a significant and positive correlation between abundance and estuarine temperatures were recorded. The abundance of *Monodactylus falciformis*, which is more abundant in the middle and upper reaches (Figure 2.12), was significantly and positively correlated with estuarine temperature and negatively correlated with salinity (Table 2.5).

Table 2.5. Spearman rank correlation between the twelve most abundant species and environmental variables.

Species	Temperature	Salinity	Winter flow rate
<i>Caffrogobius</i> spp.	ns	<0.001 ⁺⁺⁺	ns
<i>Diplodus capensis</i>	ns	<0.001 ⁺⁺⁺	ns
<i>Galeichthys feliceps</i>	ns	<0.001 ⁺⁺⁺	ns
<i>Gilchristella aestuaria</i>	<0.05 ⁺	ns	ns
<i>Hyporhamphus capensis</i>	ns	ns	ns
<i>Liza richardsonii</i>	ns	<0.05 ⁺	ns
<i>Liza dumerili</i>	ns	ns	ns
<i>Monodactylus falciformis</i>	<0.01 ⁺⁺	<0.05 ⁻	ns
<i>Psammogobius knysnaensis</i>	ns	<0.05 ⁻	ns
<i>Rhabdosargus holubi</i>	ns	<0.001 ⁺⁺⁺	ns
<i>Solea turbynei</i>	ns	<0.05 ⁺	ns
<i>Syngnathus temminckii</i>	ns	<0.001 ⁺⁺⁺	ns

ns= not significant ($P > 0.05$); + or - = slightly significant ($P < 0.05$); ++ or -- = significant ($P < 0.01$); +++ or --- = highly significant ($P < 0.001$)

Assemblage structure and estuary-scale environmental variables

CANOCO analyses indicated that flow rates in the months preceding (winter flow) and during sampling (summer flow) were not significant for any species (Table 2.5). Similarly, when freshwater flow was plotted against the CPUE of individual species, there was no obvious relationship between flow rates and peak abundance, although severe flood events did result in the decreased abundance of many estuarine resident species (Figure 2.14). Peak catches of *Caffrogobius* spp. were recorded in 2003 (203 fish haul⁻¹) after freshwater flow reached 140 m³ s⁻¹ in September 2002. A similar flow rate was recorded in October 2001 (141 m³ s⁻¹), but the CPUE in 2002 was the second lowest recorded during the study period for this species (7 fish haul⁻¹). CPUE for *Gilchristella aestuaria* peaked following high flow (>100 m³ s⁻¹) and low flow conditions (<50 m³ s⁻¹). Peak CPUE values were recorded for *Psammogobius knysnaensis* in 2005 and 2011 following low flow conditions (<45 m³ s⁻¹), while *Syngnathus temminckii* recorded peak CPUE in 2005, which was preceded by flow conditions that did not exceed 45 m³ s⁻¹. All estuarine resident species, with the exception of *G. aestuaria*, recorded a decrease in CPUE after flow exceeded 250 m³ s⁻¹ (Figure 2.14).

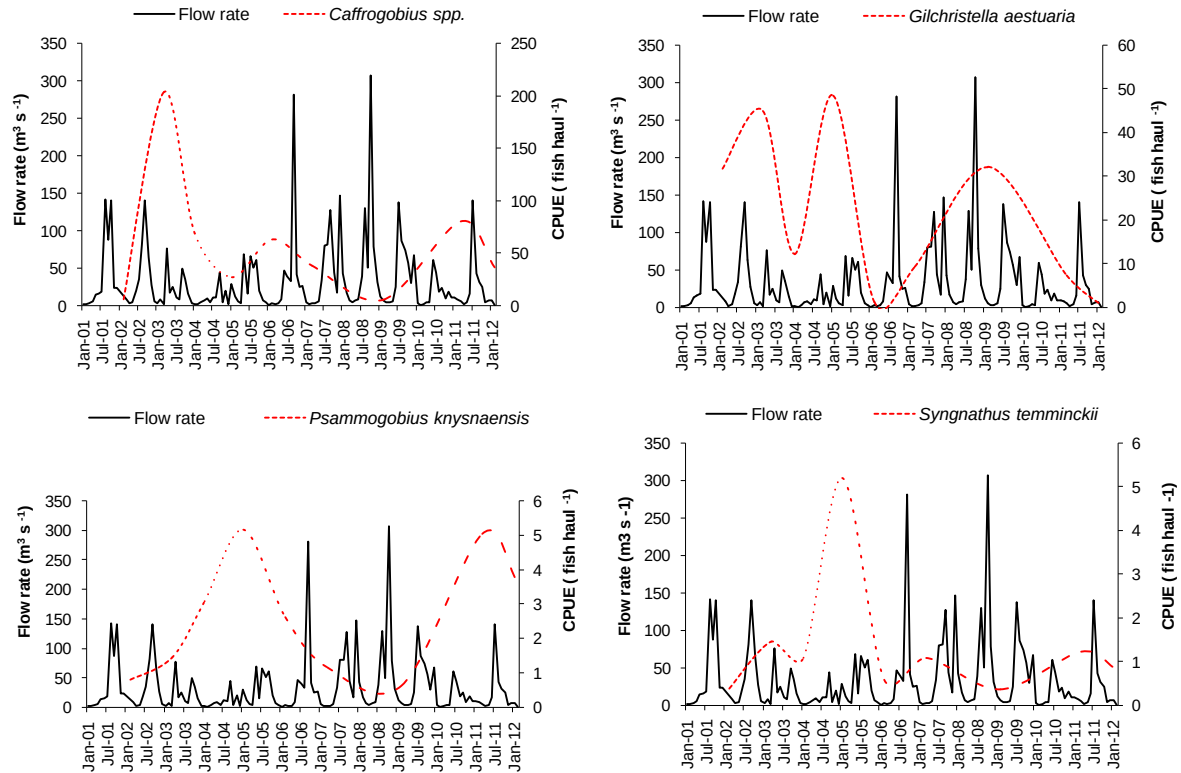


Figure 2.14. The effect of freshwater flow rates on the relative abundance (CPUE) of the four most abundant estuarine resident species.

Temporal peaks in the abundance of estuary-associated marine species differed between species. The highest catches of *Galeichthys feliceps* (2004), *Liza dumerili* (2006, 2011), *Liza richardsonii* (2006) and *Solea turbynei* (2011) occurred following relatively low flow conditions ($<50 \text{ m}^3 \text{ s}^{-1}$) (Figure 2.15).

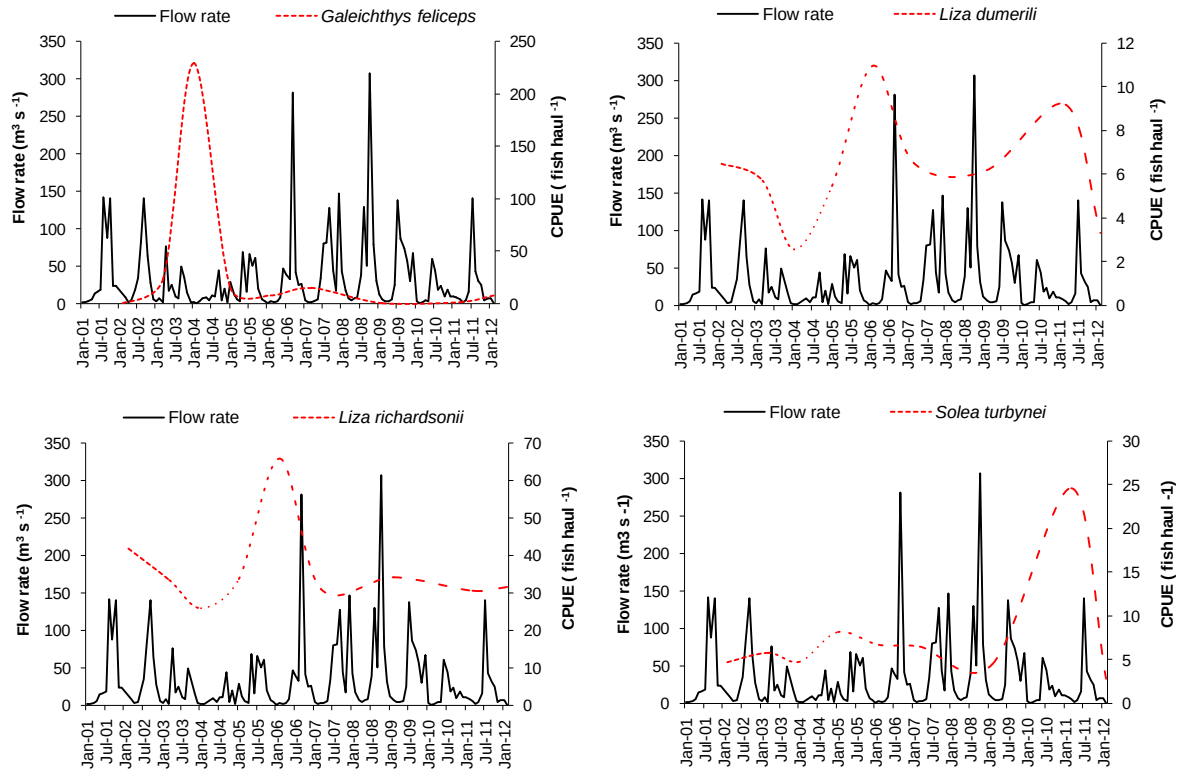


Figure 2.15. The effect of freshwater flow rates on the relative abundance (CPUE) of the four most abundant estuary-associated marine species.

The number of species and total abundance (CPUE) of both tropical (Group 1 and Group 2) and temperate (Group 4 and 5) species recorded each year showed no obvious trend relating to sea surface temperature (SST) (Figure 2.16, Figure 2.17, Figure 2.18). Both tropical and temperate species, however, followed similar patterns with regard to sea surface temperature and total abundance (Figure 2.17, Figure 2.18). However, the abundance of tropical species was highest in 2003 ($20 \text{ fish haul}^{-1}$), which coincides with the warmest SST during the study period (Figure 2.17). Temperate species CPUE ranged from $43 \text{ fish haul}^{-1}$ in 2009 to $331 \text{ fish haul}^{-1}$ in 2004 (Figure 2.18). The abundance of temperate species was lowest in 2009 ($43 \text{ fish haul}^{-1}$), which coincided with the coldest summer temperature during sampling, but no other obvious trends with SST and temperate species were recorded (Figure 2.18).

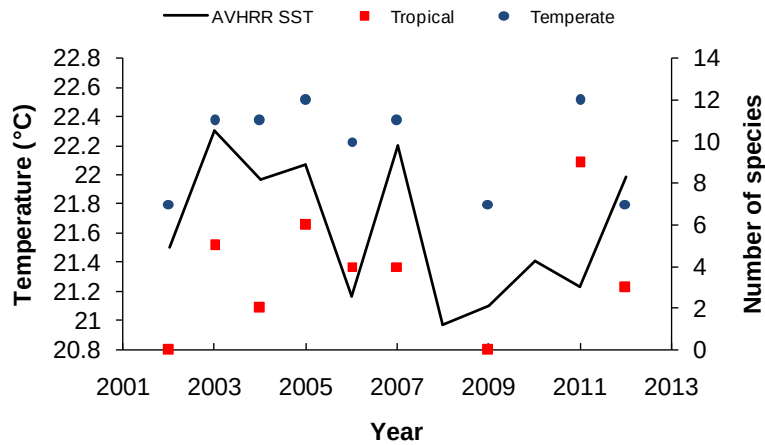


Figure 2.16. The relationship between the total number of tropical (Groups 1 and 2) and temperate (Groups 4 and 5) species captured in the Breede Estuary between 2002 and 2012 and AVHRR (advanced very high resolution radiometer) SST (sea surface temperature).

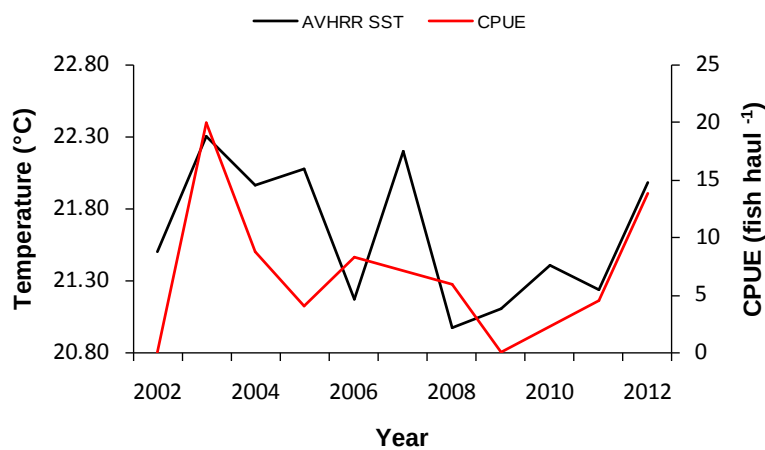


Figure 2.17. The relationship between the total CPUE of tropical species caught in the Breede Estuary and AVHRR (advanced very high resolution radiometer) SST (sea surface temperature).

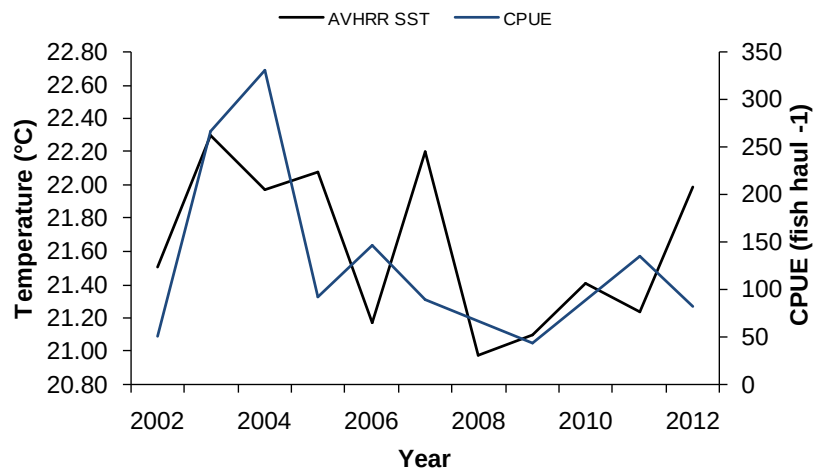


Figure 2.18. The relationship between the total CPUE of temperate species caught in the Breede Estuary and AVHRR (advanced very high resolution radiometer) SST (sea surface temperature).

DISCUSSION

The summer fish assemblage showed year-to-year variation in the annual CPUE, which reflected the considerable variation in the individual species CPUE. Since marine species were the most dominant species in the Breede Estuary, the abundance of juvenile marine taxa are likely to reflect the variability in annual recruitment success (Whitfield 1994b), and may account for the inconsistent annual CPUE in this system. Variable recruitment and migration of species has been shown to impact estuarine fish populations (Costa *et al.* 2002), with recruitment strengths for species changing significantly over time (e.g. Potter *et al.* 2001, Martinho *et al.* 2008, Dolbeth *et al.* 2010).

Variable recruitment caused year-to-year changes in the catch contribution of the most abundant species which, in turn, affects the relative contributions of estuary resident, estuary-associated marine and marine straggler species to the overall fish composition. However, the majority of the annual Breede Estuary fish assemblage comprised six species that were consistently caught throughout the study period, and are also common in permanently open warm-temperate estuaries (Harrison 2005), including the Breede system (Harrison 1999, Lamberth *et al.* 2008). Estuaries are characteristically dominated by a few species (e.g. Jackson and Jones 1999, Hagan and Able 2003, Clark *et al.* 2009, Sheppard *et al.* 2011, Nyitrai *et al.* 2012).

Assemblage stability indicates resilience to disturbances through persistence of species and relationships within a system (Holling 1973, Dugan and Livingston 1982). This was measured using Kendall's coefficient of concordance (W) and taxonomic distinctness indices (Δ^* and Δ^+). Kendall's coefficient of concordance (W) compared the ranking among species per year and a consistent rank abundance amongst years indicates a natural persistence of species in a system (Dugan and Livingston 1982). The coefficient of concordance in the Breede Estuary (0.74) indicates a 74% concordance of the rankings of the species caught each year which implies stability and a persistence of species in the main Breede Estuary fish assemblage structure over the study period irrespective of the year-to-year variation in relative abundance and species richness. Dugan and Livingston (1982) recorded a coefficient of concordance of 0.65 among the ranking of the macroinvertebrate species in the Econfinia Estuary in Apalachee Bay (United States of America) which indicated resilience to natural disturbances. Similarly, stability in the main assemblage structure of the temporarily open/closed East Kleinemonde Estuary (South Africa) (James *et al.* 2008b) and Cedar Fork Creek, United States of America (Meffe and Berra 1988) occurred with coefficient of concordance values of 0.57. Taxonomic distinctness indices can determine different aspects of biodiversity, which can be used to interpret assemblage structure (Rogers *et al.* 1999). No significant temporal changes occurred in the taxonomic distinctness indices during this study. Systems with a wider taxonomic range are likely to maintain assemblage stability during natural or anthropogenic disturbances (Tilman 1996).

The fish communities in the lower, middle and upper reaches showed distinct fish communities irrespective of time, which suggests a fairly stable spatial trend within the Breede Estuary fish assemblage structure. Similarly, tropical estuaries on the north-eastern Australian coastline have estuary-specific faunal compositions that are repeated each year (Sheaves 2006). The nine species that contributed most of the dissimilarity between the reaches consistently showed zonal preferences. *Caffrogobius* spp., *Clinus superciliosus*, *Diplodus capensis*, *Heteromycteris capensis* and *Syngnathus temminckii* were primarily recorded in the lower reaches. *Caffrogobius gilchristi* and *Caffrogobius nudiceps*, the two most abundant species of *Caffrogobius* in the Breede Estuary (Lamberth *et al.* 2008) prefer the lower reaches in other South African estuaries (e.g. Whitfield 1998, Whitfield and Paterson 2003, Booth 2007, Clark *et al.* 2009, Kruger and Strydom 2010). The preference of *S. temminckii* and *C. superciliosus* for the lower reaches has also been recorded in other estuarine studies (Whitfield and Paterson 2003, Lamberth *et al.* 2008, Vorwerk *et al.* 2008,

Clark *et al.* 2009). These species are strongly associated with eelgrass beds, *Zostera capensis* (e.g. Beckley 1983, Whitfield 1998, Whitfield 1999, Paterson and Whitfield 2000, Booth 2007) which are found in the lower reaches of the Breede Estuary (Lamberth *et al.* 2008). The distribution of *H. capensis* in estuaries is attributed to sediment characteristics and this species prefers the sandier sediment type, which characterises the lower reaches of estuaries (Whitfield 1998, Richardson *et al.* 2006, Bailey and James 2013). Sediment preference for estuarine demersal fish species may be determined by occurrence of prey items, foraging success, burying ability and predator avoidance (Richardson *et al.* 2006). This may explain the absence of *H. capensis* from the fluvial sand habitat in the upper reaches of the Breede Estuary. *Diplodus capensis* also prefers the lower reaches of other South African estuaries which include the Sundays (Beckley 1984), Swartvlei (Whitfield 1988) and Kariega (Ter Morshuizen and Whitfield 1994, Whitfield and Paterson 2003) estuaries which is attributed to the higher salinities in this reach (Cowley and Whitfield 2001).

Galeichthys feliceps and *Rhabdosargus holubi* were most abundant in the lower and middle reaches and these distribution patterns have been noted in other studies for *G. feliceps* (Beckley 1984, Whitfield 1998) and *R. holubi* (Whitfield and Paterson 2003). However, *R. holubi* has been recorded being equally abundant in the lower, middle and upper reaches in some Eastern Cape estuaries (Beckley 1984, Hanekom and Baird 1984, Vorwerk *et al.* 2001). *Rhabdosargus holubi* is associated with submerged macrophytes (e.g. Beckley 1983, Hanekom and Baird 1984, Whitfield 1988, Sheppard 2010) and consequently its distribution in estuaries is limited to the distribution of submerged macrophytes (Cowley and Whitfield 2001). The peak abundance of *Monodactylus falciformis* in the upper reaches in this study corresponds with findings from the Kariega and East Kleinemonde estuaries (South Africa), where a higher abundance of *M. falciformis* was also found in the upper reaches (e.g. Ter Morshuizen and Whitfield 1994, Cowley and Whitfield 2001). *Monodactylus falciformis* uses the upper reaches of estuaries as nursery areas (Beckley 1984, Hanekom and Baird 1984). *Gilchristella aestuaria* was recorded throughout the Breede Estuary as found in the Berg (Clark *et al.* 2009), East Kleinemonde (Cowley and Whitfield 2001), Kariega (Whitfield and Paterson 2003) and Sundays (Beckley 1984) estuaries.

The CCA ordination plot explained 27% of the species variation and 77% of the species-environmental variation. A low percentage of variation in ordination of the CCA is still considered to be informative (Ter Braak 1986), as 100% of the variance cannot be explained

due to part of the variance resulting from ‘noise’ in the data (Gauch 1982). Globally, a low percentage of variation has been recorded in the Canche Estuary, France (Selleslagh and Amara 2008), Elbe (Germany) and Tagus (Portugal) estuaries (Thiel *et al.* 2003), Mad Island Marsh Estuary, United States of America (Akin *et al.* 2003) and Patos Lagoon Estuary in Brazil (Garcia *et al.* 2003b). The high proportion of unexplained variation in the fish assemblage may be the result of other abiotic and biotic factors (Akin *et al.* 2005, Selleslagh and Amara 2008), such as prey abundance and distribution, predator avoidance, feeding behaviour and competition (Lankford and Targett 1994, Martinho and Able 2003, Islam *et al.* 2006). The high percentage obtained for the species-environmental variation shows that the environmental variables tested in the CCA ordination are important in structuring the fish assemblage in the Breede Estuary.

Salinity strongly influenced the spatial structure of the Breede Estuary fish assemblage and the remaining environmental variables were not considered important. Inter-annual variation did not influence the fish assemblage structure over the study period, which suggests that the assemblage structure in each reach was fairly stable over the 10 year period (2002-2012) and is only influenced by local environmental parameters (salinity). This contradicts the findings of Thiel *et al.* (2003) and Desmond *et al.* (2002), who noted that inter-annual variability was important in structuring the fish assemblage in the Elbe Estuary (Germany) and the invertebrate assemblage in three southern Californian estuaries respectively. Salinity has been shown to structure the fish communities of estuaries both locally and globally. Locally, abundance of 15 species has been positively correlated with salinity, while several tropical species showed a negative correlation with salinity in South African estuaries (Harrison and Whitfield 2006). Globally, Allen (1982) found salinity to be the best predictor of the littoral fish abundance in the upper Newport Bay in California. In the Humber, England, (Marshall and Elliot 1998) and the Canche, France, (Selleslagh and Amara 2008) estuaries, salinity influenced species richness.

Salinity increased from the upper to the lower reaches of the Breede Estuary and the CCA ordination showed that most species recorded over a 10 year period were associated with higher salinities when compared to the less saline waters. Similarly, Marais (1988) recorded the lowest species diversity in the upper reaches of several South African estuaries, and Beckley (1984) found that the number of species decreased from the lower to the upper reaches in the Sundays Estuary (South Africa). Salinity was also an important factor in the

spatial distribution of species within the Elbe (Thiel *et al.* 1995), Mngazi, South Africa (Patrick *et al.* 2007), Mullica River-Great Bay, United States of America (Martinho and Able 2003) and Río de la Plata, Argentina-Uruguay (Jaureguizar *et al.* 2004) estuaries, amongst others.

The mean monthly winter freshwater flow preceding summer sampling showed no observable correlation with abundance of estuary resident and estuary-associated marine species, except for the severe floods before 2007 and 2009. Wingate and Secor (2008) found the mean winter flow affected the summer-fall fish assemblage structure in the Patuxent River Estuary (United States of America). Freshwater flow influences environmental variables which affects the distribution and abundance of fish species within estuaries (e.g. Cyrus and Blaber 1992, Drinkwater and Frank 1994, Marshall and Elliot 1998, Whitfield 1999, Childs *et al.* 2008). Consequently, flooding will have long-term impacts on the abundance and distribution of many estuary-resident and estuary-associated marine species (Martin *et al.* 1992). Habitat diversity can reduce the negative effects of a flood on fish species (Marais 1982, Pearsons *et al.* 1992). The Great Fish and Thukela estuaries (South Africa) contain limited habitat diversity (Strydom *et al.* 2002, Van Niekerk *et al.* 2012a) and have shown a decline in the abundance of fish species during high river flow conditions (Ter Morshuizen *et al.* 1996, Whitfield and Harrison 2003). Therefore, the diverse range of habitats present within the Breede Estuary (Carter 1983, Bohlweki SSI Environmental 2008) could explain why no trends were observed between the abundance of both estuary resident and estuary-associated marine species and mean monthly freshwater flow, until severe flooding occurred ($>250 \text{ m}^3 \text{ s}^{-1}$).

Severe flooding in the winter that preceded summer sampling in 2007 and 2009 lead to a decrease in the abundance of the majority of the most abundant species. Similarly, the fish assemblage structure of the Sundays and Swartkops estuaries (South Africa) took several months to recover from flooding (Marais 1982). In contrast, the abundance of *Gilchristella aestuaria* increased after these severe flood events. Similarly, *G. aestuaria* were more abundant in Lake St Lucia (South Africa) after severe cyclonic flooding (Martin *et al.* 1992), which suggests pulse spawning events resulting from freshwater input into estuaries (Newton 1996, Strydom *et al.* 2002). In addition, severe flooding can result in a loss of submerged aquatic plants (Henry *et al.* 1996, Sheppard 2010). Talbot *et al.* (1990) noted that *Zostera capensis* is sensitive to sedimentation and scouring associated with floods. This can lead to a

decline in the abundance of associated fish species (e.g. Hanekom and Baird 1984, Whitfield 1986, Whitfield 1999, Paterson and Whitfield 2000) as well as limiting the refuge available to fish taxa, which increases the likelihood of species being flushed from an estuary (Marais 1982, Lamberth 2003). Another factor that could have led to the decreased abundance of fish species in the Breede Estuary during severe flooding is the loss of food resources (e.g. Marais 1982, Plumstead 1990, Whitfield 1999). Decreased abundance of three demersal species *Croilia mossambica*, *Oligolepis acutipennis* and *Solea turbynei* in Lake St Lucia was attributed to the removal of sediments and the deposition of allochthonous litter and detrital material as a result of flooding, which likely resulted in the destruction of habitats and food items (Martin *et al.* 1992). Similarly, the export of silt, detritus and organic rich sediments into the marine environment as a result of river flooding in the Sundays, Gamtoos and Mbashe estuaries (South Africa) was believed to be the cause in the decrease in the abundance of mugilid species in these systems (Marais 1982, Marais 1983, Plumstead 1990).

Sea surface temperatures (SST) had no observable effect on the number of tropical species. *Monodactylus falciformis* and *Redigobius dewaali* were the only two tropical species consistently recorded throughout the study period, with *M. falciformis* being commonly recorded in South African warm-temperate estuaries (Harrison and Whitfield 2006). *Redigobius dewaali* is known to occur from southern Mozambique to the Breede Estuary (Smith and Heemstra 2003, Montoya-Maya and Strydom 2009). Tropical species are known to extend their distributional range into South African estuaries during the warmer summer months (Branch and Grindley 1979, Blaber 1981, Whitfield 1994b). The sporadic presence of some of the tropical marine species occurring in the Breede Estuary during this study could be the result of marine temperatures being outside the preferred thermal range of this biogeographic group resulting in them retreating to the warmer waters that prevail in estuaries during summer (Day 1981). Similarly, Hanekom *et al.* (1989) recorded an influx of species into the Keurbooms Estuary and Storms River mouth (South Africa) during periods of upwelling as the estuarine waters were warmer than the adjacent coastal waters. The highest tropical species abundance correlated with the warmest SST and the lowest abundance correlated with the coldest SST. Temperature has been shown to have a positive correlation with the abundance of tropical species (e.g. Cyrus and Blaber 1987, Harrison and Whitfield 2006).

The environmental variable most important in structuring the Breede Estuary fish community (i.e. salinity) can be altered through global climate change. This would have profound effects on the species diversity and composition of the Breede Estuary fish assemblage. Intensification of the hydrological cycle as a result of climate change would increase the incidence and magnitude of heavy precipitation events, as well as modify the frequency of dry and wet events (Fauchereau *et al.* 2003, Burroughs 2007). This would modify river flow (Burroughs 2007) into estuaries and therefore impact the salinity (Rudek *et al.* 1991, Whitfield 1999). The section of the Western Cape that incorporates the Breede Estuary is projected to see a decrease of up to 10% in annual precipitation by the end of the 21st century as a result of climate change (De Wit and Stankiewicz 2006, Engelbrecht *et al.* 2009). Lamberth *et al.* (2008) predicted that under various reductions in freshwater flow scenarios the Breede Estuary would shift from a freshwater rich system to a marine dominated system. These authors suggested that estuarine-resident, catadromous and freshwater species would become less abundant and estuary-dependent species such as *Argyrosomus japonicus* and *Pomadasys commersonnii* could collapse by 50% of historical numbers in association with an extreme river flow reduction of 64%. However, an increase in overall diversity is predicted for the Breede Estuary as warm-temperate and subtropical marine species become more diverse.

Warming of estuarine and sea surface temperatures (SST) is the result of warming air temperatures (e.g. Schumann *et al.* 1995, Rijnsdorp *et al.* 2009). Significant increases in the annual mean maximum and minimum air temperatures for Cape Agulhas and Cape St. Blaize on the southern cape coastline were recorded between 1960-2003 (Kruger and Shongwe 2004). Consequently, it is plausible that over the same time period sea surface temperatures have increased resulting in an increase in estuarine temperatures in the Breede Estuary. Increasing sea surface temperatures can result in shifts in the distributional patterns of species and the composition of species assemblages (Perry *et al.* 2005, Clark 2006), and increase species richness in temperate regions (e.g. Kennedy 1990, Daufresne and Boët 2007, Henderson 2007, Hiddink and Ter Hofstede 2008).

Conclusion

The high variability in the inter-annual relative species abundance and diversity in the Breede Estuary shows the importance of conducting medium to long-term studies on estuarine fish communities. The current medium-term study indicates that the Breede Estuary fish

assemblage remained relatively stable between 2002 and 2012 and highlights the fact that climate change trends may only become discernible once data for several decades is collated. Nevertheless, salinity was identified as the most important environmental variable responsible for changes in the summer fish assemblage and climate change is likely to affect this variable. Any anthropogenic changes to environmental variables such as temperature (marine and estuarine), flow rate and salinity would therefore alter the species composition, diversity and structure of the Breede Estuary fish assemblage.

CHAPTER 3

THE FISH COMMUNITY OF THE MBASHE ESTUARY

INTRODUCTION

Many tropical and temperate species reach their southern and northern distributional limit, respectively within South African estuaries in the subtropical/warm-temperate transition-zone (e.g. Plumstead *et al.* 1989, Plumstead *et al.* 1991, Harrison and Whitfield 2006). Very little information is however, available on the fish communities of estuaries occurring within the former Transkei region which fall within the subtropical/warm-temperate transition-zone (e.g. Plumstead *et al.* 1985, Wasserman *et al.* 2010, Whitfield and Baliwe 2013). The role of environmental variables in structuring fish communities in these estuaries is also poorly understood (Branch and Grindley 1979, Mbande *et al.* 2005, Pattrick *et al.* 2007, Wasserman *et al.* 2010). Consequently, studies need to be conducted on these transition-zone estuaries.

The Mbashe Estuary occurs within the Dwesa-Cwebe Marine Protected Area (MPA) and as such fishing is prohibited in the lower three kilometres of the system (MLRA, Act No. 18 of 1998). The Mbashe Estuary is the 28th ranked estuary in terms of conservation importance along the South African coastline (Turpie *et al.* 2002a) and is considered an important biodiversity hotspot. This estuary also provides an important nursery area for *Argyrosomus japonicus*, whose stocks have collapsed (Van Niekerk *et al.* 2012a).

The Mbashe Estuary ichthyofauna has been documented in once-off seine and gill net (Harrison *et al.* 1998) and angling (Venter and Mann 2012) surveys as well as a multi-year gill net study (Plumstead *et al.* 1989, Plumstead 1990). Only the study by Harrison *et al.* (1998) used seine nets to sample the ichthyofauna. In 2010, a multi-disciplinary study was initiated to study and predict the effects of climate change on the ecology of the Mbashe Estuary over the long-term as well as contributing towards gathering extensive baseline data on the biodiversity of this estuary. During spring (October) each year, the ichthyofauna, phytoplankton, macrophytes and benthic invertebrates are sampled together with physico-chemical variables. In this study, the first three years of ichthyofauna data were analysed to determine if the fish community is typical of those found in estuaries within the transition

zone and whether climatic variables (particularly rainfall) influence species composition and abundance.

MATERIALS AND METHODS

Study site description

The Mbashe Estuary (32°14'55.4"S, 28°54'03.7"E) is a permanently open, turbid channel-like estuary (Marais 1983, Plumstead *et al.* 1989, Plumstead 1990) that is located approximately 70 km north of the Kei Estuary (Figure 3.1). The estuary is located centrally within the Dwesa-Cwebe Nature Reserve which is situated between the Ntlonyane (32° 20' S; 28° 48' E) and Nqabara (32° 12' S; 28° 58' E) rivers and covers an area of 57 km² (Timmermans and Naicker 2002, Shackleton *et al.* 2007, Fearon 2010). The estuary is approximately 8km long and has a catchment of 6030 km² (Day 1981, Plumstead *et al.* 1989, Harrison *et al.* 2001), with a mean annual run-off (MAR) of 836x10⁶ m³ per year (Van Niekerk *et al.* 2012a). The estuarine bed at the mouth consists of sand and the rest of the estuary is covered by a thick layer of soft silt (Day 1981, Plumstead *et al.* 1989).

Most of the catchment area is characterized by natural vegetation (43%), which consists of grassland, thicket and forest. The remaining land cover in the catchment area is comprised of agriculture (25%), degraded land (28%) and rural residential development (4%) (Harrison *et al.* 2001). Poor catchment management, along with bad farming practices has increased soil erosion and has had a direct effect on sediment loads within the system (O'Keeffe 1989, Plumstead 1990). In 1984, a hydro-electric power station was constructed approximately 50 km upstream at Collywobbles (Plumstead 1990). The dam is currently silted up and only 10% of the original dam capacity is available for hydro-electric power (Matthews 2008). As a result of catchment degradation and freshwater abstraction, the flow regime of the Mbashe River has been altered (Matthews 2008).

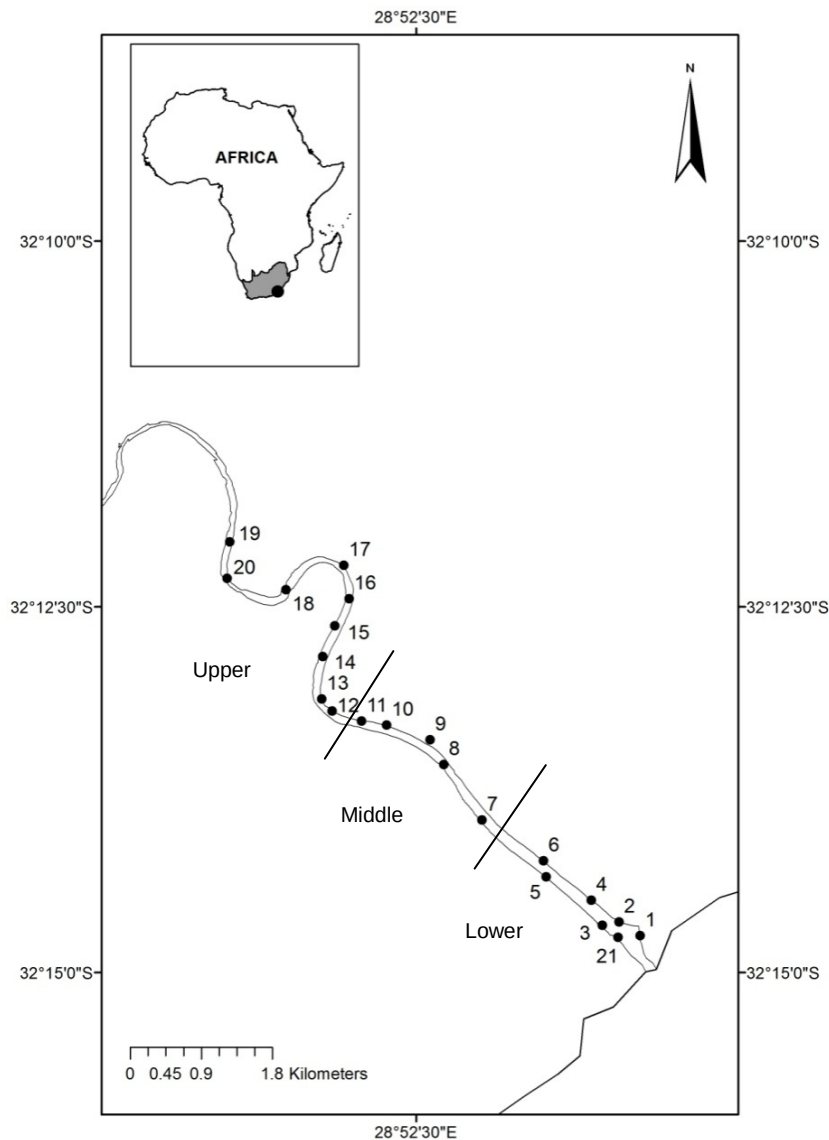


Figure 3.1. Map showing the location of the Mbashe Estuary and the location of the 21 sampling sites in the lower, middle and upper reaches.

Climatic conditions

Rainfall and river flow

Mean monthly rainfall (mm) values for the period 1972-2010 measured at the Dwessa Bos rainfall station (32 °16'59.88"S, 28 °46'59.88"E) in the Dwesa-Cwebe Nature Reserve were provided by the South African Weather Service. The mean monthly rainfall (mm) between 1972 and 2012 showed seasonal variation, with the wettest months occurring between October and April and the driest months occurring between May and September (Figure 3.2).

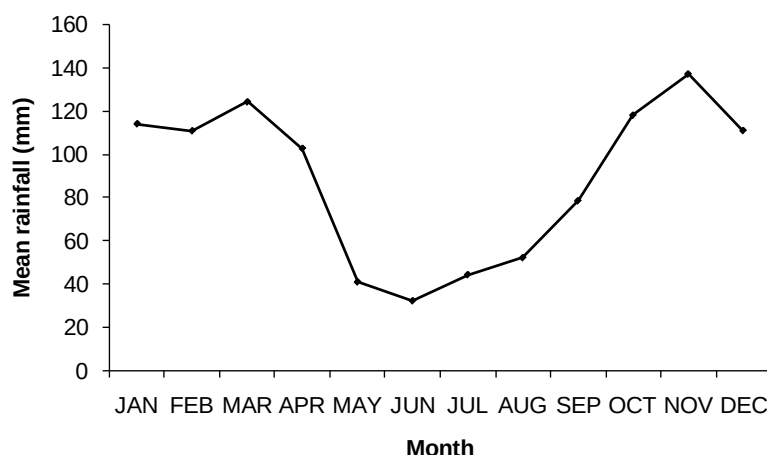


Figure 3.2. Mean monthly rainfall (mm) recorded at the Dwesa-Cwebe Nature Reserve between 1972 and 2012.

Hydrological data (freshwater flow rates) for the Mbashe River were obtained from the South African Department of Water Affairs (DWA) (www.dwaf.gov.za/Hydrology/). The verified mean monthly flow rate data (March 2006-December 2012) were obtained from the Rune T1H014 (31 °51'3.6"S, 28 °23'34.08"E) station. Mean monthly flow rates were compared to historical mean monthly values (2006-2012) to determine higher than average flow rates (above historical average) and lower than average flow rates (below historical average).

Air temperature

The Dwesa-Cwebe Nature Reserve has warm, humid summers and cool, frost free winters, with humidity being influenced by cold fronts that pass over the area during summer (Timmermans and Naicker 2002, Fearon 2010). The monthly mean air temperature (°C) values between 1970 and 2012 showed seasonal patterns, with the lowest temperatures occurring between May and September and the warmest temperatures occurring between November and April. The warmest month was February (23.0 °C), while the coldest month was July (17.5 °C) (Figure 3.3).

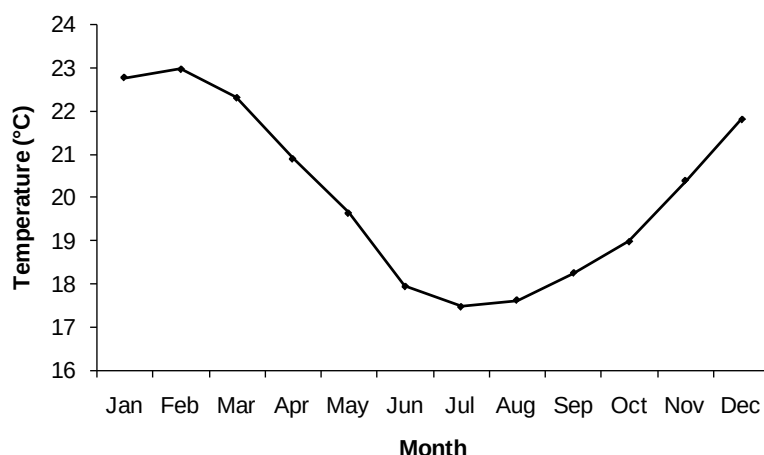


Figure 3.3. Mean monthly mean air temperatures (°C) recorded at the Dwesa-Cwebe Nature Reserve between 1970 and 2012.

Oceanographic conditions

At the eastern limit of the Agulhas Bank persistent upwelling occurs (Lutjeharms *et al.* 2000, Heileman *et al.* 2008). The core of this upwelling occurs at Port Alfred (Lutjeharms *et al.* 2000, Lutjeharms 2006) and coincides with a broadening of the continental shelf and increasing offshore location of the Agulhas Current. Upwelling is evident approximately 45% of the time at Port Alfred and is less frequent further north, with upwelling occurring as far as the Mbashe Estuary approximately 5% of the time (Lutjeharms *et al.* 2000). When upwelling occurs, inshore waters can be at least 5 °C colder than the surrounding shelf water (Lutjeharms *et al.* 2001) and the phenomenon influences nutrient availability and primary productivity (Lutjeharms *et al.* 2000).

Sea surface temperature (SST) values were obtained from the Pathfinder version 5.0 sea surface temperature (SST) dataset. The ocean pathfinder SST data set is a reprocessing of global SST data from the advanced very high resolution radiometer (AVHRR) on board NOAA (National Oceanic and Atmospheric Administration) satellites. Daily sea surface temperature (SST) values from Pathfinder data are available at a 4 km resolution and are accurate within 0.03 °C (Kilpatrick *et al.* 2001). Data were downloaded from the NOAA (National Oceanic and Atmospheric Administration) CoastWatch program (http://las.pfeg.noaa.gov/oceanWatch/oceanwatch_safari.php). Annual mean temperatures were calculated for the block 32° 13' 58"S, 29°1'29"E, 32 16' 37.2"S, 29°4'8.4"S incorporating the adjacent coastline for the period 1982-2012.

The monthly mean AVHRR (advanced very high resolution radiometer) SST (sea surface temperature) values showed clear seasonal differences and were lowest during winter (July-September) and peaked during summer (January-March). The month with the highest SST was February (25.0 °C) and the lowest SST value occurred in August (20.9 °C) (Figure 3.4).

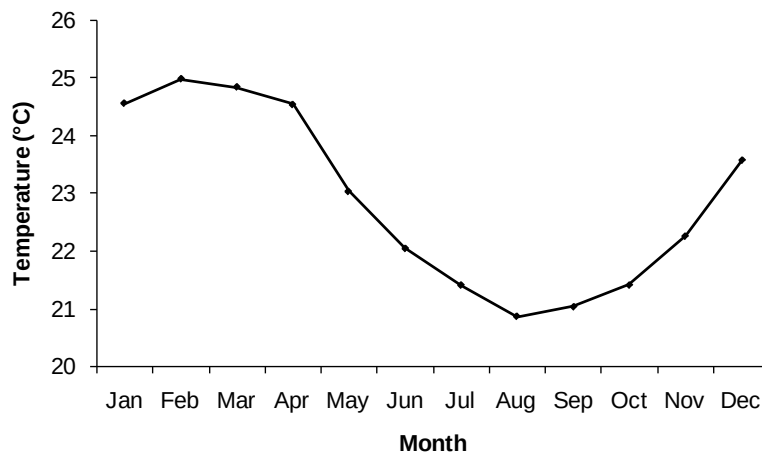


Figure 3.4. Monthly mean AVHRR (advanced very high resolution radiometer) sea surface temperature (SST) values recorded between 1982 and 2012.

Field sampling

During this study a seine net (30 m x 1.7 m x 15 mm bar mesh seine net with a 5 mm bar mesh purse and 1 m of 5 mm bar mesh on either side of the bag) was used to sample the littoral habitats of the Mbashe Estuary in October (spring) 2010, 2011 and 2012. Seine-netting was carried out in daylight hours. The Mbashe Estuary was sampled at 21 locations from the lower to upper reaches (Figure 3.1). The seine net was deployed using a small motorized boat, placed in a semi-circle formation offshore and subsequently hauled ashore by four people. Fish collected in the seine nets were identified, measured (mm standard length) and then returned to the system (alive whenever possible). If a fish could not be identified in the field it was placed into a labelled bag, preserved in (10%) formalin and identified in the laboratory using Smith's Sea fishes (Smith and Heemstra 2003). Where large catches of a species were made, only a sub-sample was measured, although all individuals were counted.

At each site, water temperature (°C), salinity (expressed as practical salinity units), pH, turbidity (NTU), Secchi depth (cm) and dissolved oxygen (mg l^{-1} and percentage) were recorded with a YSI multi-parameter meter.

Data Analysis

Environmental variability

Temperature (°C), salinity, turbidity (NTU) and dissolved oxygen(mg l⁻¹) recorded at each site in 2010 were used to determine the lower, middle and upper estuary reaches within the Mbashe Estuary using Principal Component Analysis (PCA) in the PRIMER v 6.1.6 (Plymouth Routines in Multivariate Ecological Research) package (Clarke and Warwick 2001). The sites were then placed into an estuary reach, based on the ordination diagram.

Comparisons between the means in temperature, salinity, turbidity and dissolved oxygen in the lower, middle and upper reaches were conducted using an analysis of variance (ANOVA). All four physico-chemical variables met the two assumptions of ANOVA (equality of variance and normality of the data set). If significant differences were detected during the ANOVA analysis, a post-hoc Tukey's test was conducted in order to determine which reaches were significantly different from each other.

Species composition and abundance

Data collected from the seine net were used to determine species composition and diversity over the three year period (2010-2012). Species were allocated an estuary-association category, according to the classifications given in Whitfield (1994a) (Table 2.1). Species were also grouped according to their biogeographical affinities, which were based on the six groups given in Harrison and Whitfield (2006) (Table 1.1).

Catch per unit effort (CPUE) was used as the index of relative abundance and given as fish per haul. The percentage contribution of individual species to the fish community was calculated by dividing each species CPUE by the total species CPUE. The percentage contribution of tropical, temperate and widespread species to the number of species and catch was calculated for each year (2010, 2011 and 2012).

Temporal trends

Annual trends of the fish assemblages of the Mbashe Estuary were analysed using the PRIMER v 6.1.6 (Plymouth Routines in Multivariate Ecological Research) package (Clarke and Warwick 2001). The seine net CPUE data from the three different reaches (lower, middle

and upper) in each year were log transformed to reduce the weighting differences between common and rare species. An association matrix was produced using the Bray-Curtis similarity measure. Analysis of similarity (ANOSIM) and an nMDS (non-metric multidimensional scaling) were carried out to determine if the annual fish assemblages were significantly different from each other. If significant differences in values were found, then a similarity percentages analysis (SIMPER) was used to identify the species primarily responsible for those differences.

RESULTS

Environmental variability

The monthly river flow rate values from March to October 2010 were below the average flow rate, with a peak monthly river flow rate of $21.4 \text{ m}^3 \text{ s}^{-1}$ occurring in March. The monthly rainfall during this same period only exceeded 50 mm twice (March and June) (Figure 3.5). Higher than normal monthly river flow and rainfall values were recorded from March to September preceding sampling in 2011. This suggests flooding of the Mbashe Estuary during this latter period, with river flow rate values, particularly in March, June and July, being at least three times higher than the historical monthly flow rate (2006-2012). Peak flow rates during this period occurred in March ($93.3 \text{ m}^3 \text{ s}^{-1}$) and July ($47.4 \text{ m}^3 \text{ s}^{-1}$), with monthly rainfall exceeding 150 mm in May and July 2011. During 2012, the flow rates between May and August were also above the historical mean monthly average, with peak flow occurring in August ($29.8 \text{ m}^3 \text{ s}^{-1}$) and peak rainfall in September (80.9 mm). These values were, however, lower than those measured in 2011 (Figure 3.5).

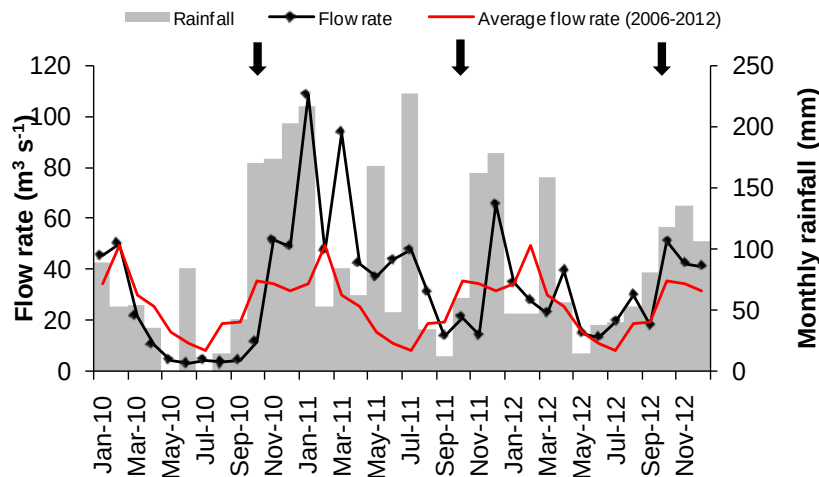


Figure 3.5. Variation in the monthly river flow rate and rainfall values between January 2010 and December 2012, plotted against the historical monthly flow rate values for the period 2006-2012 (arrows indicate sampling events).

Long-term air temperature data for the period 1970-2012 were measured at Port St Johns (Station 0129068A4) and provided by the South African Weather Service. No significant trends were recorded in the annual (a) maximum ($r^2 = 0.08$, $P > 0.05$), (b) minimum ($r^2 = 0.04$, $P > 0.05$) and (c) mean ($r^2 = 0.09$, $P < 0.05$) temperature values between 1970 and 2012 (Figure 3.6a, b, c). The period with the most significant increase ($r^2 = 0.59$, $P < 0.005$) in maximum air temperature values occurred between 1970 and 1983 (Figure 3.6a).

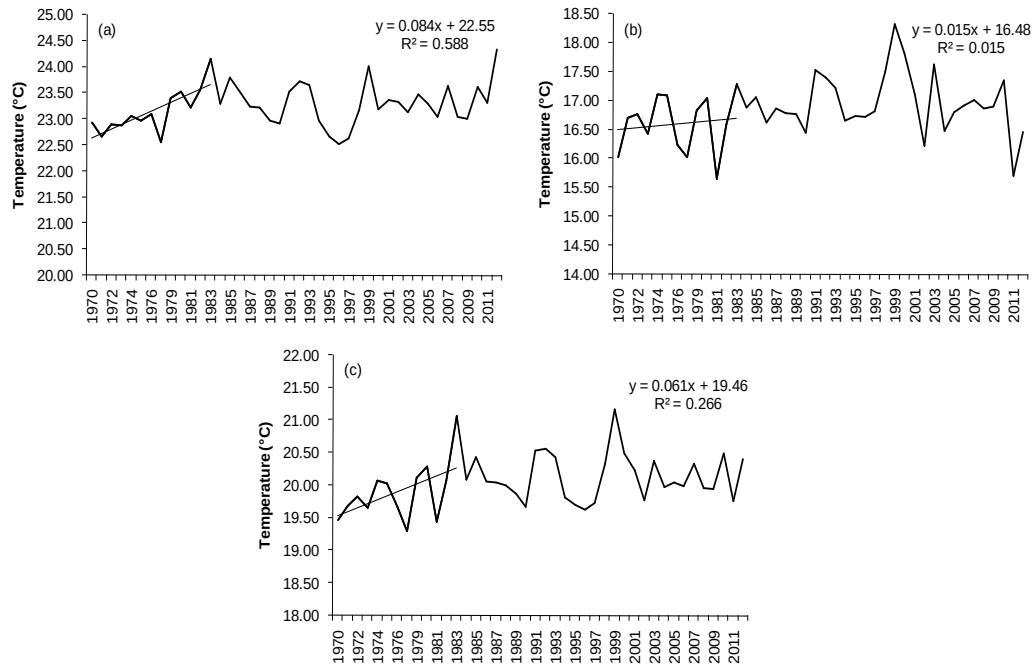


Figure 3.6. The (a) maximum, (b) minimum and (c) mean annual air temperature values recorded at Port St Johns for the period 1970-2012 (solid line = trendline between 1970 and 1983).

The mean annual AVHRR (advanced very high resolution radiometer) SST (sea surface temperature) values recorded between 1982 and 2012 showed an observable and significant increase ($r^2 = 0.52$, $P < 0.001$), with an increase of $0.06\text{ }^{\circ}\text{C}$ per year. The highest increase in SST values occurred prior to sampling between 1982 and 1991 ($r^2 = 0.74$, $P < 0.005$) with an average increase in SST values of $0.11\text{ }^{\circ}\text{C}$ per year. The annual AVHRR SST increased from $23.3\text{ }^{\circ}\text{C}$ to $23.5\text{ }^{\circ}\text{C}$ during the study period (Figure 3.7).

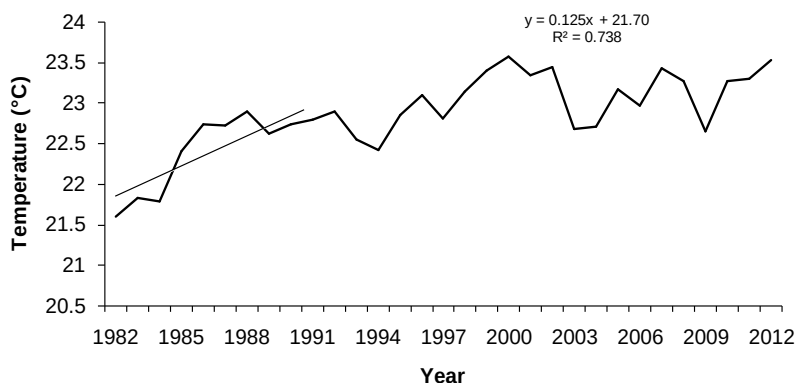


Figure 3.7. Mean annual AVHRR (advanced very high resolution radiometer) sea surface temperature (SST) values over the period 1982-2012 (solid line = trend line from 1982 to 1991).

The results of PCA for the four variables in 2010 revealed that the first two PC axes accounted for 98% of the variation between sites. The first axis was related to salinity and turbidity (NTU) and the second axis to temperature (°C) and dissolved oxygen (mg l⁻¹). Sites in the lower reaches (Sites 1-6, Site 21) were associated with high salinities, sites in the middle reaches (Sites 7-10) with high temperatures and sites in the upper reaches (Sites 11-20) with high turbidities (Figure 3.8).

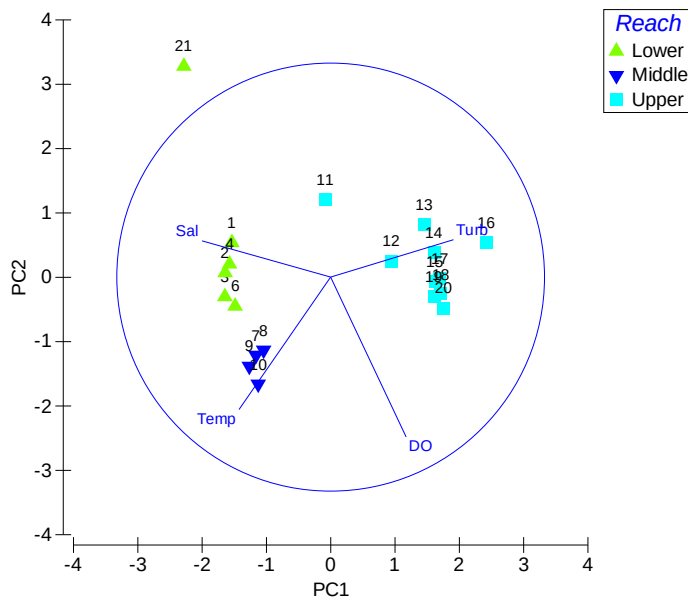


Figure 3.8. The results from the PCA ordination determining the upper, middle and lower reaches based on the temperature (Temp), salinity (Sal), turbidity (Turb) and dissolved oxygen (DO) at each site.

During the study period, mean turbidity values of 92.2 NTU (range = 22-219 NTU), 75.0 NTU (range = 27-145 NTU) and 158.8 NTU (range = 121-280 NTU) were recorded in the lower, middle and upper reaches, with a significant difference between the three reaches ($F = 12.18$, $P < 0.0001$). The post-hoc test (Tukey's test) determined that the mean turbidity in the upper reaches was significantly higher compared to the middle and lower reaches ($P < 0.05$) (Figure 3.9a). The mean salinity increased significantly from 0.3 (range = 0-3) in the upper reaches to 10.8 (range = 1-35) in the lower reaches ($F = 13.92$, $P < 0.0001$). The salinity values in the lower reaches was significantly higher than the middle and upper reaches (Tukey's test $P < 0.05$) (Figure 3.9b). Mean estuarine water temperature values of 22.1 °C (range = 18-25 °C), 23.1 °C (range = 22-25 °C) and 21.9 °C (range = 19-26 °C) were recorded in the lower, middle and upper reaches and were not significantly different from

each other ($F = 1.05$, $P > 0.05$) (Figure 3.9c). The mean dissolved oxygen values were 8.3 mg l^{-1} (range = 7–10), 8.7 mg l^{-1} (range = 8–10) and 8.4 mg l^{-1} (range = 7–9) in the lower, middle and upper reaches and were not significantly different from each other ($F = 2.53$, $P > 0.05$) (Figure 3.9d).

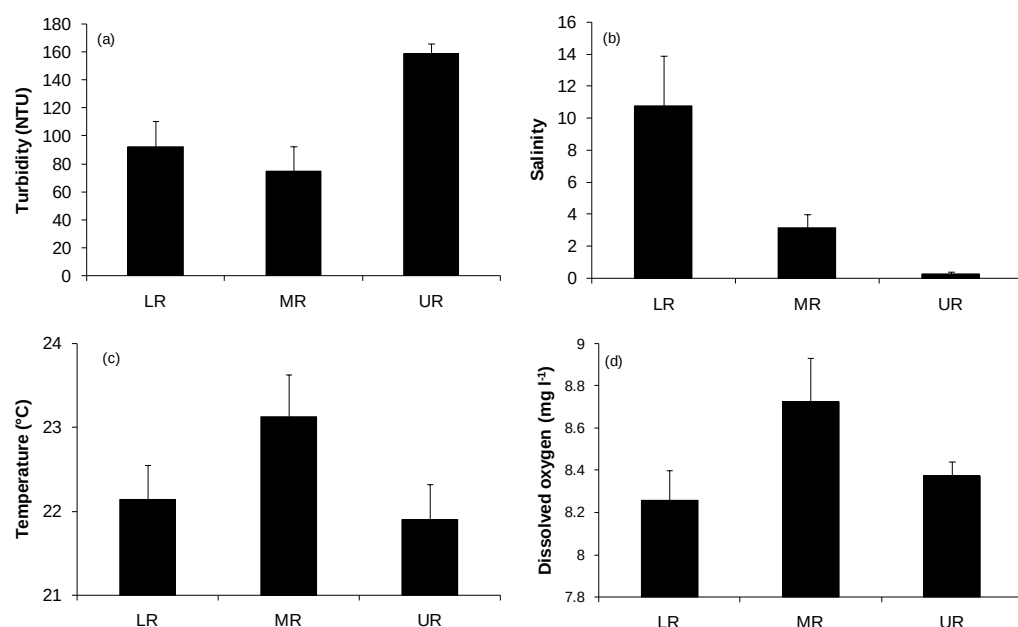


Figure 3.9. Mean ($\pm$ SE) environmental values of the three estuary reaches of the Mbashe Estuary between 2010 and 2012. LR-lower reach, MR-Middle reach and UR-Upper reach.

Species composition

A total of 34 species from 14 families were recorded in the Mbashe Estuary between 2010 and 2012. The number of species caught peaked in 2010 with 32 species, with only 14 and 15 species caught in 2011 and 2012, respectively. The most important families were Gobiidae and Mugilidae, with seven (21%) species each, followed by Sparidae with four species (15%). Out of the 34 species recorded, 15 (44%) are endemic to southern Africa (Table 3.1). Estuary-associated marine species (category II) accounted for 56% of all species recorded, followed by estuarine resident species (category I), with 32%. Only three marine straggler species (category III) were recorded: *Engraulis japonicus*, *Pomadasys olivaceus* and *Pseudorhombus arsius*. *Myxus capensis* was the only facultative catadromous species present (category Vb) and no freshwater species (category IV) were recorded during the study period (Table 3.1).

Overall, tropical and temperate species were equally abundant as they each accounted for 47% of the total number of species caught. Tropical species near their distributional limit (Groups 1 and 2) accounted for 32% (11) of all species recorded, while temperate species near their distributional limit (Groups 4 and 5) accounted for 24% (eight). Widespread species, namely *Gilchristella aestuaria* and *Mugil cephalus*, accounted for 6% of all the species recorded (Table 3.1).

The mean CPUE varied between the three years and peaked in 2010 with 171 fish haul⁻¹ and was lowest in 2011 with 46 fish haul⁻¹. The catch variability was greatest in 2012 and lowest in 2011 (Figure 3.10). The variable mean CPUE reflected the considerable variation of individual species each year (Table 3.1). The CPUE of *Gilchristella aestuaria* was lowest in 2010, with 9 fish haul⁻¹ and highest in 2012 with 97 fish haul⁻¹. *Atherina breviceps* were most abundant in 2010 (7 fish haul⁻¹), but not recorded in 2011 and 2012. The CPUE of *Rhabdosargus holubi* peaked in 2010, with 76 fish haul⁻¹. The CPUE of *Myxus capensis* increased from 0.1 fish haul⁻¹ in 2010 to 11 fish haul⁻¹ in 2012, while the peak abundance of *Argyrosomus japonicus* and *Solea turbynei* occurred in 2010 with 6 fish haul⁻¹ and 7 fish haul⁻¹ respectively (Table 3.1).

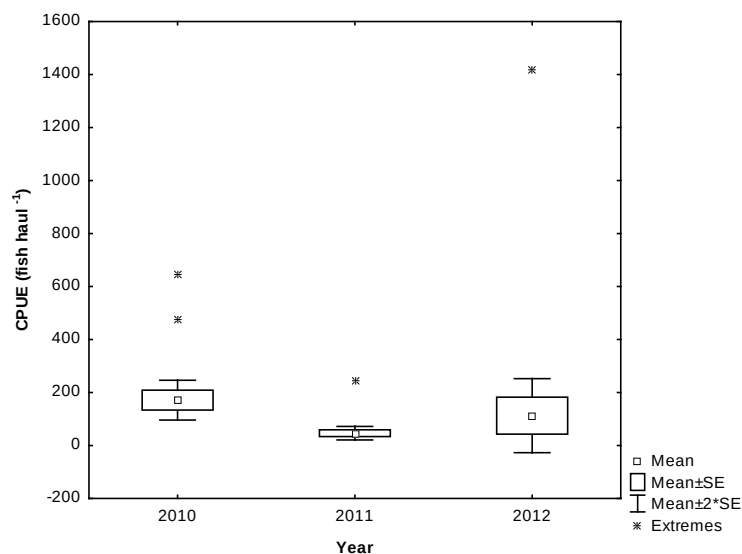


Figure 3.10. Box and whisker plot of the mean CPUE in the Mbashe Estuary between 2010 and 2012.

Table 3.1. Annual CPUE for each species, sorted by estuary-association categories, caught in the Mbashe Estuary between 2010 and 2012.

Family	Species	Est. Ass. Cat.	2010	2011	2012
Ambassidae	<i>Ambassis ambassis</i>	Ia	0.14	0	0
Clupeidae	<i>Gilchristella aestuaria</i> *	Ia	8.67	9.29	92.67
Gobiidae	<i>Oligolepis acutipennis</i>	Ia	0.29	0	0
Gobiidae	<i>Oligolepis keiensis</i>	Ia	4.33	0	0
Ambassidae	<i>Ambassis</i> spp.	Ia/Ib	0.05	0	0
Gobiidae	<i>Psammogobius knysnaensis</i> *	Ia/Ib	1.76	0	0.81
Atherinidae	<i>Atherina breviceps</i> *	Ib	7.05	0	0
Gobiidae	<i>Caffrogobius</i> spp.*	Ib	4.38	0.20	0.29
Gobiidae	<i>Glossogobius callidus</i> *	Ib	5.14	0.10	0.48
Gobiidae	<i>Trypauchen microcephalus</i>	Ib	0.10	0	0
Haemulidae	<i>Pomadasys commersonnii</i>	IIa	3.76	0.20	0.19
Monodactylidae	<i>Monodactylus falciformis</i>	IIa	0.57	0.19	0.05
Mugilidae	<i>Mugil cephalus</i>	IIa	0.19	0.62	1.62
Mugilidae	<i>Valamugil cunnesius</i>	IIa	0.62	0	0
Mugilidae	<i>Valamugil robustus</i>	IIa	0.05	0	0
Sciaenidae	<i>Argyrosomus japonicus</i>	IIa	5.95	0.10	2.43
Sparidae	<i>Acanthopagrus vagus</i>	IIa	0.43	0	0
Sparidae	<i>Lithognathus lithognathus</i> *	IIa	0.05	0.05	0
Sparidae	<i>Lithognathus</i> spp.	IIa	0.10	0	0
Sparidae	<i>Rhabdosargus holubi</i> *	IIa	75.67	14.60	0
Teraponidae	<i>Terapon jarbua</i>	IIa	0.19	0	0
Haemulidae	<i>Pomadasys</i> spp.	IIa?	9.95	0	0
Mugilidae	<i>Liza dumerili</i>	IIb	0.95	0.43	0.52
Mugilidae	<i>Liza tricuspidens</i> *	IIb	0	0.05	0.33
Soleidae	<i>Heteromycteris capensis</i> *	IIb	0.10	0	0.38
Soleidae	<i>Solea turbynei</i> *	IIb	7.43	0	0.48
Mugilidae	<i>Liza richardsonii</i> *	IIc	0	0.05	0.71
Platycephalidae	<i>Platycephalus indicus</i>	IIc	0	0.05	0
Sparidae	<i>Diplodus capensis</i> *	IIc	2.57	0	0
Engraulidae	<i>Engraulis japonicus</i> *	III	0.10	0	0.10
Haemulidae	<i>Pomadasys olivaceus</i>	III	1.76	0	0.10
Paralichthyidae	<i>Pseudorhombus arsius</i>	III	0.05	0	0
Mugilidae	<i>Myxus capensis</i> *	Vb	0.10	1.40	10.29

*Endemic to southern Africa

The catch composition of the dominant estuarine and marine species fluctuated considerably each year (Figure 3.11, Figure 3.12). A total of five taxa accounted for 93% of the estuarine catch in 2010, these were: *Atherina breviceps* (22.1%), *Gilchristella aestuaria* (27.2%), *Glossogobius callidus* (16.1%), *Caffrogobius* spp. (13.7%) and *Oligolepis keiensis* (13.6%) (Figure 3.11). In contrast, in 2011 and 2012 *Gilchristella aestuaria* accounted for more than 97% of the catch (Figure 3.11).

The marine catch was dominated by different species each year, namely: *Rhabdosargus holubi* (51%) in 2010, juvenile mugilids (52%) in 2011 and *Myxus capensis* (57%) in 2012. Interestingly, although *R. holubi* was an important component of the catch in 2010 and 2011, it was absent from catches in 2012. Similarly, Mugilidae comprised 20% of the catch in 2010, 52% of the catch in 2011 and only 7% of the catch in 2012 (Figure 3.12).

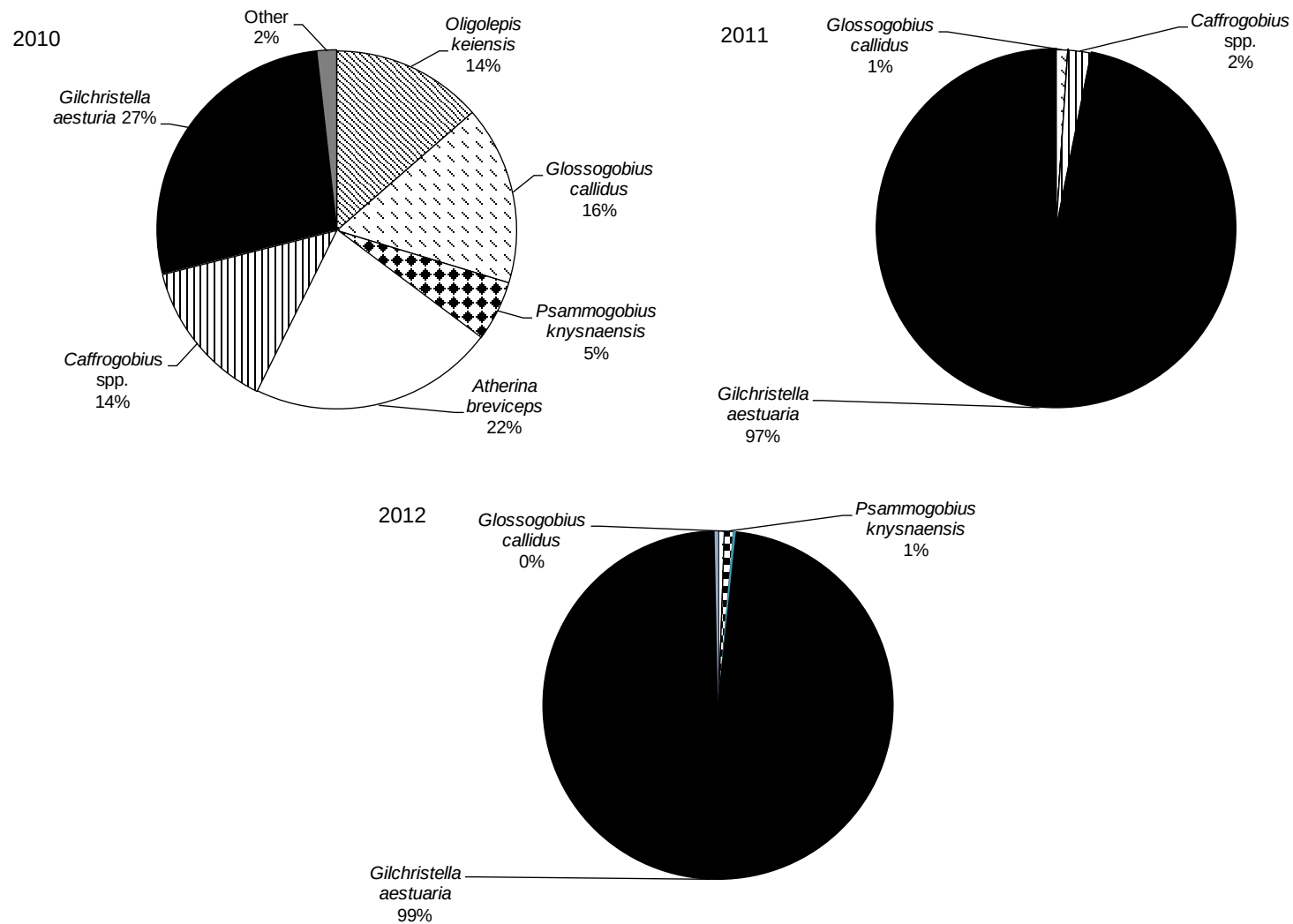


Figure 3.11. Catch composition (%) of the estuarine species recorded in the Mbashe Estuary between 2010 and 2012.

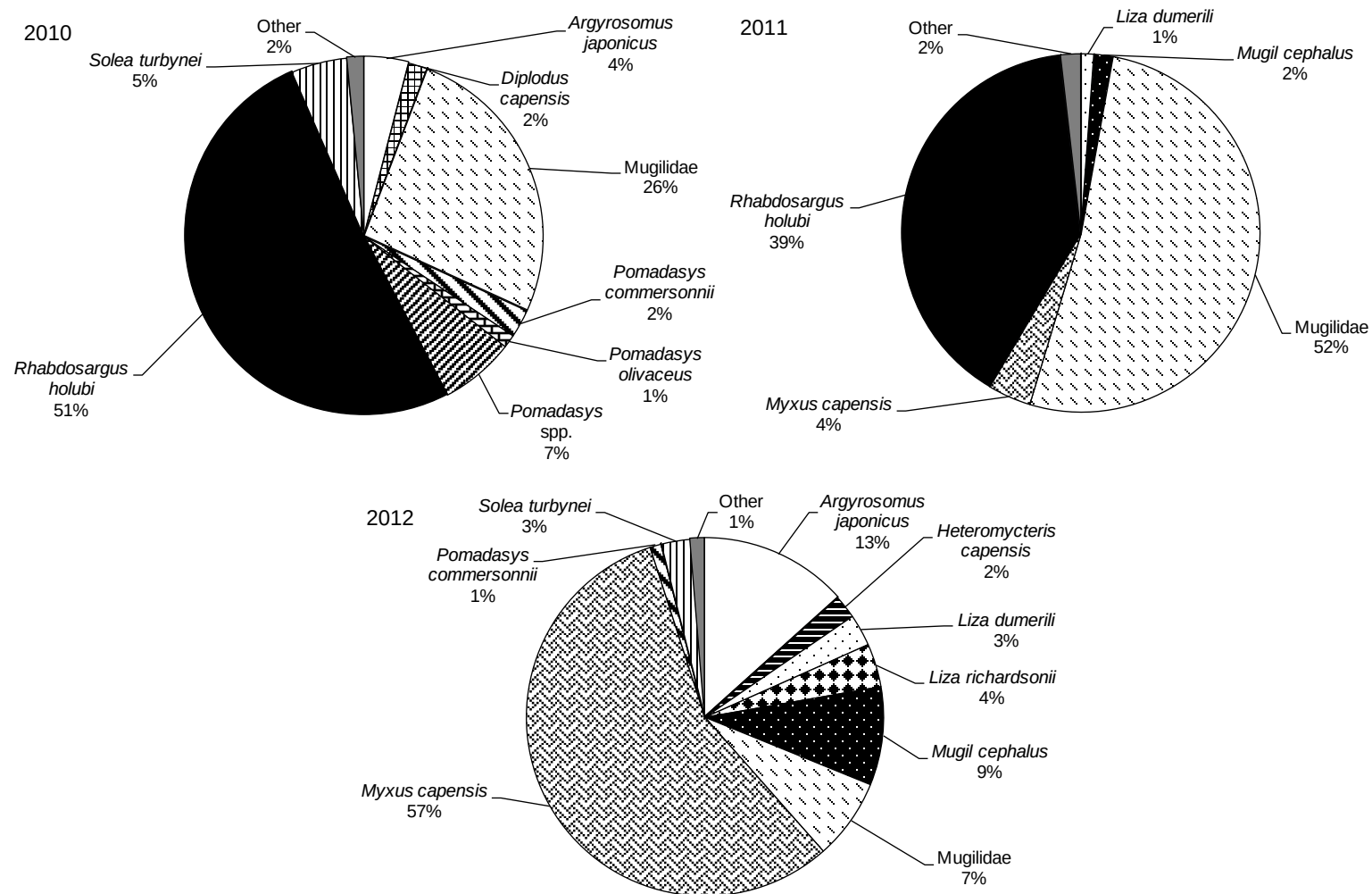


Figure 3.12. Catch composition (%) of the marine species recorded in the Mbashe Estuary between 2010 and 2012.

Temporal trends

The MDS ordination plot (Figure 3.13) separated the fish communities based on year. ANOSIM revealed significant differences in the fish communities between the three years (Global $R = 0.71$; $P < 0.005$). The highest dissimilarity occurred between 2010 and 2011, with an R -value of 0.96. The dissimilarity between 2010 and 2012 was also significant ($R = 0.82$). The low R -value (0.26) between 2011 and 2012 indicated a weak dissimilarity in the fish assemblages between these years.

SIMPER analysis revealed that 10 species were responsible for the differences in the fish assemblages between years: *Rhabdosargus holubi*, *Pomadasys* spp. *Solea turbynei*, *Oligolepis keiensis*, *Glossogobius callidus*, *Myxus capensis*, *Gilchristella aestuaria*, *Mugil cephalus*, *Argyrosomus japonicus* and *Pomadasys commersonnii*. Of these ten species, *R. holubi*, *O. keiensis*, *G. callidus*, *P. commersonnii*, *Pomadasys* spp., *S. turbynei* and *A. japonicus* were more abundant during 2010. *Gilchristella aestuaria* occurred in all years but was most abundant in 2012. The remaining species, *M. capensis* and *M. cephalus*, were largely absent from catches in 2010, with peak abundance occurring in 2012 (Figure 3.14).

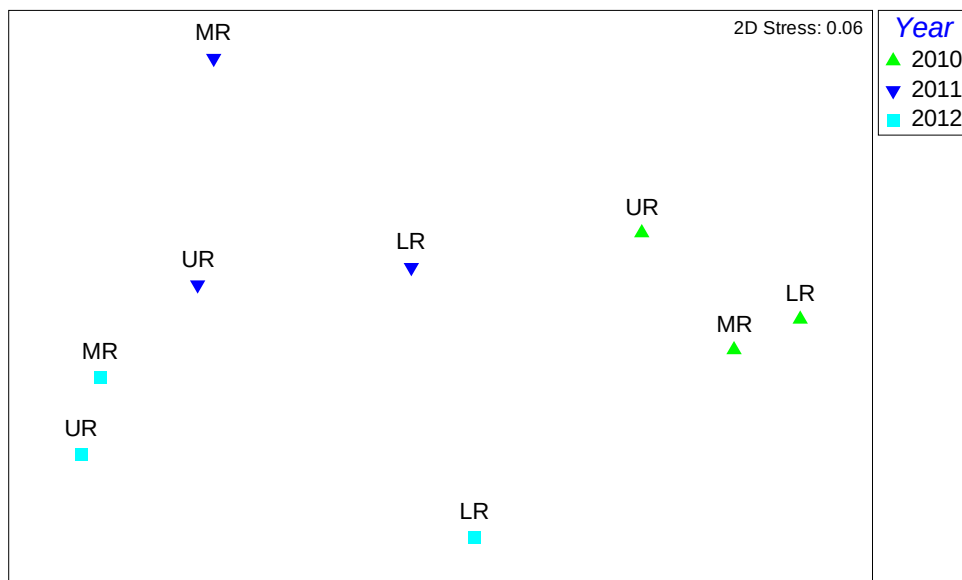
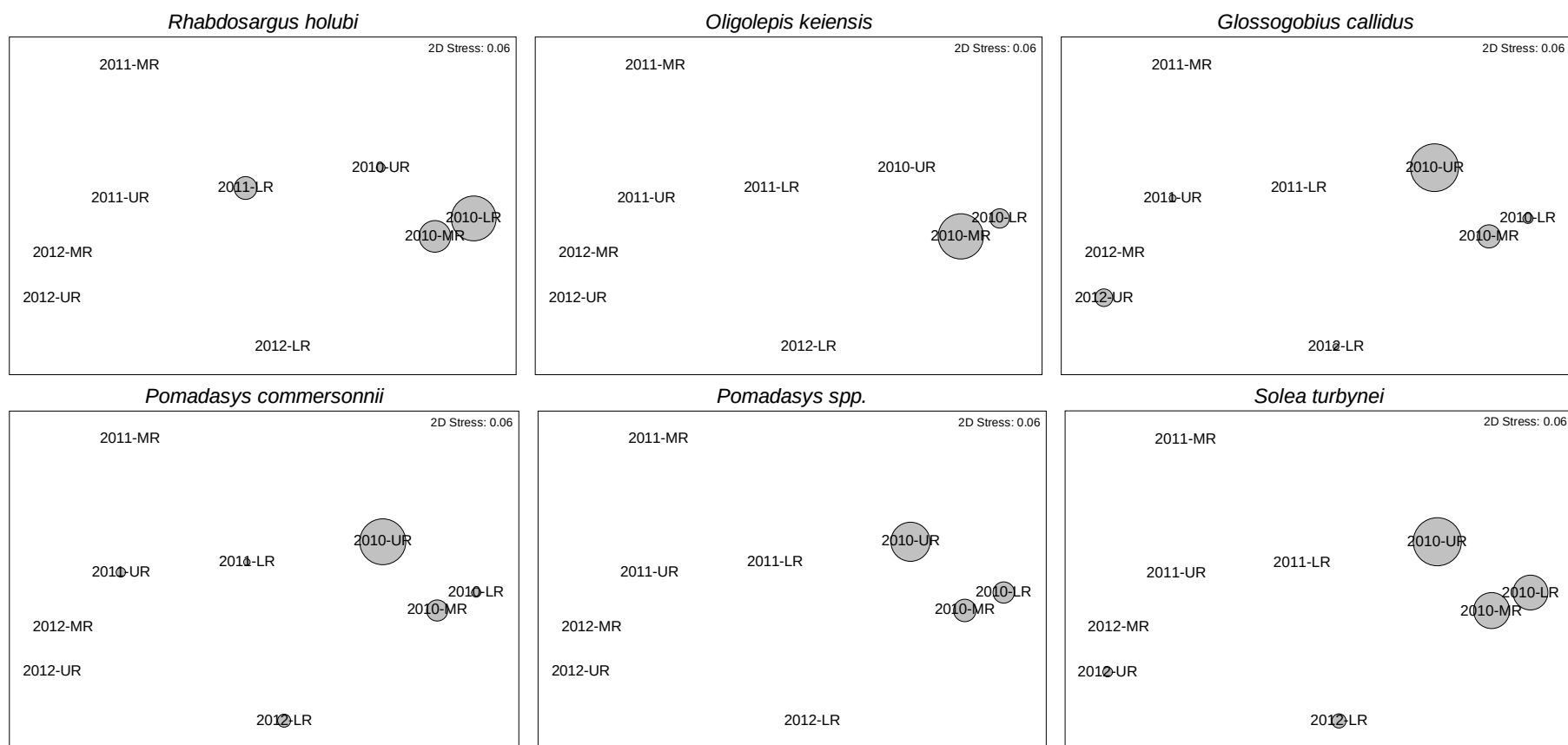


Figure 3.13. Multidimensional scaling (MDS) plot of the Mbashe Estuary fish assemblage in the three estuarine reaches for the period 2002-2012.



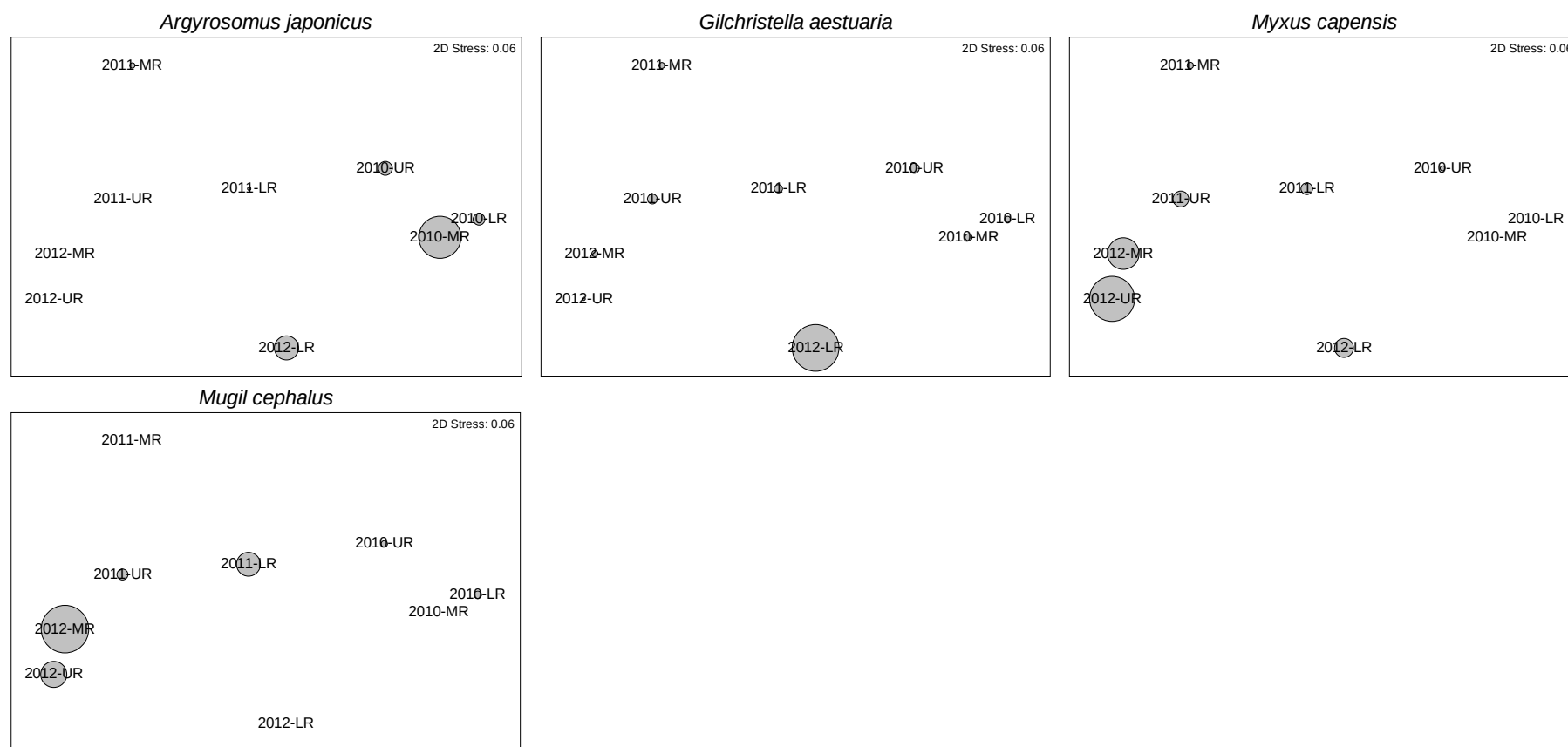


Figure 3.14. MDS ordination plots with superimposed relative abundances of 10 discriminating species as determined by SIMPER analysis along the upper, middle and lower estuary reaches for each year. Sizes of bubbles indicate relative abundance for that particular species, but are not comparable between species (UR-upper reaches, MR-middle reaches and LR-lower reaches).

DISCUSSION

Although authors have disagreed on the exact location of the estuarine biogeographic boundary between the subtropical and warm-temperate regions along the South African coastline (e.g. Day 1981, Whitfield 1994a, Maree *et al.* 2000, Harrison 2002), there is general agreement that it lies between the Great Kei Estuary and Port St Johns in the former Transkei (Harrison 2002). Maree *et al.* (2000) determined that the subtropical and warm-temperate transition-zone for estuaries incorporates an area where the contribution of tropical and temperate species is roughly equal, i.e. 50%. The overall proportions of tropical and temperate species were identical (47%) in the current study, which is indicative of the Mbashe Estuary being well placed within the subtropical/warm-temperate transition-zone.

Temperature affects the distribution and abundance of tropical fish species in South African estuaries (e.g. Cyrus and Blaber 1987, Turpie *et al.* 2000, Harrison and Whitfield 2006). Warming of sea surface temperatures as a result of climate change is likely to result in species or species assemblages shifting their distributional patterns, and this will be particularly noticeable at species geographic range boundaries (Myers 1998). Mbande *et al.* (2005) recorded an increase in the overall contribution of tropical species and a decrease in temperate species in the Mngazana Estuary (South Africa) in 2003 compared to a study conducted in 1975 (Branch and Grindley 1979). These findings were attributed to climate change (warming of marine and estuarine temperatures). The Mngazana Estuary is located ± 80 km north of the Mbashe Estuary and falls within the subtropical/warm-temperate transition-zone (Mbande *et al.* 2005). Similarly, warming temperatures have altered the number of species and relative contribution of warm-water and cool-water species in estuarine and marine waters (e.g. Barry *et al.* 1995, Cabral *et al.* 2001, Stebbing *et al.* 2002, Poulard and Blanchard 2005, James *et al.* 2008c). Figueira and Booth (2010) noted that if current warming of winter sea surface temperatures in temperate southeast Australia continues, four species of tropical damselfishes would be able to survive winter temperatures and expand their range as far south as Sydney by 2080. Although overall coastal sea surface temperatures off the Mbashe Estuary have increased by an average of 0.06 °C per year over a 30 year period, most of the warming occurred in the 1980s prior to any sampling in this estuary (Figure 3.7).

Climate change is also expected to increase the base flow and flood events in Transkei estuaries due to increased precipitation (Van Niekerk *et al.* 2012b) and this could have significant consequences for the Mbashe fish assemblage structure. Freshwater input into estuaries affects physico-chemical variables (e.g. Whitfield 1999, Kennish 2002) which in turn impacts fish abundance and distribution in estuaries (e.g. Ter Morshuizen *et al.* 1996, Loneragan and Bunn 1999, Bate *et al.* 2002, Vorwerk *et al.* 2008). Freshwater input also plays an important role in the recruitment of larval and juvenile estuary-dependent marine species into estuaries (Whitfield 1994b, James *et al.* 2008a). The quantity of river flow impacts benthic communities (Marais 1982, Chaino *et al.* 2006), which many estuarine fish species utilise as a food source. Floods can result in flushing of estuarine resident species from estuaries (Strydom *et al.* 2002) and also results in the possible removal of muddy sediments from turbid estuaries (Marais 1982). Changes in flow rates and the frequency of floods could impact the Mbashe Estuary fish communities through possible changes in recruitment, food resources, habitat availability and estuarine environmental variables.

An additional factor to consider is that changes in freshwater input into an estuary have been shown to affect the abundance of fish species in the marine environment. The monthly raw catch per unit effort (CPUE) of *Engraulis encrasicolus* recorded on the Ebre River continental shelf in the northwestern Mediterranean, between April and August, was linked to freshwater input into the Ebre River (Spain) (Lloret *et al.* 2004). Locally, Lamberth *et al.* (2009) recorded significant relationships between the abundance of 14 species caught on the Thukela Banks and flow from the Thukela River. Similarly, changing flow rates and frequency of floods due to climate change may have a similar impact on marine spawning by *Lithognathus lithognathus*. This species breeds in the surf-zone off the Transkei coastline in the vicinity of mudbanks (Bennett 1993), with the coastal area adjacent to the Mbashe mouth noted as the only confirmed spawning area (Van Niekerk *et al.* 2012a, Venter and Mann 2012). Changes in freshwater input could affect the extent of these fluvial fans, which may impact the breeding success or the extent of the spawning habitat available to this species, which is heavily exploited (e.g. Bennett 1993, Van Niekerk *et al.* 2012a).

In this study, species richness, abundance and community composition were significantly different between 2010 and the other years in the study, i.e. 2011 and 2012. The differences in the fish assemblages between these years are likely to be as a result of the differing freshwater input into the Mbashe Estuary prior to sampling. In the months preceding sampling in 2011, high rainfall and consequently high mean monthly river flow rates were recorded, which resulted in flooding of the Mbashe Estuary. Flooding of a channel-like estuary, such as the Mbashe Estuary can lead to scouring of the system which removes silt and organic rich mud (Plumstead 1990). This can disrupt or deplete the food web which would affect food availability and impact fish taxa within an estuary (Plumstead 1990, Whitfield and Harrison 2003). Channel-like estuaries also have narrow intertidal areas (Marais 1982, Beckley 1984, Bennett 1989, Schlacher and Wooldridge 1996b), which limits habitat variability. A lack of habitat diversity, especially the absence of sheltered side channels, means estuarine fauna are more likely to be flushed into the marine environment in this type of estuary (Marais 1982, Plumstead *et al.* 1985).

The lowest mean CPUE and species richness were recorded in 2011, which coincided with river flooding prior to sampling. Similarly, flooding of the Mbashe Estuary in 1985 was followed by a reduction in the total fish abundance, with fish species taking up to six months to return to pre-flood levels (Plumstead 1990). Similarly, the long-term effects of flooding have also been recorded in Lake St Lucia (Martin *et al.* 1992) as well as the Kei (Plumstead *et al.* 1985) and Gamtoos (Marais 1983) estuaries. Of the ten species responsible for the dissimilarity between the three years, *Rhabdosargus holubi*, *Oligolepis keiensis*, *Glossogobius callidus*, *Pomadasys commersonnii*, *Pomadasys* spp., *Solea turbynei* and *Argyrosomus japonicus* had peak abundances during the 2010 sampling period.

Rhabdosargus holubi is strongly associated with submerged aquatic vegetation (e.g. Whitfield 1986, Whitfield 1988, Paterson and Whitfield 2000). Submerged aquatic plants have been shown to be negatively impacted by floods through scouring and silt deposition (Day and Grindley 1981, Talbot *et al.* 1990, Adams and Talbot 1992) and these plants can take months to several years to recover to pre-flood levels (Hilmer *et al.* 1988, Talbot *et al.* 1990, Adams and Talbot 1992, Wortmann *et al.* 1997). The floods in 2011 could have negatively affected the abundance of *R. holubi* through a loss of the limited emergent aquatic vegetation present along the margins

of this system (Van Niekerk *et al.* 2012a). The lack of a recovery by these plants in 2012 could explain the absence of *R. holubi* in this year.

Ter Morshuizen *et al.* (1996) found that *Oligolepis keiensis* and *Pomadasys commersonnii* were most abundant in the Great Fish Estuary (South Africa) during low river flow, while the lowest numbers of *P. commersonnii* caught in the Mbashe Estuary between 1980 and 1988 occurred in the two years after river flooding in 1985 (Plumstead 1990). The abundance of the *Glossogobius callidus* and *Solea turbynei* within estuaries has been linked to sediment characteristics, with these two species preferring sediments with high silt content (Richardson *et al.* 2006). Similarly, *G. callidus* was more abundant in the Damba and Zotsha estuaries (South Africa), which are muddy systems, compared to the Mhlanga Estuary (South Africa) which has a mainly sandy mud substratum (Harrison and Whitfield 1995). *Solea turbynei* was also more abundant in the turbid muddy areas of Lake St Lucia (Cyrus 1991). The removal of these sediments into the marine environment, due to the heavy flooding that occurred in 2011, could remove or severely limit the habitat available to *G. callidus* and *S. turbynei*, and as a result they were largely absent from the catch in 2011.

The main food source of *Oligolepis keiensis*, *Pomadasys commersonnii*, *Glossogobius callidus* and *Solea turbynei* are benthic invertebrates (Harrison 2003, Mbande *et al.* 2005, Nhleko 2011). Scouring of the soft mud/silt from the Mbashe Estuary in 2011 could have led to a decrease in suitable benthic food resources which would affect the abundance of all four species. Similarly, Marais (1982) noted that benthic fauna may be flushed from channel-like estuaries during floods. Deposition and accumulation of this layer of silt and organic matter can take from months to years to accumulate after a flood (Plumstead 1990). A partial recovery of the fluid silt layer was noted in 2012 (N. James, pers. comm.) and this may explain why some benthic feeders, i.e. *Solea turbynei* and *Glossogobius callidus*, were recorded again in 2012.

Argyrosomus japonicus was most abundant in 2010 and recorded its lowest abundance (0.1 fish haul⁻¹) and contribution (0%) to the marine catch in 2011. The change in the relative abundance of this species during the study period could be due to a number of factors. The severe flooding in 2011 could have flushed the juveniles of this species out of the Mbashe Estuary due to the

limited refuge available within this system. Another possible explanation for the low abundance of *A. japonicus* in 2011 is limited food availability during that period. Juvenile *A. japonicus* are wholly dependent on estuaries (Whitfield 1994a) and mysids are an important component of the diet of juveniles less than 150 mm (total length) (Griffiths 1997). The floods in 2011 could have flushed the zooplankton (e.g. mysids) into the marine environment in this channel-like estuary. Similarly, flushing of estuarine zooplankton into the marine environment during floods has been recorded in the Mngazana Estuary and Richard Bay, South Africa (Grindley and Wooldridge 1974, Louw 2007). Studies conducted on South African estuaries have shown that zooplankton can fully recover after a few months of a flooding event (Grindley and Wooldridge 1974, Wooldridge 1986). The fact that the monthly flow rate values from March 2011-August 2011 were well above the historical monthly flow rate suggests that this could have prevented sufficient replenishment of zooplankton densities to support high numbers of juvenile *A. japonicus*. The lower flow rates in 2012 are likely to have resulted in an increase in zooplankton biomass, which may explain the slight recovery in the number of *A. japonicus* in that year.

The abundance of *Gilchristella aestuaria*, *Myxus capensis* and *Mugil cephalus* peaked in 2012. Peak abundance of *G. aestuaria* was associated with higher freshwater flow into Lake St Lucia as well as in the Kromme and Kariega estuaries (South Africa) following river pulses entering the systems (Martin *et al.* 1992, Strydom and Whitfield 2000, Strydom *et al.* 2002). James *et al.* (2008b) suggested the peak abundance of *G. aestuaria* (1662 fish haul⁻¹) during the period 1996–2005 in the East Kleinemonde Estuary could be linked to peak rainfall which occurred between December 2004 and December 2005 (1436 mm). Similarly, increased abundance in *G. aestuaria* could arise from the increased freshwater input into the estuary which has been indirectly linked to increased primary productivity which leads to an increase in zooplankton in the estuary (Martin *et al.* 1992). It has also been suggested that *G. aestuaria* adults pulse spawn in response to increased freshwater input (Newton 1996).

Myxus capensis is a catadromous (category Vb) freshwater mullet that is commonly recorded in the less saline reaches of estuaries and rivers (Bok 1979, Cowley and Whitfield 2001, Skelton 2001). *Mugil cephalus* is also found in freshwater, but is common throughout estuaries (Bok 1979, Whitfield 1990). The lowest abundance of *Myxus capensis* and *Mugil cephalus* occurred in

2010, which coincided with low river flow rates prior to sampling. A study on the effects of freshwater inputs on the fish assemblages in the Great Fish Estuary (South Africa) showed a similar trend, with the lowest abundance of *M. capensis* and *M. cephalus* being recorded under low flow conditions ($<10 \times 10^6 \text{ m}^3 \text{ month}^{-1}$) (Ter Morshuizen *et al.* 1996). The main food source of *M. cephalus* and *M. capensis* is detritus (Masson and Marais 1975, Blaber 1976, Blaber 1977, Whitfield 1988) and the peak abundance of these species in 2012 could be a result of the recovery in detritus and organic rich sediments within the Mbashe Estuary.

Conclusion

The data obtained in this study gives a preliminary understanding of the composition, diversity and relative abundance of the estuary-associated fish taxa that occur in the Mbashe Estuary. The Mbashe Estuary acts as an important nursery area for many estuary-dependent species, such as *Argyrosomus japonicus*, *Rhabdosargus holubi* and *Mugil cephalus*. Changing freshwater inputs to the Mbashe Estuary appear to be primarily responsible for the variation in the annual fish assemblage structure. Climate change, specifically changes in freshwater input and sea surface temperature could impact the abundance, composition and diversity of the estuary-associated fish species within this system. However, in order to determine the environmental variables responsible for structuring the estuarine fish assemblage, inter-annual variations in biodiversity as well as impacts of climate change, a much longer-term data set is required.

CHAPTER 4

GENERAL DISCUSSION

Environmental or climatic variables have been shown to affect the abundance and distribution of fish species within estuarine environments in numerous studies (e.g. Morin *et al.* 1992, Thiel *et al.* 2003, Barletta *et al.* 2005, Meynecke *et al.* 2006, Ramos *et al.* 2006), with species and ecological guilds showing specific responses to particular environmental variables (e.g. Cyrus and Blaber 1987, Marshall and Elliot 1998, Harrison and Whitfield 2006, Martinho *et al.* 2007). Taxa that occur within transition-zones are often near their distributional limit and the impact of abiotic factors on species is more easily understood (Myers 1998, Martinho *et al.* 2012). This is as a result of species being more variable in their occurrence and at the limit of their physiological tolerances at the edge of their distributional limit (Brown 1984, Kling *et al.* 2003). Therefore, estuaries within South African transition-zones should be ideally placed to monitor changes in environmental or climatic variables and their impacts on fish communities. The environmental or climatic variables responsible for structuring the fish assemblages in the Breede and Mbashe transition zone estuaries, were determined, with an emphasis on temperature, salinity and rainfall.

Temperature

Temperature is an essential factor that governs many processes in fish taxa (e.g. Fry 1971, Fonds *et al.* 1992, Planque and Frédou 1999, Pankhurst and Munday 2011). Consequently, species are found within a thermal range where physiological processes function optimally (Magnuson *et al.* 1979, Jobling 1981, Kling *et al.* 2003) and, as such, temperature affects the geographic distribution of these taxa (e.g. Clark *et al.* 2008, Harrison and Whitfield 2006, Pörtner and Peck 2010). During this study, the Breede and Mbashe estuaries recorded different proportions of the different biogeographic groups of fishes due to the influence of the Agulhas Current. Predictably, the Breede Estuary, which occurs in the cool-temperate/warm-temperate transition zone, was dominated by temperate taxa (Chapter 2), while the Mbashe Estuary, which occurs in the subtropical/warm-temperate transition zone, recorded a higher proportion of tropical species compared to the Breede Estuary (Chapter 3).

On the east coast of South Africa, the warm Agulhas Current flows in close proximity to the coastline and thus influences coastal sea surface temperature (Turpie *et al.* 2000, Harrison 2002, Mbande *et al.* 2005). The continental shelf widens past Port Alfred, at the eastern edge of the Agulhas Bank, with the Agulhas Current moving offshore in this area (Lutjeharms *et al.* 2000, Lutjeharms 2006, Heileman *et al.* 2008, Branch *et al.* 2010). Lutjeharms *et al.* (2000) recorded average sea surface temperatures of 19°C in Algoa Bay between 1961 and 1991. Sea surface temperatures along the Agulhas Bank show marked seasonality in values, with average temperatures between 15-19 °C, depending on locality (Schumann *et al.* 1995, Demarcq *et al.* 2003).

Maree *et al.* (2000) suggested that many tropical species are found at their southern distributional limit in Algoa Bay due to these species being intolerant of the lower temperatures associated with the Agulhas Current moving offshore. Whitfield (1994a) also found many tropical species such as *Acanthopagrus vagus* and *Caranx sexfasciatus* were rare west of Algoa Bay. Similarly, in tropical shelf regions, tropical species are often unable to survive average temperatures below 20 °C in the coldest months (Briggs 1974). Consequently, the higher proportion of tropical species occurring in the Mbashe Estuary when compared to the Breede Estuary was predictable.

Sea surface temperature (SST) also appeared to impact the total abundance of tropical species near their distributional limit in the Breede Estuary, i.e. peak abundance coincided with the warmest SST and the lowest total abundance coincided with the coldest SST (Chapter 2).

Salinity

Salinity is an important environmental variable determining the spatial distribution of fish assemblages in estuaries (e.g. Whitfield 1996, Martinho and Able 2003), with this environmental variable responsible for significantly influencing the Breede Estuary fish assemblage (Chapter 2). Fish species that are associated with South African estuaries are able to survive in a wide range of salinities (Whitfield 1996). However, salinity gradients, which are a major feature of many temperate estuaries (Marques 2006) have an effect on the composition and distribution of estuarine fish species (Cyrus and Blaber 1992, Thiel *et al.* 1995, Strydom *et al.* 2003, Akin *et al.* 2005). The Breede Estuary encompassed a full longitudinal salinity gradient, with salinity values

ranging from 35 in the lower reaches to 0 in the upper reaches, resulting in a higher species diversity in the lower reaches and lower diversity further upstream (Chapter 2, Figure 2.13). Similar gradients in salinity and fish diversity have been recorded by Loneragan *et al.* (1986) from the Peel-Harvey Estuary, in temperate Western Australia. Stenohaline marine species are also recorded in small numbers in the more saline lower reaches of some estuaries (Whitfield 1996) and this would further boost the species diversity in this reach when compared to the upper reaches.

The spatial distribution of species in the Breede Estuary according to salinity could be an important factor in the communities from each reach being significantly different from one another (Global $R = 0.76$, $P < 0.01$). Similarly, the spatial fish assemblage structure in the Mambucaba Estuary, Brazil, was primarily influenced by salinity, which resulted in the fish assemblages in each reach (lower, middle and upper) being significantly distinct from one another (Neves *et al.* 2011). Conversely, the absence of a normal salinity gradient in the warm-temperate Kariega Estuary (South Africa) resulted in no significant spatial changes being recorded in the littoral zone fish assemblages in this system (Ter Morshuizen and Whitfield 1994). However, there are other factors, besides salinity, that would also have influenced the spatial distribution of individual fish species in the Breede Estuary, e.g. substrate preference, turbidity and food availability.

Rainfall

Rainfall is the primary variable that affects the volume of water in a catchment (Schulze and Perks 1999, Lutjeharms *et al.* 2001), which in turn dictates the amount of freshwater that can enter an estuary, thereby influencing the composition and structure of estuarine systems (Sklar and Browder 1998).

Riverine water that is exported into the marine environment influences olfactory cues that are associated with the recruitment of certain euryhaline marine species into estuaries (e.g. Whitfield 1994b, Whitfield 1999, Martinho *et al.* 2009). Similarly, increased recruitment of estuary-dependent marine species i.e. *Argyrosomus japonicus* and *Rhabdosargus holubi* into the Kariega Estuary (South Africa) was associated with increased olfactory cues entering the adjacent coastal

area as a result of increased river flow rates (Vorwerk *et al.* 2008). Recruitment variability of *Solea solea* in the Mondego Estuary (Portugal) was linked to variation in river runoff (Martinho *et al.* 2009). Larvae accumulate in the adjacent coastal zone and need to locate the estuary which is facilitated by several factors, including olfactory cues that act as “point-source” stimuli for recruiting larval fish (Boehlert and Munday 1988). Support for the olfactory cueing hypothesis is provided by James *et al.* (2008a) who using a rectangular choice chamber showed attraction of postflexion *Rhabdosargus holubi* to estuary and river water, probably based on odour.

The presence of juveniles of species that are totally dependent on estuaries such as *Rhabdosargus holubi*, *Lithognathus lithognathus*, *Pomadasys commersonnii* and *Argyrosomus japonicus* in the Breede and Mbashe estuaries (Category IIa) suggests that these estuaries are functioning as nursery areas and that adequate olfaction and other cues are being exported into the adjacent marine environment to facilitate recruitment of these estuary-dependent species. Riverine flow also influences turbidity within an estuary (Whitfield *et al.* 1994), which provides juvenile fish with refuge from predators as well as increasing their feeding success (e.g. Gregory and Levings 1998, Whitfield 1999). Rainfall and the associated freshwater flow into estuarine systems are thus important for the successful functioning of these systems as fish nursery areas.

High rainfall events, which occurred prior to sampling in the Mbashe and Breede estuaries, lead to flooding of these systems. However, flooding had different impacts on the fish communities of these particular estuaries (Chapter 2, Chapter 3). Extremely high rainfall in the months prior to spring sampling in 2011 caused flooding of the Mbashe system, which lead to low mean total abundance and species diversity. In contrast, the previous year was associated with low spring rainfall and river flow, with this scenario corresponding to peak species diversity and mean total abundance (Chapter 3). Similarly, higher fish densities were recorded under low river flow conditions in the Tagus Estuary (Portugal) (Costa *et al.* 2007). The high rainfall and subsequent freshwater flooding that occurred in the Mbashe during 2011 and 2012 lead to interannual variation in fish species and mean total CPUE over the three year study period, with the fish communities in each year being significantly different from one another.

River flooding in the Breede Estuary did not appear to impact significantly on the fish assemblage structure (Chapter 2), with the exception of severe flooding prior to sampling in two of the years, which negatively impacted the abundance of the estuarine species *Caffrogobius* spp., *Psammogobius knysnaensis* and *Syngnathus temminckii* (Figure 2.14). The difference in the impact of river flooding in the two study estuaries could be partially due to estuary morphology and habitat availability. The Mbashe Estuary is channel-like and more susceptible to flooding, whilst the Breede Estuary has diverse habitats and a larger intertidal area. Additionally, the Breede Estuary falls within the winter/bimodal rainfall transition-zone and as a result experiences higher flow variability and therefore more high flow events. The above mentioned factors would reduce the physical effects of river floods in the Breede Estuary. Similarly, Marais (1982) showed that floods have different impacts on the fish fauna of the channel-like Sundays Estuary and the Swartkops Estuary, which has a wide floodplain and extensive fish refuge areas. Mugilidae increased in the Swartkops Estuary, probably due to a deposition of allochthonous detritus, while a major decrease in mugilid abundance occurred in the Sundays Estuary and was linked to the removal of detritus from the estuary into the marine environment by the floodwaters (Marais 1982).

High rainfall and associated river flooding did not negatively impact all fish species in the Mbashe and Breede estuaries. In the Mbashe Estuary, *Mugil cephalus*, *Myxus capensis* and *Gilchristella aestuaria* were more abundant under high river-flow conditions, with *G. aestuaria* becoming more abundant after severe floods in the Breede Estuary. Similarly, the abundance of *G. aestuaria* increased in Lake St Lucia as a result of torrential rains and subsequent flooding as a result of Cyclone Domoina (Martin *et al.* 1992).

Climate change

Amongst other impacts, global climate change is expected to alter the hydrological cycle and river and sea surface temperatures which will in turn impact the freshwater input and estuarine water temperatures in permanently open estuaries (Van Niekerk *et al.* 2012b). Global warming is expected to impact organisms during all life stages (Pörtner and Peck 2010) and would therefore impact community composition, abundance and diversity. Shifts in species distribution is the most commonly predicted scenario with global warming (e.g. Pörtner *et al.* 2001, Perry *et al.*

2005, Clark 2006), with these shifts often first documented at the edge of a species' distributional range (Pörtner and Peck 2010). Global warming is expected to alter rainfall patterns, increasing the magnitude of heavy rainfall events, but also the frequency and duration of drought periods (Fauchereau *et al.* 2003), with concomitant impacts on the distribution and frequency of freshwater pulses to estuarine systems.

The possible influence of climate change, especially with regards to temperature and rainfall, on the fish communities of the Breede and Mbashe estuaries is expected to differ. To determine any climate change related warming or cooling in sea surface temperature (SST) values, AVHRR (advanced very high resolution radiometer) SST values between 1982 and 2012 were analysed. Trends in the SST values during the 30 year period differed for each location. The AVHRR SST values for the Breede Estuary marine region did not show any warming (Figure 2.5), while significant warming was observed for the Mbashe Estuary marine region in the 1980s (Figure 3.7). These findings are similar to those recorded by Rouault *et al.* (2010) using Reynolds SST data. The above authors found warming of 0.55 °C for the Transkei region for all months of the year and no changes for the south coast between 1982 and 2009 during summer. This suggests potential climate change related warming for the coastal region adjacent to the Mbashe Estuary, but not the Breede Estuary. However, Smit *et al.* (2013) noted that satellite derived SST datasets show large biases compared to *in situ* temperature values recorded within 400 m of the South African coastline. Furthermore, a study by Dufois *et al.* (2012) showed that AVHRR Pathfinder v5 SST data was warmer than *in situ* measurements recorded off Cape Town. Consequently, satellite derived AVHRR SST Pathfinder data does not appear to give accurate SST values for inshore marine water temperatures. The use of AVHRR SST data to determine warming trends, and the possible impacts of these on fish communities, has limitations due to the bias of this type of data for nearshore localities.

Warming of the SST would lead to warming of the estuarine water temperature due to the permanent open mouth connection of the Mbashe Estuary to the marine environment. Tropical species are likely to increase their distributional limit and become more abundant and diverse in this system. Conversely, temperate endemics species such as *Atherina breviceps* and *Liza richardsonii*, which already contribute minimally to the overall marine and estuarine catch of the

Mbashe Estuary, could become rare or absent from this system due to higher temperatures. This would impact the overall composition of the Mbashe Estuary fish assemblage. Similarly, summer and early winter temperatures have increased significantly between 1950 and 2000 in Lake Ontario (United States of America) with continued warming predicted to affect the recruitment and fish community structure in this system. Warm-water species such as *Micropterus dolomieu* will record a threefold increase in recruitment with a 2 °C increase in water temperature with a simultaneous significant decrease in recruitment of cool-water and cold-water species (Casselman 2002).

Since the Breede Estuary falls within the warm-temperate/cool-temperate transition-zone and tropical species reach their distributional limit in this region (Harrison and Whitfield 2006), all tropical species recorded in this estuary were near their distributional limit. If warming of sea surface temperatures were to occur, tropical species are more likely to frequent the area and become more abundant in the estuary. However, Rouault *et al.* (2010) noted that cooling around the south coast (-0.35 °C) and for the Port Alfred/Port Elizabeth upwelling system (-0.40 °C) occurred between 1982 and 2009 during autumn and winter as a result of an increase in upwelling favourable winds or a combination of this and intensification of the Agulhas Current. The movement of many tropical species may therefore be limited to summer range extensions, as cooling along the south coast could act as a thermal barrier to southerly moving coastal tropical fish species. However, certain tropical species i.e. *Pomadasys commersonnii* and *Redigobius dewaali* have established populations in the Breede Estuary and have been recorded all year round (S. Lamberth, pers. comm.), and as such, suggests these two tropical species are potentially more tolerant of cooler temperatures and may occur in the system when other tropical species are absent.

It must be noted that during this study, most tropical species were not consistently recorded, which suggests that climate change is not having a significant impact on water temperatures in the region or that the sea surface temperature (SST) data must be longer than 30 years. Significant warming of annual mean air temperatures has been recorded in localities on either side of the Breede Estuary between 1960 and 2003 (Kruger and Shongwe 2004), and this warming would be expected to increase Breede River water temperatures entering the estuary.

Climate change is expected to increase rainfall and the frequency of flood events in the Transkei region (Van Niekerk *et al.* 2012b), which would impact on the abundance and composition of the associated estuarine fish communities. Higher river flow rates entering the Mbashe Estuary may limit the intrusion of saline marine waters into the system which would, in turn, affect the distribution and diversity of estuary-associated species. River flooding of the Thukela Estuary in Kwazulu-Natal resulted in a temporary loss of marine species abundance and diversity due to freshwater conditions prevailing throughout the system (Whitfield and Harrison 2003).

Increased river flood frequency could increase scouring of muddy sediments and removal of certain estuarine biota and their food resources. This would affect the food web and the fish communities within the estuary. Another possible impact of increased rainfall is higher suspended silt levels and increased sedimentation within the Mbashe Estuary. This could negatively impact the abundance and diversity of fish species within this estuary. Many fish species recorded in the turbid Mbashe Estuary are abundant in turbid estuaries e.g. *Gilchristella aestuaria*, *Solea turbynei* and *Argyrosomus japonicus* (Cyrus and Blaber 1987, Whitfield 1998). However, increased silt loads in the Mbashe River could trigger mass mortalities of fishes in the Mbashe Estuary, especially if excessive suspended sediment were to accompany an episodic flood event. Floods that carry high suspensoid loads can cause mass mortalities to both estuarine-resident and marine species in South African estuaries (Whitfield and Paterson 1995). Increased silt deposition could also occur following dissipation of the flood waters and this could change the sediment profiles within the estuary, resulting in a more homogenous muddy habitat occurring throughout the estuary. This could have major implications for fish species that are associated with different bottom sediments (Blaber 1976, Blaber 1977).

The predicted decrease in baseflow and rainfall in the southern Cape region is expected to result in the penetration of saline waters into some of the larger permanently open estuaries within this region (Van Niekerk *et al.* 2012a). This may lead to an increase in the abundance and species richness in these estuaries due to marine waters extending further upstream, which has been predicted under low-flow conditions in the Breede Estuary (Lamberth *et al.* 2008). Similarly, as a consequence of predicted reduced freshwater input in the Breede Estuary, the abundance of certain estuarine-resident species and estuary-dependent species could decrease (Lamberth *et al.*

2008). Certain submerged macrophytes (e.g. *Zostera capensis*) could expand their distribution as these plants prefer higher salinities (Adams *et al.* 1992, Adams and Bate 1994). This would increase the nursery habitat for many vegetation-associated species such as *Rhabdosargus holubi*, *Syngnathus temminckii* and *Clinus superciliosus*.

A study by Turpie *et al.* (2002b) noted that the total value of estuarine fisheries was estimated to be R463 million in 2000 with an equivalent worth of R909.8 million per year in 2013 using the average annual inflation which is based on the CPI (consumer price index). Climate change is expected to result in a decrease in the total estuarine catch by 35% in 2050, which will result in a loss of R318.3 million per year based on the 2013 values. Any changes in species composition and abundance of estuarine associated species could therefore have economic consequences.

The Breede Estuary is a nursery area for many important fish species and is the 18th ranked estuary along the South African coastline in terms of conservation importance (Turpie *et al.* 2002a). The predicted reduction in rainfall as a result of climate change in the southern Cape will negatively impact the amount of freshwater available to the Breede Estuary fish community. The National Water Act (Act No. 36 of 1998) states that a freshwater reserve needs to be determined to sustain the health of aquatic ecosystems. The National Water Resources Strategy, in line with the National Water Act, prioritises the Freshwater Reserve for aquatic systems. It is important that the Department of Water Affairs, scientists and other affected parties regularly re-assess that there is sufficient water for the sustainable ecological functioning of the Breede Estuary, especially if reduced rainfall occurs as a result of climate change and compounded by increased freshwater abstraction by existing schemes in the catchment.

Recommendations

In order to get a better understanding of the species within systems, a variety of sampling methods need to be employed as different gear types target different species (Harrison and Whitfield 1995, Lazzari *et al.* 1999). Therefore, the fish community of the Mbashe Estuary needs to be sampled using a variety of gear types, e.g. seine nets, trawls, hook and line. Further research is also required to determine the seasonal fish assemblage structure over a long-term period for both estuaries, especially as seasonal differences in environmental variables have been

shown to affect estuarine fish assemblages (Morin *et al.* 1992, Marshall and Elliot 1998, Hagan and Able 2003, Jaureguizar *et al.* 2004). Long-term monitoring of physico-chemical, plant, invertebrate and fish community characteristics in the above estuaries would also provide information on the impact of climate change and longer-term weather events such as ENSO (El Niño-Southern Oscillation) on the composition and structure of biotic assemblages and their food sources in these two transition-zone estuaries. The monitoring programme in the Mbashe Estuary is currently ongoing and the data collected on the physico-chemical, ichthyofauna, benthic invertebrates, macrophytes and phytoplankton needs to be re-analysed once another 5-10 years worth of data has been collected.

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